Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

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# Chapter One: Introduction and Approach

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# 1. INTRODUCTION AND APPROACH

## Key points

- The Gnangara groundwater system area covers approximately 2200 $km^2$, supplies over 60% of Perth's public water and supports a significant component of the biodiversity of south-west Western Australia, which is recognised as a global biodiversity 'hotspot'.
- The sustainability of the Gnangara groundwater system is under threat because of declining groundwater levels. The decline has been attributed to climate change, increased abstraction and interception loss from pine plantations.
- A multi-agency government taskforce was established in 2007 to develop the *Gnangara Sustainability Strategy* (GSS), which aims to ensure sustainable use of water and protection of the environment, and to develop land, environment and water management options to achieve these aims.
- Historical anthropological changes have resulted in declines in the extent of vegetation communities, as well as extinctions and declines of flora taxa, vertebrates (particularly mammals), and aquatic invertebrates on the Gnangara groundwater system.
- Despite these declines, the Gnangara groundwater system encompasses the largest areas of contiguous bush on the Swan Coastal Plain, significant threatened species and ecological communities and highly diverse groundwater-dependent ecosystems including wetlands, caves and tumulus mound springs.
- The major threats to biodiversity have been identified as climate change (declining rainfall and groundwater), groundwater abstraction, habitat loss and fragmentation, inappropriate fire regimes, the plant pathogen *Phytophthora cinnamomi* and introduced weeds and predators.
- The scope of this report is to determine current knowledge of the distribution and status of biodiversity values on the Gnangara groundwater system, and the risks, impacts and occurrence of identified threatening processes.
- This report reviews and summarises information obtained from previous studies, together with studies undertaken during the GSS project (2007–09), and proposes recommendations and priorities for conserving and protecting biodiversity at landscape and community levels, and for managing threatening processes.

# Introduction

## The Gnangara Sustainability Strategy

In response to growing concerns about future rainfall and groundwater declines in the Perth region, the Gnangara Sustainability Strategy was established to provide a framework for balancing water, land and environmental issues and to develop a water management regime that is socially, economically and environmentally sustainable. A multi-agency taskforce was established in 2007 to undertake the project which incorporates reviews of existing land and water use policies, commencement of a range of studies on the complex system, and development of a decision making process to integrate values, risks and planning processes in regards to the Gnangara groundwater system (Government of Western Australia 2009b). The taskforce, led by the Department of Water (DoW), includes the Department of Agriculture and Food (DAFWA), Department of Environment and Conservation (DEC), Department of Planning and Infrastructure (DPI), Forest Products Commission (FPC), Water Corporation and CSIRO (Appendix 1). The Gnangara Coordinating Committee (GCC), comprising of senior officers from each of the key agencies, and observers from the City of Wanneroo, City of Swan and Swan Catchment Council, oversees the work of the taskforce (Appendix 1).

The Gnangara groundwater system consists of several different aquifers on the Swan Coastal Plain. The groundwater system is recharged directly by rainfall. It covers an area of approximately 2200 $km^2$ and underlies the city of Perth as well as seasonal and permanent wetlands, pine plantations and extensive areas of native *Banksia* woodlands of the northern Swan Coastal Plain. Approximately 60% of Perth's water is obtained from the system, and declining rainfall and runoff in the past 30 years have had a major impact on water availability, and the ecosystems that depend on it.

The groundwater level of the Gnangara groundwater system is determined by past and future climate, land use, land management (e.g. prescribed burning regimes, thinning of pine plantations) and the amount and timing of water extraction. Declining groundwater levels have resulted from declining rainfall, resulting in less recharge into the aquifers. Other contributing factors to declining groundwater levels include groundwater abstraction for public and private uses and water use associated with pine plantations. Population growth in metropolitan Perth has also increased demands on the system, which are expected to continue for the next 20 years.

Examples of the relative impacts of these processes on groundwater levels are illustrated in the 2005 Department of Environment's 'State of the Gnangara Mound' report. Other reports on land use and water management in the study area include:

- *East Wanneroo Land Use and Water Management Strategy*. Western Australian Planning Commission (2004)
- *Section 46 Progress Report – State of the Gnangara Mound*. Department of Environment (2005b)
- *Environmental management of groundwater abstraction from the Gnangara Mound 2000–2003* (Water and Rivers Commission 2004)
- *Managing a sustainable future for the Gnangara groundwater resources: analysis of stakeholder issues and perspectives* (Department of Environment 2005a)
- *Gnangara Mound Strategic Planning Workshop* (May 2005) at CSIRO, Floreat
- *Identification of economic values associated with the groundwater of the Gnangara Mound*. Marsden Jacob Associates (2006)
- *In situ social values of groundwater-dependent features on the Gnangara Mound*. (Beckwith Environmental Planning Pty Ltd 2006)
- *Gnangara groundwater areas – Water management plan* (Draft for public comment) (DOW 2008a)
- *Review of ministerial Conditions on the Groundwater Resources of the Gnangara Mound* (Draft for public comment) (DOW 2008b)
- *Gnangara Sustainability Strategy Situation Statement*, Government of Western Australia. (2009b).

The results of these investigations and reports established that water sustainability of the Gnangara groundwater system was of great concern, particularly when considered with regard to predicted climate change scenarios. Further, under conditions of significantly lower rainfall this decade, protection of environmental values of groundwater-dependent ecosystems had not been fully achieved, and there had been a failure to meet Ministerial conditions at many designated sites. As a consequence of these findings and predictions, the Government of Western Australian committed $7.5 million to develop the GSS. The strategy is an across-government initiative established to develop an action plan that will recommend options to ensure the sustainable use of water for drinking and commercial purposes and to protect the environment and its biodiversity. The substantial funding for projects under the GSS is indicative of the significant body of research and technical work that underpins the recommendations contained within the GSS.

The *Gnangara Sustainability Strategy Situation Statement* (Government of Western Australia 2009b), was also developed, to provide background information on the context and issues addressed by the strategy and should be consulted for more information on those topics not covered in this report. It describes the Gnangara groundwater system, historical land and water uses, and environmental, social and economic dependencies within the system in the context of climate variability. It also discusses the current governance arrangements for land use and water management.

The GSS taskforce has adopted a consultative approach to ensure that issues such as land, water and biodiversity on the Gnangara groundwater system are considered by the community. A number of climate scenarios and management options, including the environmental, economic and social consequences, will inform decision making.

As a result of the work of the GSS a number of proposed land use options will be presented to the community in a draft strategy in 2009 (Government of Western Australia 2009a). These may include revising groundwater allocation to public and private water supplies or other options, recycling wastewater for other uses, assessments of alternative land uses after the pine plantations are cleared and changing vegetation management such as burning *Banksia* woodland to increase recharge. Following this, preferred options to ensure long-term sustainable use of water for consumption and to protect the environment will be recommended to the state government. In addition to the draft strategy, the *Gnangara groundwater system zone plans* will be presented concurrently (Government of Western Australia 2009a). These provide details of land use and water management at a more local level. For the purposes of developing the strategy the study area was subdivided into 30 subareas (see Figure 1.3), and these subareas were grouped into seven zones according to land use similarities. The zone plans provide an overview of each zone, highlight the major issues and list the GSS recommended options.

One of the challenges involved in developing a land and water use management plan for the study area is the strong interconnectedness between land uses and the hydrological balance, which in turn affects water yields and the health of water-dependent and other terrestrial ecosystems. The development of groundwater models such as the Perth Regional Aquifer Modelling System (PRAMS) has made a significant contribution to understanding water allocation and land use on the Gnangara groundwater system. PRAMS is a numerical modelling tool that can provide predictions of relative groundwater level changes at a regional scale. It has been used to provide information on changes in water levels across the Gnangara groundwater system and has provided valuable information for reviews of conditions such as abstraction and rainfall responses of the groundwater (Department of Environment 2005a; b; Vogwill *et al.* 2008). The model has been used to predict the relative influence of climate, abstraction (public and private) and land use (pines and native vegetation). It is a dynamic, spatially distributed model in which rainfall, land use, vegetative water demand and abstraction are interrelated.

In contrast, our current understanding of biodiversity values, ecosystem processes and dynamics of the Gnangara groundwater system, particularly at landscape scales, are inadequate (Government of Western Australia 2009b). Gaps in our capacity to measure impacts on biodiversity, landscape condition and ecosystem processes as a result of disturbances (climate change, changed water regimes, fire, plant pathogens) are likely to result in ineffective management actions and low quality outcomes. The ability to develop successful planning relies on the quality of the biodiversity information (Pressey 1999; Wilson *et al.* 2005). Indeed, unless an adequate understanding of these issues is accomplished, justification of changed management in the face of potentially degrading impacts on biodiversity is unlikely.

The Department of Environment and Conservation (DEC) thus undertook three (of twelve) GSS technical projects. The three interrelated projects were:

## *1. Biodiversity values on the Gnangara groundwater system*

This project aimed to address knowledge gaps, to produce maps and landscape-scale models of biodiversity elements (e.g. flora and fauna) and habitats, and to produce models of the functional processes for the ecosystems. It planned to assess data from ecological studies that have identified temporal changes in vegetation and fauna. The project also developed an ecological risk assessment to provide a suitable decision-making framework for biodiversity outcomes for the evaluation of land use and water management options.

### *2. Future land uses for Crown land*

The Department of Environment and Conservation, as a major land manager on the Gnangara groundwater system, will provide significant input into the evaluation of other land use proposals that may have an impact on the DEC-managed estate and other areas of high conservation value. A range of proposals for alternative land tenure and land use were assessed, including the possibility of returning the existing pine plantations to native woodland or groundwater recharge areas, proposals for horticultural precincts, and of converting state forest to privately owned urban and industrial development.

### *3. Fire regimes for potential water recharge*

Groundwater modelling using PRAMS has indicated that an increase in the frequency of prescribed burning of the native vegetation from 2.5 to 7.5% of area per annum will potentially provide a significant increase in recharge. This project is continuing into 2010 and is investigating the feasibility of altering current fire management practises to promote groundwater recharge, and will assess the risk of detrimental impacts on biodiversity from changed burning regimes.

## Aims of this report

This report is thus a major component of project 1 – Biodiversity values on the Gnangara groundwater system (Appendix 1). Some issues associated with projects 2 and 3 are also covered – for instance fire and biodiversity, and ecological linkages. This report aims to greatly improve our knowledge and assessment of biodiversity values and processes, and the threats to them. Further it has been designed to provide information on the environmental dependencies on the Gnangara groundwater system in the context of climate change and the predicted population growth of Perth and the implications of these for the groundwater system. It does not discuss future land use and water management options for the groundwater system. These are addressed in the sustainability strategy itself (Government of Western Australia 2009a).

Biodiversity assessments depend on high quality data of both a spatial and temporal nature. While field-based collection of this information over large areas is labour intensive and expensive, techniques such as remote sensing and the use of spatial models can provide a more cost-effective approach. This project has investigated the potential to improve the efficiency and effectiveness of mapping and ecological modelling for assessing threats to biodiversity. A combination of ground-based and remotely-sensed data has been employed to produce spatial models and maps showing the distribution of key habitats and biodiversity values, and the impacts of disturbances on these habitats and values, including altered water availability, fire regimes and plant pathogens on these elements. The outcomes include a compilation of spatial data together with spatial and conceptual models that can be used to evaluate a range of land use and water management scenarios.

The scope of this report was to determine the current state of knowledge of the distribution and status of biodiversity values, and the functional processes that maintain ecosystems and threatening processes. This included reviewing and summarising the information obtained from previous studies, together with studies undertaken during the GSS project (2007–09) (Appendix 2). The main aims included:

- determination of the occurrence and distribution of biodiversity and conservation values of the Gnangara groundwater system
- assessment of the susceptibility of biodiversity elements to climate change, changing water regimes and other threatening processes
- production of data, models and analyses that provides biodiversity information to determine impacts of a range of climate and management scenarios
- conducting risk analyses of threats, consequences and likelihoods to determine priorities
- development of landscape-scale assessments identifying habitats and biodiversity elements and their distribution and connectivity.

# The study area – Gnangara groundwater system

The Gnangara groundwater system is located on the Swan Coastal Plain, north of Perth and covers an area of approximately 2200 $km^2$. It occurs within the subregion named 'Perth Coastal Plain' (SWA2), as defined under the Interim Biogeographic Regionalisation of Australia (IBRA) (Thackway and Cresswell 1995) which is referred to as the Swan Coastal Plain in this report. The Gnangara groundwater system incorporates a number of aquifers, including the Gnangara Mound or Superficial aquifer, which is a shallow unconfined aquifer, the semi-confined Mirrabooka aquifer and the Leederville and Yarragadee aquifers that are deep and more confined aquifers that extend north and south beyond the extent of the Gnangara Mound (Allen 1981; DOW 2008a; Figure 1.1). The Gnangara Mound is about 70 m above sea level at its highest point between Muchea and Lake Pinjar and water flows away from this high point towards the Indian Ocean, the Swan River, Ellen Brook and Gingin Brook (Western Australian Planning Commission and Water and Rivers Commission 2001). The Superficial aquifer exists across the entire study area and thus defines the study area boundary for the GSS. A detailed explanation of the aquifers can be found in *Hydrogeology and groundwater resources of the Perth region, Western Australia* (Davidson 1995).

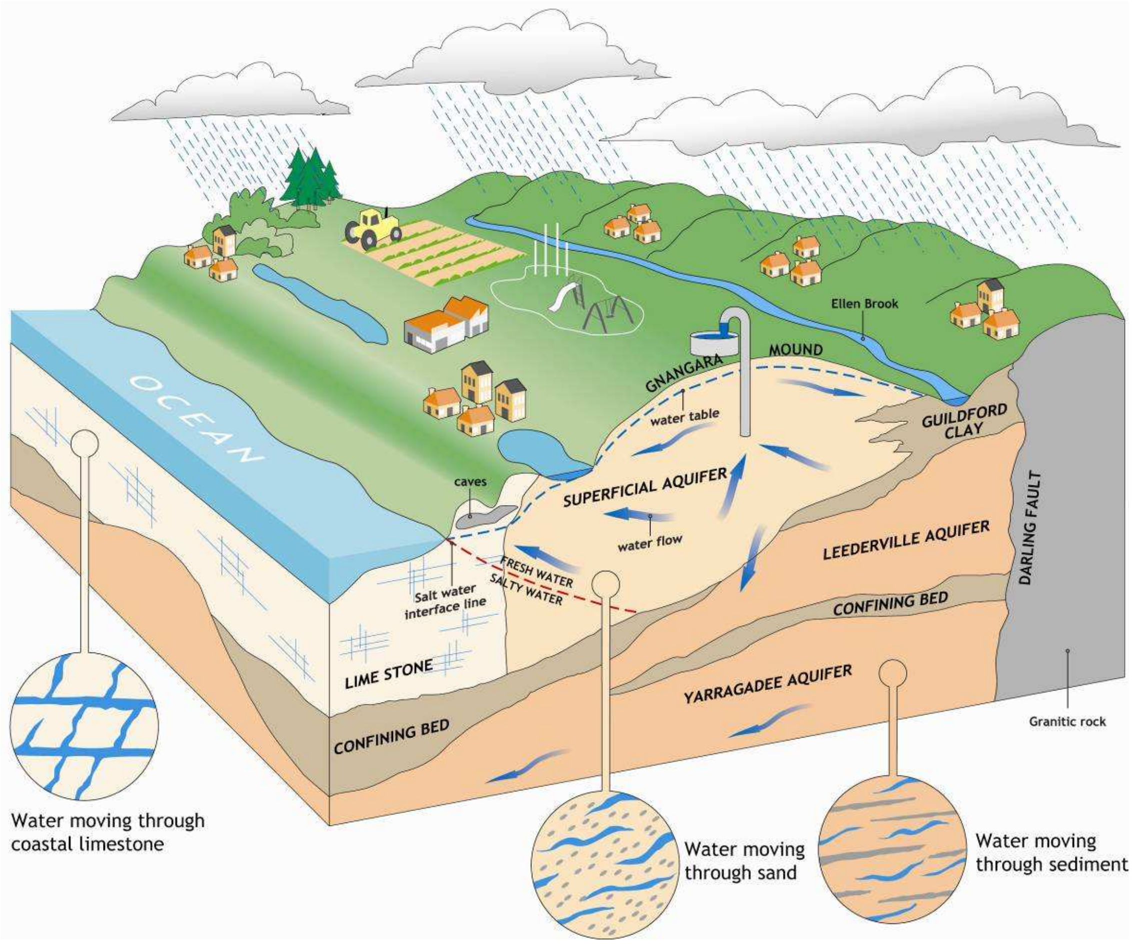



Figure 1.1: Gnangara groundwater system hydrogeological cross-section

The study area examined in the GSS encompasses the area bounded by the Moore River and Gingin Brook to the north, Ellen Brook to the east, the Indian Ocean to the west and the Swan River to the south (Government of Western Australia 2009b; Figure 1.2). Parts of the area are heavily populated and consequently biodiversity experiences high development impacts. Land clearing and fragmentation of native ecosystems for agriculture, and extensive urbanisation, has resulted in predominantly small remnants of the original vegetation remaining (Government of Western Australia 1998). The significance of remnant vegetation in maintaining biodiversity, species, and communities has been the focus of a number of studies (Government of Western Australia 2000a; b).


Figure 1.2: Location of the GSS study area

The native woodland on the Gnangara groundwater system contains the largest area of continuous bushland on the Swan Coastal Plain south of the Moore River. The Department of Environment and Conservation manages 72 447 ha of native woodlands and 23 072 ha of pine plantation on land vested in the Conservation Commission for the protection of assets under the *CALM Act 1984* (plantation timber, flora, fauna, biodiversity, recreational, water) (Figure 1.3). The DEC-managed estate occurs across 21 of the 30 subareas designated by the GSS, although the majority occurs across 8 subareas in the northern section the GSS (Figure 1.3).

The remnant vegetation within the GSS study area has state biodiversity significance, containing a number of Bush Forever sites, threatened species and ecological communities and approximately 600 wetlands. It also maintains water quality on the Gnangara Mound. The majority of the remnant bushland is designated Priority 1 Water Resource Protection – the most stringent priority classification for drinking water source protection. It is thus managed to ensure there is no degradation of the drinking water source, by preventing development of any potentially harmful activities.

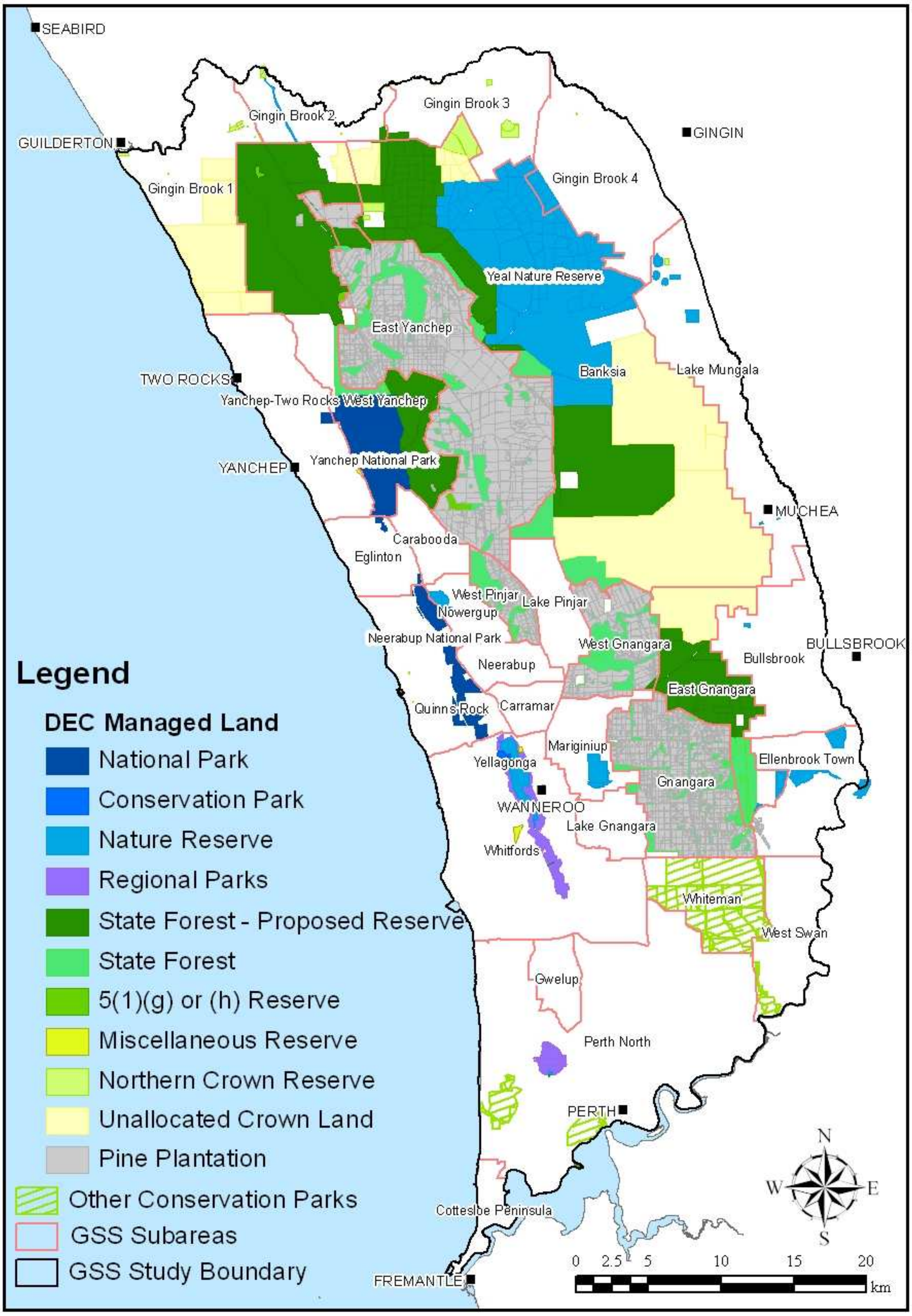



Figure 1.3: Conservation reserves, pine plantations and GSS subareas within the GSS study area

## *Perth Basin – surface geology and soils*

The two main geomorphic features of the Swan River region are the Darling Scarp and the Swan Coastal Plain which both run in a north–south direction. The Darling Scarp divides the sediments of the Swan Coastal Plain from the Darling Plateau. Fluctuations in sea level over the last two million years have resulted in significant changes to the coastline and shaped the coastal plain (Davidson 1995). The plain overlies the Perth Basin which is composed of thick sedimentary rocks and is built up from foothill, aeolian, lake, river and estuarine deposits laid down to the west of the scarp (Davidson 1995). Sand dune systems, beaches and estuaries were produced with the fluctuating sea levels. In the Perth region, the coastal plain is 20 to 34 km wide.

The Swan Coastal Plain can be divided into a sequence of broad geomorphological systems lying parallel to the coast that are characterised by a progression of aeolian sands in the west to alluvial and/or colluvial deposits in the east (McArthur and Bettenay 1960; Playford *et al.* 1976). Within the GSS study area, the major landforms include the Quindalup, Spearwood and Bassendean dunes, the Pinjarra Plain and a minor element of the Dandaragan Plateau, including the Gingin and Darling Scarp (Government of Western Australia 2000b; McArthur and Bettenay 1960; Playford *et al.* 1976). These landform units are composed of a variety of soils (Figure 1.4) and varying surface geology (McArthur and Bettenay 1960), and consist of the following elements, moving from east to west:

- Dandaragan Plateau – composed of Cretaceous sandstones and shales that have been laterised to produce lateritic and colluvial sandy soils
- A narrow band (1–3 km) of foothills which lie immediately below the granitic rises of the Darling Scarp and which comprise alluvial and colluvial material from the scarp and ancient beach sands. Soils are light grey and white quartz sands.
- Pinjarra Plain – consists of relatively recent alluvium transported from the scarp by rivers flowing to the west. Soils include clays, silts, sands and peats and are generally well-structured and relatively fertile.
- Bassendean dunes – a coastal sandplain formation (mid Pleistocene) consisting of a series of low hills (40–80 m high in parts) of heavily leached aeolian sands comprising light grey quartz sands, interspersed with extensive areas of poorly drained soils or seasonally waterlogged flats. The landform is internally drained, with groundwater-fed

wetlands occurring in lower areas. These landforms are naturally nutrient-poor, with virtually no capacity to absorb introduced nutrients and are poorly buffered.

- Spearwood dunes – the surface expression of the Tamala limestone, formed in the Pleistocene, which is overlain by yellow and siliceous sand. Lakes occur in chains in the swales of the dunes parallel to the coast. The dissolution of the limestone in this unit has resulted in some areas with a karstic environment with numerous caves. The landform is also internally drained, with extensive lakes and wetlands lying in a north–south orientation including Lake Monger, Herdsman Lake and Lake Joondalup. The sands of this system are less leached and hence more fertile than those of the Bassendean dunes.
- Quindalup dunes **–** located between the Spearwood dunes and the ocean. Mobile and fixed sand dunes which trend north-south along the coast were formed 6000–4500 years ago. The soils are calcareous aeolian deposits, also overlying Tamala limestone, and include sand, loam and clay in the swales, and acid peat in the swamps (Davidson 1995; Figure 1.4).

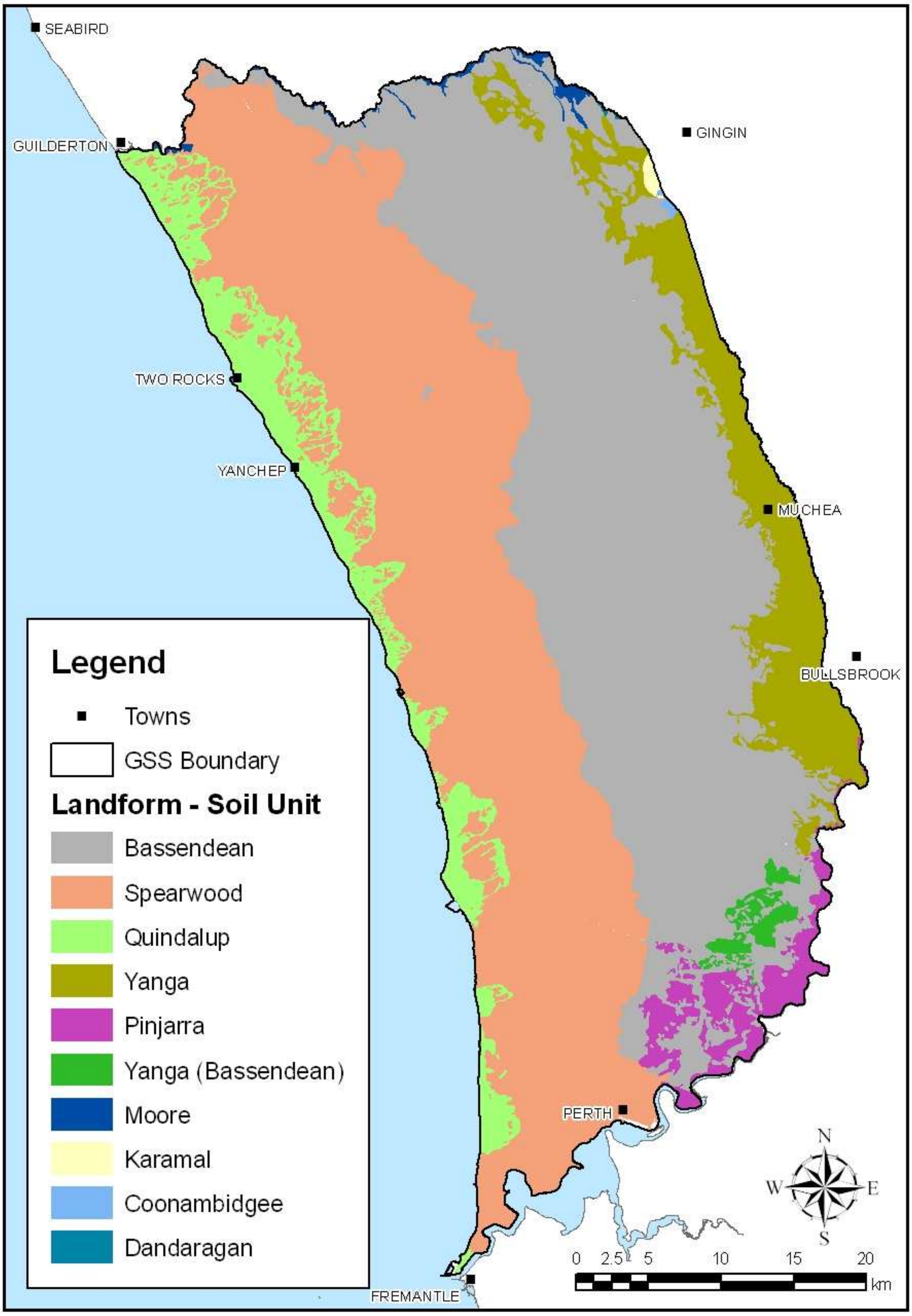



Figure 1.4: Major landform and soil types found across the Gnangara groundwater system.

## *Vegetation communities*

As mentioned previously, the GSS study area is located within the subregion called the Perth Coastal Plain (SWA2), as defined under the Interim Biogeographic Regionalisation of Australia (Thackway and Cresswell 1995), which is referred to as the Swan Coastal Plain in this report. Biogeographical regions are used to describe patterns of ecological characteristics across the landscape, including geomorphology, climate, geology, soils and vegetation and are considered a coarse-scale surrogate of detailed assessments of biodiversity. They are utilised as a basis to undertake planning to achieve a 'comprehensive, adequate and representative' conservation reserve system (May and McKenzie 2003). The Swan Coastal Plain bioregion is characterised by woodlands and heaths on colluvial and aeolian sands, alluvial river flats and coastal limestone, with a complex series of seasonal wetlands (Mitchell *et al.* 2003). The vegetation is dominated by heath and/or tuart woodlands on limestone, *Banksia* and jarrah– *Banksia* woodlands on dune systems of various ages, marri on colluvial and alluvial soils, and paperbarks in swampy areas (Mitchell *et al.* 2003). Forming approximately 16% of the Perth Coastal Plain (SWA2) subregion, the GSS study area represents a vital component of this biogeographical element. Although large areas (~ 50%of original) of vegetation have been cleared for urbanisation and agriculture, the total remnant native woodland in the GSS study area covers more than 100 000 ha (see Chapter 2).

The predominant influences on the distribution of vegetation on the Swan Coastal Plain are the underlying landforms and soils, the depth to watertable and the climatic conditions (Cresswell and Bridgewater 1985; Heddle *et al.* 1980). Heddle *et al*. (1980) defined broad vegetation complexes across the plain in relation to these landform–soil units (Churchward and McArthur 1980), and the varying climatic conditions. Twenty-one of these vegetation complexes occur in the GSS study area (see Chapter 2). Three of the vegetation complexes are located entirely within the GSS area (Bassendean Complex Central and South Transition, Karrakatta Complex North Transition and Pinjar Complex). In general, the main dune systems (Quindalup, Spearwood and Bassendean) and their associated vegetation complexes are dominated by a *Banksia* overstorey and sporadic stands of *Eucalyptus*, *Corymbia* and *Allocasuarina*, and an understorey consisting mainly of low shrubs from the Myrtaceae, Fabaceae and Proteaceae families. In addition, there are many seasonal damplands, swamps and permanent wetlands, typically fringed by *Melaleuca* spp.

and *Banksia littoralis* with variable understorey species from the Cyperaceae, Juncaceae and Myrtaceae families (Semeniuk *et al.* 1990).

When assessed in conjunction with abiotic data, the broad vegetation complexes are considered to be only moderately successful surrogates for conservation assessments (Keighery *et al.* 2007). More detailed mapping of vegetation types would improve the conservation assessment. However, this has been completed for only a proportion of remnant vegetation of the GSS study area (Mattiske Consulting Pty Ltd 2003). Mattiske Consulting Pty Ltd (2003) identified 32 different vegetation types within the mapped GSS study area, not including open waterbodies and pine plantations. Vegetation types described range from closed heath communities to *Banksia* woodlands and open eucalypt forests (see Chapter 2).

## Climate change

### *Climate overview*

The climate of the region is typical Mediterranean with dry, hot summers (mean monthly maximum greater than 30° C) and wet, mild winters (mean monthly minimum greater than 8° C). The mean annual temperature for the region is 17.1° C (Sadler 2007) and the average annual rainfall is 606 mm. The summers are dry and more than 80% of the annual average rainfall occurs between May and October in the winter 'wet season'. This seasonality is produced by the wintertime northward movement of cyclonic systems bringing westerly maritime air masses with embedded cold fronts and troughs, in contrast to predominantly dry continental air from the east over summer months. Most flooding occurs in winter although infrequent rainfall events originating from the tropics between January and April can produce widespread rainfall and flooding.

Evaporation from soil and other surfaces and water use by plants (transpiration), collectively known as evapotranspiration, dominates the water balance of the region and determines how much of the rainfall becomes streamflow or recharge to groundwater. The Bureau of Meteorology average annual pan evaporation map, based on data for 1975 to 2005, indicates that potential evaporation is more than 2200 mm a year in the Gnangara groundwater system. Evapotranspiration accounts for a large per cent of the average annual rainfall in the water balance across much of the region, leaving only a small amount to recharge to groundwater.

The most extensive observations and predictions for global climate change come from the Intergovernmental Panel on Climate Change (IPCC) assessment reports. The most recent IPCC report, published in 2007, states that 'warming of the climate system is unequivocal' (IPCC 2007). Observations are that the global mean temperature has already increased and is predicted to continue to do so. Climate change is seen as one of the most significant global challenges. It is considered to be due to a combination of human activity and cycles of natural variation (Government of Western Australia 2007a). Western Australia has experienced noticeable changes in climate, especially in the southern half of the state where there has been a general trend of decline in annual rainfall since the mid 1970s. Based on climate change predictions the expectation for the future climate of the Perth region is for continued warming with increases in mean temperature and lower rainfall (CSIRO 2006; Ryan and Hope 2006; Sadler 2007). Further declines in winter rainfall are also expected, and it is projected that reduced rainfall will result in a significant decrease in streamflow and groundwater recharge (CSIRO 2006; Ryan and Hope 2006; Sadler 2007). In addition to placing pressure on water supply security, reduced rainfall is projected to affect ecosystems, water quality and recreational and other values. Adjustment to these changes will be necessary. Climate modelling by CSIRO shows that average annual rainfalls are projected to decline in the south-west of Western Australia by as much as 20% by 2030 and 60% by 2070, compared with average recorded rainfalls to 1990. In the last 35 years, reduced rainfalls have resulted in decreases in flows to public water supply dams by more than 50% on average and decreased recharge to aquifers has also occurred due to climate variability (Government of Western Australia 2007b; Vogwill *et al.* 2008).

## Groundwater declines on the Gnangara groundwater system

The reliability of the Gnangara groundwater system for water supply depends directly on rainfall. A rainfall decline of approximately 11% has been recorded across south-west Western Australia compared to the wetter rainfall period that occurred prior to the 1970s (1914–1975). Comparison of medium-term (1975–2007) with short-term average rainfall (1997–2007 and 2001–2007) show further declines to those since the 1970s. At Wanneroo there is a clear illustration of rainfall declines during the last 50 to 100 years (Figure 1.5). These declines represent a shift to a drier climate (DOW 2008a).

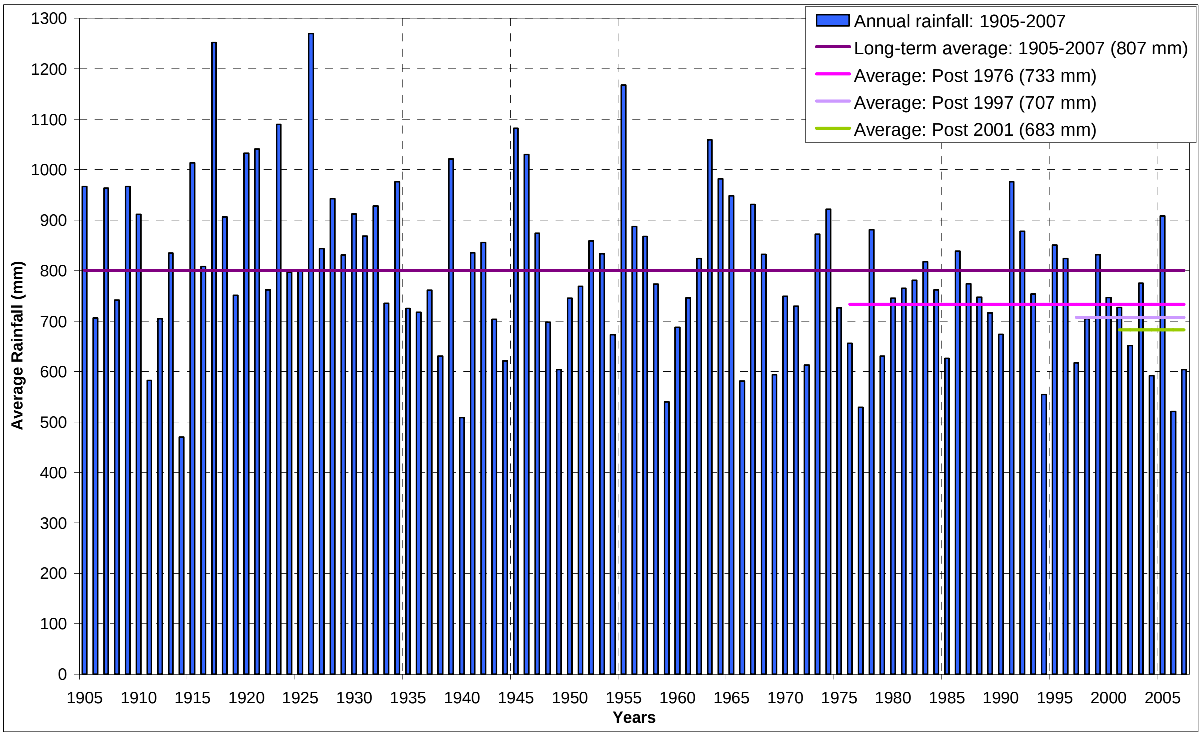



Figure 1.5: Long-term (1905–2007) annual rainfall for Wanneroo site 9105. Data source: Bureau of Meteorology (Government of Western Australia 2009b).

Groundwater declines across the Gnangara groundwater system have been recorded since the 1970s and a decrease in groundwater storage of the Superficial aquifer of approximately 45 GL per year has occurred since 1998 (Figure 1.6). Significant rainfall declines combined with increased supply for public and commercial water needs, increased evapotranspiration, and interception loss from pine plantations have contributed to these outcomes. Yesertener (2007), however, found that the major factor involved was reduced rainfall. Groundwater falls of up to 4 m were recorded over the period 1979–2004. Long-term cumulative impacts of abstraction in the Gnangara groundwater were identified as occurring on the Pinjar, Wanneroo, Gwelup and Mirrabooka borefields with declines of a maximum of 1.5 to 3.0 m within a 6 km radius of the borefields. The pine plantations (Gnangara, Pinjar and Yanchep) in the GSS have caused groundwater declines, resulting from reduced recharge, in the order of 3.5 m, particularly in areas where pines were dense. The results showed that maximum groundwater decline resulting from reduced rainfall occurred at the top of the mound. The Yeal Nature Reserve and the north-eastern part of the Lake Pinjar area experienced the most significant declines in levels, with falls of up to 4 m resulting from the reduced rainfall. Areas toward the coast and on the north-eastern and eastern parts of the mound showed declines of 1 to 2 m (Yesertener 2007). The shift to a drier climate has major implications for the management of ecosystems and biodiversity on the Gnangara groundwater system.

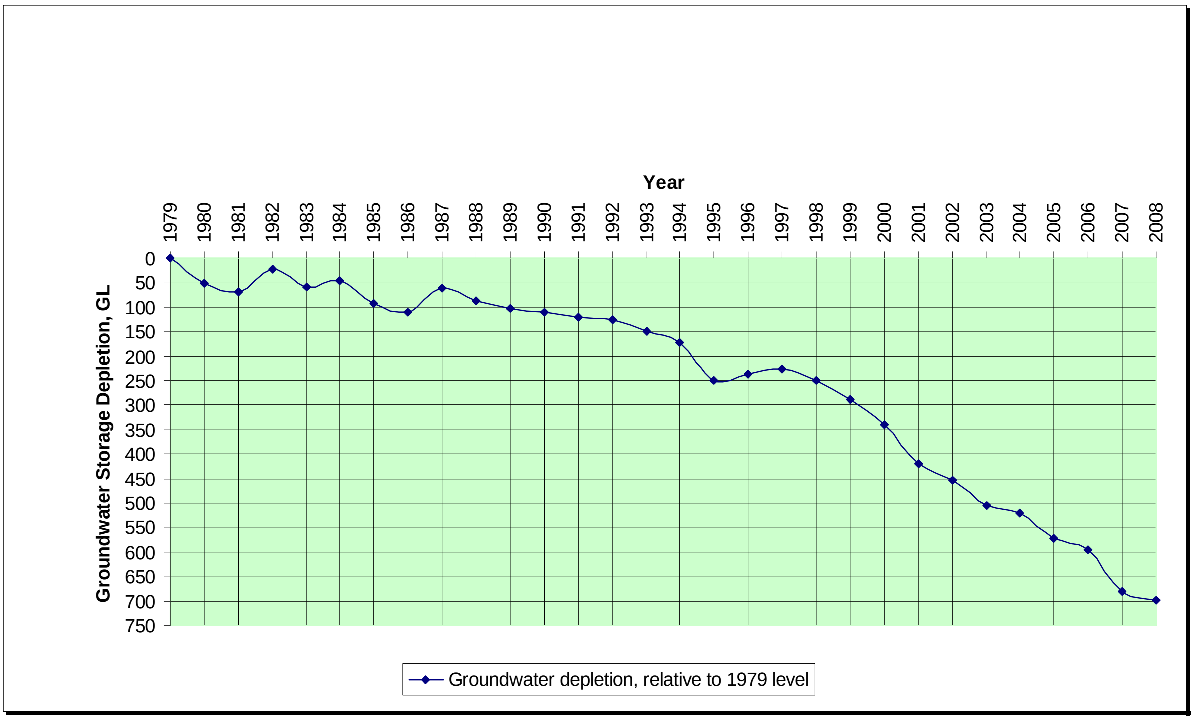



Figure 1.6: Changes in groundwater level in the Superficial aquifer expressed as a reduction in groundwater storage since 1979 (Government of Western Australia 2009b).

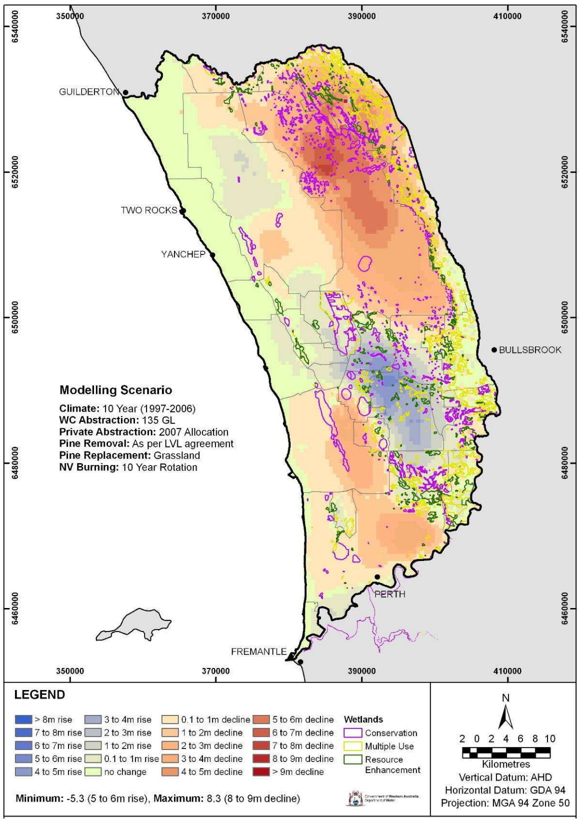
Declining rainfall and groundwater levels, as a result of climate change and abstraction, are having a severe impact on groundwater-dependent ecosystems on the Gnangara groundwater system. The degree of dependence of these ecosystems on groundwater varies. Stygofauna in caves and aquatic invertebrates in lakes need permanent water, and so are highly dependent. Riparian plant communities in fringing areas (e.g. *Melaleuca rhaphiophylla*) need seasonal inundation, and communities of phreatophytic (groundwater-dependent) terrestrial vegetation (e.g. *Banksia* woodlands) can utilise groundwater when it is available, but are able to survive dry periods. There is evidence over recent decades that lake systems are being converted to swampy flats, while unique wetlands are drying and some are becoming acidic (Appleyard *et al.* 2004). The declining health of these groundwater-dependent systems has major impacts on the vegetation, water, soil quality and on macroinvertebrates within the systems (Sommer and Horwitz 2001). Unique groundwater-dependent stygofauna in caves and mound-spring ecosystems have been severely affected (Government of Western Australia 2009b).

Long-term changes in vigour and distribution of *Banksia* spp., *Melaleuca* spp. and shrubs, particularly wet-tolerant species such as *B. littoralis* and *M. preissiana*, have been documented. In addition, there is evidence of large areas of phreatophytic vegetation being affected by the drying climate, showing a shift towards species tolerant of drier conditions (a process called terrestrialisation) (Groom *et al.* 2001). Such changes to vegetation may affect fauna by altering habitat and food availability (Government of Western Australia 2009b).

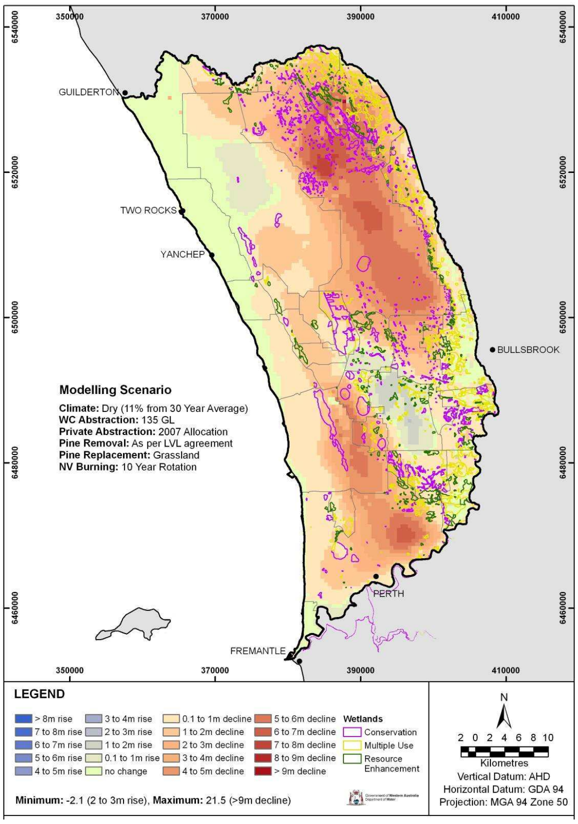
Many vertebrate fauna species and communities have been, or are likely to be, severely affected, including the critically endangered Western Swamp Tortoise, *Pseudemydura umbrina*, waterbirds, frogs and, native mammals (e.g. water rats, bush rats, bandicoots, honey possums) (Government of Western Australia 2009b).

## Modelling future impacts

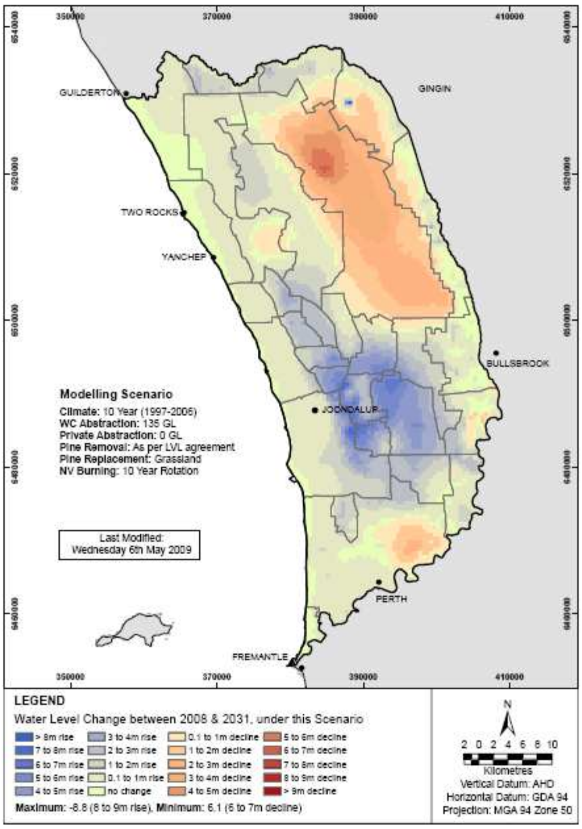
A major project of the GSS was to undertake groundwater modelling, using the Perth Regional Aquifer Modelling System (PRAMS), to predict future water balances of the Gnangara groundwater system (Government of Western Australia 2009a; Vogwill *et al.* 2008). A number of scenarios consisting of different climate predictions combined with different land and water management actions, were modelled for a period extending from 2008 until 2031. A ‘base case scenario’ was defined as having the average of the climatic parameters recorded between 1997 and 2006 and the assumptions that pine plantations would be harvested by 2028 and land maintained as grassland, that urbanisation would occur in accordance with the Metropolitan Region Scheme and *Banksia* woodlands would be burnt every 10 years on average. Groundwater abstraction was set at 135 GL per year for public water supply, licensed private water use remained at the 2007 level and there was a 3% increase per year in extraction from garden bores, reflecting increased urban growth. The PRAMS model outputs showed the probability of changes occurring and for the base case scenario it predicted that groundwater levels are likely to decline significantly at the crest of the Superficial aquifer (Figure 1.7a), which implies the loss of significant wetlands on the system. Rises in groundwater levels of approximately 1 to 5 m were predicted in areas where pines had been harvested. Climate modelling by CSIRO and the Indian Ocean Climate Initiative (IOCI) has predicted a more pronounced decline in rainfall in future years. Modelling based on a climate that is 11% drier than the 1997–2006 period predicts a likely decline in groundwater levels of up to 21 m on the crest of the Superficial aquifer (Figure 1.7b). If, as in this model, the climate does become drier over the next few years then the GSS recommendations made in 2009 will need to be revised. Another modelling scenario, albeit unlikely, was based on no private extraction (Figure 1.7c). This predicted water level rises in an increased area both to the north-west and south-east of the area in the base case scenario.




**a)**




**b)**




**c)**

Figure 1.7: Predicted changes to the watertable (2008–31) in the Superficial aquifer modelling a) base case land use and water abstraction b) rainfall 11% less than year average c) no private extraction (Government of Western Australia 2009a; Vogwill *et al.* 2008).

# Biodiversity, threats and resilience

## *Assessment of biodiversity*

The maintenance of biodiversity is fundamental to maintaining ecosystems, and is currently an environmental priority of federal and state governments in Australia. The term 'biodiversity' is widely interpreted as species richness, but should refer to all the fundamental levels of organisation including ecosystems, the species (all life forms) and the genetic composition (Noss 1990; Saunders *et al.* 1998). 'Biodiversity values' refers to the inherent values in the variety of life forms, including fauna, plants and micro-organisms, their genes and the ecosystems of which they are a part.

Assessments of biodiversity can include measures of diversity, ecosystem condition and ecosystem services (e.g. carbon storage, pollination). There are many direct (e.g. species richness) and indirect or surrogate (e.g. cover, nutrient cycling and biomass) measures of biodiversity (Noss 1990; 1991), including a range of different indicators for monitoring biodiversity (Noss 1990; Saunders *et al.* 1998). Nos (1990) defined the different attributes (composition, structure and function) at different hierarchical levels at the regional or landscape level down to the genetic level. The different levels of organisation addressed in the chapters of this report range from the regional and landscape level to the population and species level (Table 1.1). For example, at the landscape scale we have assessed biodiversity attributes such as distribution and size of remnant vegetation and of vegetation complexes and of connectivity of remnant vegetation patches (Chapters 2 and 6). At the community and ecosystem level we have assessed species richness of flora, terrestrial fauna and aquatic invertebrates (Chapters 2, 3 and 4). We have assessed structure by assessing vegetation biomass and canopy cover utilising remote sensing and ground-truthed data (Chapters 2 and 9).

Table 1.1: Hierarchical levels of attributes for indicators for monitoring biodiversity (Noss 1990) and chapters in this report where attributes are discussed.

| **Level of organisation** | **Attributes of biodiversity** | | |
|---|---|---|---|
| | Composition | Structure | Function |
| Regional / landscape | Habitat types<br>Ch 2, 3, 4, 11 | Heterogeneity and connectivity<br>Ch 4, 5, 6, 11 | Patch persistence<br>Ch 6 |
| Community / ecosystem | Species richness<br>Ch 2, 3, 4 | Vegetation biomass and canopy<br>Ch 2, 9 | Biomass and patch dynamics<br>Ch 2, 9 |
| Population / species | Abundance and cover,<br>Ch 2, 3, 4 | Dispersion and population structure | Metapopulation dynamics and fertility |
| Genetic | Allelic diversity | Effective population size | Genetic drift and inbreeding |

Protection of biodiversity involves maintaining ecosystem function and evolutionary processes. While this includes protection at all levels (e.g. landscape, community, species, genetic) there is seldom sufficient time or resources to adequately assess the conservation status of all components of biodiversity directly (Keighery *et al.* 2007; Reyers *et al.* 2000). This has led to the use of surrogates in conservation planning, including vegetation and land-system maps (Bowker 2000; Reyers *et al.* 2001), or rapid inventory techniques (Higgins and Ruokolainen 2004). These are considered ‘coarse filter approaches’ (Noss 1990) and need to be combined with more specific data or models for particular biodiversity assets (e.g. threatened species) the distribution patterns of which are not well correlated with the coarse scale mapping (Keighery *et al.* 2007). The advantage of a combined approach is that these fine filter datasets provide a greater degree of precision thereby refining the coarse filter assessment.

## *Assessment of threats and threatening processes*

A ‘threatening process’ is any process or activity that threatens to destroy or significantly modify the ecological community and/or affect the continuing evolutionary processes within any ecological community (Australian and New Zealand Environment and

Conservation Council State of the Environment Reporting Task Force 2000). The *Environment Protection and Biodiversity Conservation Act 1999* (EBC Act) provides for the identification and listing of ‘key threatening processes’. A threatening process is defined as a key threatening process if it threatens or may threaten the survival, abundance or evolutionary development of a native species or ecological community (for example, predation by the European red fox). A process can be listed as a key threatening process if:

- it could cause a native species or ecological community to become eligible for inclusion in a threatened list
- it could cause an already listed threatened species or threatened ecological community to become more endangered
- it could adversely affect two or more listed threatened species or threatened ecological communities.

The identification of a particular threat as being a key threatening process is the first step to address its impact under Commonwealth law. We identified the major processes threatening biodiversity for the Gnangara groundwater system from the literature, government documents and GSS documents (see Table 1.3).

This project has focused on a selection of these threats to identify the impacts on biodiversity. These threats include:

- alterations to water regimes (Chapters 5 and 10)
- impacts of habitat loss and fragmentation (Chapter 6)
- impacts of altered fire regimes (Chapter 7)
- effects of introduced species (Chapter 8)
- impacts of the plant pathogen *Phytophthora cinnamomi* (Chapter 9)
- impacts of climate change (Chapter 10).

## *Resilience*

Assessment of impacts should evaluate the resistance and resilience of communities, and their capacity for restoration. Disturbances disrupt community structure by changing, temporarily or permanently, factors such as the availability of substrates and resources. They also alter the physical environment, including factors such as temperature and climate. They can be classified as an exogenous disturbance such as land clearing, or an endogenous disturbance that results from biological interactions such as predation processes. There are differences in the response of communities and taxa to various forms of disturbance. For example, the regeneration of *Banksia* woodland after land clearance will differ from regeneration following fire. In addition, each ecosystem or community will respond differently to a given disturbance factor (e.g. different responses of wetlands or *Banksia* woodland to fire).

The ability of the biotic components of an ecosystem to withstand disturbances depends on the resilience and resistance of the system (Scheffer *et al.* 2001; Walker *et al.* 2002). *Resistance* can be defined as the capacity of the system to withstand change in structure and function. *Resilience* represents the capacity of a system damaged by disturbance to restore structure and function once the disturbance is removed (Carpenter *et al.* 2001; Peterson *et al.* 1998). The process involves the degree, manner and pace of restoration with some systems possessing the capacity to return to their prior state (Westman 1986). In this project we have concentrated on the resilience and resistance of communities in the Gnangara groundwater system to the disturbances listed in the previous section.

While ecological phenomena normally vary within bounded ranges, rapid, nonlinear changes can be triggered by even small differences if threshold values are exceeded. It is important to understand and anticipate nonlinear responses and ecological thresholds because the outcomes of classical models commonly described in the literature, such as ‘Clementsian’ succession, may differ significantly for these situations. Models to describe such changes have been developed and are described as state-transition models. The main features of these models are the identification of alternative ‘states’ of the systems (vegetation complexes) that remain the same or change slowly, and a set of ‘transitions’ that can occur between states. In addition, certain ‘thresholds’ of environmental factors are essential for the states. If an ecosystem which has degraded has not crossed certain thresholds, transition back to the original state is possible; but if it has crossed certain thresholds, transition back to the original state will not occur without management intervention.

In relation to key hydrological processes, state and transition models are likely to improve our understanding of the groundwater-dependent vegetation communities of the Gnangara groundwater system mound (Pettit *et al.* 2007) and, subsequently adaptive management for these communities. The *Gnangara ecohydrological study* (Sommer and Froend 2009) is currently investigating models and applications for groundwater-dependent vegetation communities. This work may provide the basis for developing similar models for climate change related effects in other communities, such as terrestrial flora, fauna and aquatic invertebrates (see Chapters 5 and 10). Such models may also assist in assessing the potential for degradation of threatened ecological communities on the Gnangara groundwater system.

Management frameworks that use state-transition models to implement adaptive management actions have been adopted by the World Conservation Union (IUCN), the World Bank, European Community, and conservation organisations around the world (See Dumanski and Pieri 2002; Hockings *et al.* 2006; Stephens *et al.* 2002). This type of management framework requires a description of the states of the current system across the biodiversity management scales of landscape, ecosystem and species.

## Approach adopted for the GSS biodiversity project

The GSS taskforce and projects commenced in July 2007. The majority of the DEC team (Appendix 3) was brought together between July and December 2007 when planning for the biodiversity project was undertaken, biodiversity data was sourced and collated and preliminary reviews of the status of biodiversity and major threatening processes initiated. Field surveys commenced during this period to assess the current status and distribution of terrestrial vertebrates, their habitats and impacts of identified disturbances and threats. Work on threatening processes including collation of *P. cinnamomi* distribution data, field mapping and remote sensing data for vegetation assessments was also initiated. During 2008 projects undertaken included development of fauna habitat models, ecological function models, continuation of field studies and reviews. This work continued in 2009 together with projects involving development of risk analyses, spatial analyses of biodiversity assets and priorities, and collation and writing of final reports.

A number of projects were undertaken by the DEC–GSS team directly (Appendix 3). In addition, specialist consultants were also chosen to undertake specific projects in order to expand the knowledge and understanding of biodiversity values in the GSS study area. There was a need to restrict these assessments to selected groups or taxons (e.g. terrestrial birds, frogs) due to the limitations of the project timelines that were recognised early in the planning of the GSS.

A scientific advisory committee was established in 2007 to provide independent advice and assessment of the three DEC projects being undertaken for the GSS. This included the appropriateness of the studies and projects in terms of their scope, quality and relevance to management of the Gnangara groundwater system, and the protection and conservation of biodiversity. Further, advice was sought from the advisory committee on the framework, planning, and methodology (experimental designs and analyses) of the DEC–GSS projects. The committee was also consulted for advice on the management principles and practices in the projects, progress of the project timelines, and the standard of reports and major documents produced by the projects. A list of members of the scientific advisory committee is provided in Appendix 4. Individuals with particular expertise (e.g. DEC, museums, universities, consultants) were also contacted to provide guidance for projects about unpublished work, major gaps in the data and the current understanding of biodiversity values and threats in the GSS study area (see Appendix 5).

The preliminary reviews and information obtained indicated that although there is substantial knowledge of specific flora and fauna taxa and the threatened ecological communities in the GSS study area, the current overall understanding of biodiversity values, particularly at the landscape scale, is inadequate (Government of Western Australia 2009b). A summary of the known status of biodiversity values at the beginning of this project is provided in Table 1.2. The information is strongly focused on threatened taxa and communities.

There are several chapters in this report that focus on the biodiversity values of the Gnangara groundwater system. These are the floristic biodiversity (Chapter 2), vegetation communities and their condition and extent (Chapter 2), terrestrial fauna (Chapter 3) and wetlands (Chapters 4 and 5). Each chapter begins with a review of the literature and is followed by a summary of research reports and field studies undertaken during the DEC–GSS project (2007–09).

Table 1.2: Summary status of biodiversity values in the GSS study area.

| **Biodiversity values (taxa/group)** | **Status** | **Selected references** |
|---|---|---|
| **Bioregion** | | |
| Swan Coastal Plain (SWA2) | | 1 |
| **Ecosystems** | | |
| Area cleared | 113 002 ha | 2,3 |
| Remaining area of remnant vegetation | 101 212 ha | 2,3 |
| Wetlands pre-European | About 9 000 ha | 3 |
| Proportion of wetlands modified or destroyed | About 80% | 4 |
| Areas affected by *Phytophthora cinnamomi* | *Banksia* woodlands | 5,6 |
| **Ecological communities** | | |
| Threatened | 10 | 1, 7 |
| Critically endangered | 4 | 1,7 |
| **Species** | | |
| Flora | DRF 10,<br>PF 37 | 1, 7, 8 |
| Mammals | V 1, P 3,<br>LEX 11 | 1, 3, 8 |
| Reptiles and amphibians | CR 1, P 3 | 1, 3, 8 |
| Birds | EX 9, CR 1,<br>P 2 | 1, 8 |
| Fish | P 1 | 1, 8 |
| Invertebrates | CR 1, P 4 | 1, 8 |

*DRF = declared rare flora. PF = priority flora. EX = extinct. CR = critically endangered.*
*V = vulnerable. P = priority, LEX = locally extinct. Pc = Phytophthora cinnamomi*
1 May & McKenzie (2003). 2 Beard (1995). 3 GSS preliminary data. 4 Balla (1994). 5 WWF (2004). 6 Project Dieback (2008), 7 Atkins (2008). 8 Government of Western Australia (2006).

The preliminary review and consultation with experts (Appendix 5) assisted in the identification of a number of the known or likely threatening processes (Table 1.3; (Government of Western Australia 2009b). The work indicated that although there is general knowledge of the threatening processes that occur in the GSS study area, the current overall understanding of the impacts, and spatial extent of impacts is inadequate (Government of Western Australia 2009b). The current understanding of the relative impacts of threats was also identified as being insufficient, thus restricting the ability to prioritise management actions and resources in order to have maximal outcomes. This report has focused on a select number of the major threatening processes and aims to identify the impacts and risks to biodiversity.

As mentioned previously, these threats include:

- alterations to water regimes (Chapters 5 and 10)
- fragmentation and habitat loss (Chapter 6)
- altered fire regimes (Chapter 7)

- introduced species (Chapter 8)
- the plant pathogen *P. cinnamomi* (Chapters 8 and 9).

The overall predicted impacts of climate change on species and ecosystems on the Gnangara groundwater system are addressed in Chapter 10.

Table 1.3: Biodiversity-threatening processes identified in the GSS study area.

| **Threat** | **Consequences** | **Selected References** |
|---|---|---|
| Clearing and fragmentation of native vegetation | Extinctions; declines of taxa, habitat; degradation of ecosystems; altered hydrology, microclimate, nutrient cycling; loss of ecosystem resilience | 1,2 |
| Climate change-<br>low rainfall, groundwater decline,<br>higher temperatures | Altered scale of existing threats; synergy with other threats, species declines and extinctions | 3,4,5 |
| Changes in water abstraction | Declines of taxa, habitat; degradation of ecosystems; altered hydrology, microclimate, nutrient cycling; | 5, 16, 17 |
| *Phytophthora cinnamomi* dieback | Declines and extinctions of flora and fauna species; degradation of fauna habitat, ecosystem function, soils and vegetation cover and condition; decline in primary productivity | 8,9 |
| Introduced predators – foxes, pigs, cats | Declines and extinctions of native animals; habitat decline; spread of weeds and disease | 6,7,14,15 |
| Fire regimes – frequency, extent, intensity, season | Decline in species, indirect impacts – increased weeds, feral animals, *P. cinnamomi* spread and grazing effects | 14,15 |
| Introduced weeds | Competition for resources with native species; fire regime changes; modification of fauna habitats; productivity and ecosystem function changes | 12,13 |

1 Hobbs & Saunders (1993). 2 Saunders *et al*. (1991). 3 Hughes (2003). 4 IOCI (2005). 5 IPCC (2007). 6 Abbott (2006). 7 Kinnear *et al.* (1988; 2002). 8 Shearer *et al*. (2007). 9 Garkaklis *et al*. (2004). 12 Keighery & Longman (2004). 13 Swan Region Weed Strategy, Keighery and Bettink, (2008). 14 Kitchener *et al*. (1978). 15 How & Dell (2000). 16. Sommer & Horwitz (2001). 17. Groom *et al.* (2001))

A significant challenge for conserving and managing biodiversity is determining the priorities and risks involved. We have developed approaches to determining biodiversity conservation priorities and ecological risks for biodiversity values on Gnangara groundwater systems. Where possible we have identified areas of high biodiversity, conservation priority and risks (Chapter 11) to direct management actions. It is important to determine which risks are most significant and assess which factors determine the vulnerability of ecosystems to current and future risks. Biodiversity components exhibit different vulnerabilities to threats with some being resilient to some extent. There is a need to identify which ecosystems, communities and taxa are resilient and which are susceptible to change driven by climate. Any current management responses should be assessed to determine if they are achieving their aims, and whether the indicators that are being monitored are appropriate. These threatening processes, impacts, current management responses and proposed management options are addressed in a number of chapters and discussed further in Chapters 11 and 12.

## Fauna and flora surveys

Although the GSS study area is known historically for the richness of its terrestrial vertebrate taxa, there have been few systematic surveys since 1977–78 and there is only a broad understanding of habitats and threatening processes. The DEC–GSS team thus instigated a number of surveys of the fauna, including reptiles, frogs, mammals, birds, and aquatic invertebrates. Surveys were undertaken by DEC–GSS staff or by consultants. The surveys were designed to satisfy five main objectives. These were to:

- determine the current status, occurrence and distribution of fauna persisting in the region
- investigate the patterns of assemblage relationships
- assess regional relationships
- determine if any changes have occurred with time
- examine patterns of distribution and abundance of taxa and communities in relation to landscape context, habitat factors and disturbance factors.

The major findings are documented in this report while the full results and analyses are provided in the technical report (see Appendix 2 for a list of reports arising from this project).

The Swan Coastal Plain is well known for its floristic diversity, and a number of projects were designed to examine patterns in floristic diversity in the GSS study area. Surveys were undertaken by DEC–GSS staff or by consultants. The specific aims were to:

- assess the floristic variability, particularly in the northern section of the GSS study area
- examine the floristic diversity at fauna trapping sites, and explore diversity relationships between flora and fauna
- compare floristic diversity between areas infested with *Phytophthora cinnamomi* and uninfected areas
- examine the short-term impact of a prescribed burn.

The major findings of these surveys are documented in this report (in Chapters 2 and 9) while the full results and analyses are reported in the technical reports (see Appendix 2).

## Alignment of GSS and DEC Nature Conservation Division projects and planning

The Nature Conservation Division (NCD) is a major division of DEC which has a major responsibility to conserve Western Australia's biodiversity. The three GSS projects being undertaken by DEC (Biodiversity values on the Gnangara groundwater system, land use on publicly managed lands and fire regimes for potential water recharge and its impacts on biodiversity) thus provide an impetus for the alignment of the NCD planning. The GSS study area comprises a proportion of the IBRA Swan Coastal Plain (SWA2) subregion and the DEC Swan Region. The policy and planning context includes the:

- Draft *100 year Biodiversity Conservation Strategy for Western Australia* (2006), which identifies key strategic directions (targets and actions) for Western Australia
- *Swan Region nature conservation service plan* (2006–09, 2009–14 (DEC 2006; 2009), which is one of nine plans that will provide the basis for the triennial delivery of the Nature Conservation Service at a regional scale for the Nature Conservation Division, Science Division and Regional Services Division
- Nature Conservation Division adaptive management projects, which include large scale active adaptive management projects that area currently in development, where biodiversity outcomes are the primary objective
- Nature Conservation Division long-term, large-scale, strategic approach to assessment and monitoring of native vegetation condition.

The nature conservation regional service plans describe three-year regional scale nature conservation outcome targets, priorities, and actions that are intended to contribute towards the Nature Conservation Division's aspirational outcomes. The plans aim to provide, when fully implemented, a major movement towards outcome-based management, and recognition of the place of active adaptive management, where research is integrated with and helps inform operational aspects of conservation management. They also introduce processes for monitoring and evaluation of biodiversity status and condition to determine effectiveness of activities in achieving desirable outcomes.

The drafts of the plans were written at expert-based workshops and informed by data and information from *A biodiversity audit of Western Australia's biogeographical subregions in 2002* (May and McKenzie 2003), then circulated to relevant staff for comment and further input for the final plan. The regional plans were developed by identifying major threats to biodiversity, outcome targets and candidate actions.

### Data protocols and standards

A major consideration for the DEC–GSS projects was to maintain information integrity for the process and within the wider DEC operations. Consultants were employed to develop a series of data protocols and information management procedures with specifications, guidelines and standards to guide the collection, manipulation, analysis and management of data developed as part of the GSS (Eco Logical Pty Ltd 2008). This included procedures to adapt standard flora and fauna field survey methodologies, proformas and supporting information and define and develop a series of tabular and geographical information system (GIS) databases for data entry, analysis and reporting. The data guidelines contain information on specifications, guidelines and standards for data, including spatial data standards, spatial referencing standards, data transfer standards and metadata (Eco Logical Pty Ltd 2008). In addition, guidelines for consideration for a data supply process and map production standards were developed. These protocols have recently been completed and await further assessment.

## Structure of the biodiversity report

This report consists of twelve chapters beginning with this chapter which provides an introduction to the *Gnangara Sustainability Strategy* and the Gnangara groundwater system, the background to the biodiversity project, and the approaches adopted for the work undertaken. The first section of the report – Chapters 2 to 4 – deals with the biodiversity values and ecosystem functions of the Gnangara groundwater system. These chapters identify knowledge gaps and limitations to the studies undertaken under the GSS, and make recommendations for further work. Each chapter of this section begins with a review of the literature and is followed by a summary of research reports and field studies undertaken during the project (2007–09). Chapter 2 presents information on the vegetation and floristic values of the study area, including information on vegetation condition in the Gnangara groundwater system. Chapter 3 highlights the diversity and values of the terrestrial fauna throughout the Gnangara groundwater system. The biodiversity values of wetlands are assessed and discussed in Chapter 4.

The second section of the report focuses on the major threats to biodiversity that have been identified on the Gnangara groundwater system. These are:

- alterations to water regimes (Chapters 5 and 10)
- impacts of habitat loss and fragmentation (Chapter 6)
- impacts of altered fire regimes (Chapter 7)
- effects of introduced species (Chapter 8)
- impacts of the plant pathogen *Phytophthora cinnamomi* (Chapter 9)

Chapter 10 reviews the literature on the impacts of climate change on species and ecosystems, with a focus on predicted impacts on wetland systems, and biodiversity on the Gnangara groundwater system. Chapter 11 reviews approaches to determining biodiversity conservation priorities and ecological risk assessment and develops an approach for prioritising threats to biodiversity values on the Gnangara groundwater system to identify and map where possible areas of high biodiversity conservation priority. The final chapter, Chapter 12 provides a summary and synthesis of the chapters, and makes recommendations for management and future work and research.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


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# Chapter Two: Floristic Biodiversity and Vegetation Condition

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Our environment, our future

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Department of Agriculture and Food WA
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CSIRO
Forest Products Commission Western Australia

# 2. FLORISTIC BIODIVERSITY AND VEGETATION CONDITION

## Key points

- The GSS study area is centrally located within south-west Western Australia, one of the world's 32 biodiversity hotspots, renowned for its floristic diversity, endemism and high rates of habitat loss.
- Species richness in the GSS is high – the total number of plant taxa recorded is 1901, including 1337 native taxa and 564 introduced taxa. The GSS study area contains some 17% of the floristic diversity in the South Western Australian Floristic Region (SWAFR), and nearly 50% of the native vascular plants recorded from the Swan Coastal Plain.
- Plant species richness varies among vegetation types, with *Banksia* woodlands containing most plant species, typically within the lower strata of vegetation (< 40 cm).
- Within the GSS study area, 10 species are gazetted 'declared rare flora' and an additional 37 taxa are considered 'priority flora'. Of these, 6 species are unique to the GSS study area.
- Ten threatened ecological communities have been identified, including the critically endangered aquatic root mat community of caves of Swan Coastal Plain (Yanchep caves), communities of tumulus springs (Organic mound springs, Swan Coastal Plain) and Perth to Gingin ironstone association (Northern ironstone), which are unique to the GSS study area.
- A number of vegetation units have been described, including 21 vegetation complexes, at least 21 floristic community types, and 32 site-vegetation types. Of the vegetation complexes, three are unique to the GSS study area, and a further two had greater than 60% of their pre-European extent within the GSS study area.
- In conservation planning, vegetation type, extent and condition are widely used as surrogate measures for biodiversity. Remote sensing is a feasible option for monitoring vegetation condition at specific sites. In the GSS study area, remote sensing detected changes in vegetation condition due to the combination of two threatening processes – *P. cinnamomi* and recurrent fire frequency.

- The condition of remnant vegetation patches located in the pine forest matrix varies considerably, with wetland habitat types being more degraded than upland areas. Vegetation condition provides an index of the health of the remnant habitat, and can be incorporated as a value to rank bushland patches and, consequently, may provide a method of identifying priority habitat.

# Introduction

The GSS study area is centrally located within south-west Western Australia and is internationally renowned for its high floristic diversity and endemism (Beard *et al.* 2000; Hopper and Gioia 2004). South-west Western Australia has been identified as one of the world's 32 biodiversity hotspots, defined as regions containing outstanding biodiversity values that also have undergone or are experiencing exceptional loss of habitat (Mittermeier *et al.* 2004). Known as the South West Botanical Province (Beard *et al.* 2000), or the South Western Australian Floristic Region (Hopper and Gioia in prep.; Hopper and Gioia 2004), the south-west of Western Australia has > 7000 vascular plant taxa, of which approximately 50% are endemic (Coates and Atkins 2001; Hopper and Gioia 2004).

Assessing the conservation status of all components in diverse areas such as south-west Western Australia is often difficult (Keighery *et al.* 2007; Reyers *et al.* 2000). For conservation planning purposes, regions are often separated into different vegetation groups and land-system maps (Bowker 2000; Reyers *et al.* 2001), where units represent surrogates of biodiversity typical of that area. In south-west Western Australia there are a number of vegetation classification systems, most of which represent a combination of soil type, floristics and climate (e.g. vegetation complexes; Heddle *et al.* 1980). In addition to species richness and vegetation units, the condition (or quality) of vegetation units is now widely used as a surrogate for biodiversity (Saunders *et al.* 1998), and may be assessed in a number of ways, including the use of remote sensing techniques.

Western Australia is covered by Beard's vegetation mapping (Beard 1990). The GSS study area is located in the IBRA Perth Coastal Plain subregion (SWA2), referred to in this report as the Swan Coastal Plain. The Swan Coastal Plain is a narrow strip of land that runs parallel to the coast and is bordered by the Darling Scarp and Gingin Scarp to the east. The area surrounding Perth is highly populated, and clearing of native habitat for agriculture and urbanisation has been extensive. However, in the centre of the GSS study area there remains the largest continuous area of remnant vegetation on the Swan Coastal Plain, being south of the Moore River and covering over 60 000 ha (Sonneman and Brown 2008). Remnant vegetation has significant state biodiversity values, as it contains a number of Bush Forever sites, threatened species and ecological communities, and numerous wetlands (Government of Western Australia 2000a).

The aims of this chapter are:

- to collate and review available information on terrestrial vascular plants and vegetation units
- to identify rare and priority flora and threatened ecological communities on the Gnangara groundwater system and the Swan Coastal Plain
- to review techniques, including vegetation condition, used for assessing conservation value of remnant vegetation.

These aims were addressed by:

- reviewing databases and literature on the floristic diversity and vegetation units within the GSS study area and the Swan Coastal Plain
- identifying the occurrences and distribution of rare and priority flora and threatened ecological communities within the GSS study area and the Swan Coastal Plain
- reviewing techniques used for classifying vegetation and applicable spatial vegetation mapping in the GSS study area
- reviewing the literature associated with assessing vegetation condition to provide background information for the DEC-GSS projects that assessed vegetation condition.

In addition, a number of DEC-GSS projects were undertaken to address identified knowledge gaps for the GSS study area. These projects are outlined below.

1) **Patterns of floristic diversity in the GSS study area**

   Numerous floristic studies have been conducted on the Swan Coastal Plain, and as a result, the floristic richness of this area has been well described. However, the patterns of floristic diversity are still not clearly understood. This project aimed to further understand how floristic diversity varied across different land form types, vegetation types and time-since-fire in the GSS study area.

2) **Monitoring vegetation condition in the *Banksia* woodlands of the Gnangara Mound: the role of remote sensing tools**

   The aim of this project was to gain a greater understanding of how remote sensing tools can be used in the monitoring of *Banksia* woodland vegetation across the Gnangara Mound. A number of threatening processes impact on these woodlands, such as *Phytophthora cinnamomi*, fragmentation, weeds, decreasing rainfall due to climate change, groundwater extraction and changed fire regimes. There is an urgent need to develop methods to adequately monitor native vegetation condition so this information can be used to underpin the management actions.

3) **Remnant vegetation in the pine plantation and Lake Pinjar**

   This project conducted rapid assessments of vegetation condition in the remnant vegetation patches within the pine plantations in the GSS study area and vegetation in the Lake Pinjar area. This assessment was undertaken to aid in the delineation of ecological linkages, to identify patches of vegetation with greater ecological value, and to ensure that patches of vegetation were included in the placement of ecological linkages.

The major findings of these projects have been summarised in this document. The full technical reports are provided as a list in Appendix 2.

# Species richness and endemicity

## The floristic diversity of south-west Western Australia

The flora of Western Australia is highly diverse, with nearly 12 000 plant taxa (taxa includes subspecies; Coates and Atkins 2001) or nearly 8500 described plant species (Beard *et al.* 2000); representing 4 to 5 per cent of the estimated world vascular flora. A large proportion of Western Australia's flora (~ 8000 plant taxa; Coates and Atkins 2001) are located in the south-west of the state, an area renowned for high floristic diversity and endemism (Mittermeier *et al.* 2004; Myers *et al.* 2000). Endemism refers to a species' or a community's distribution being ecologically unique to an area. In this report we use the following categories of endemism:

- GSS unique – a species or community which is only found within the GSS study area
- local endemic – refers to a species or community which is only found within the Swan Coastal Plain (IBRA Perth Coastal Plain SWA2 subregion)
- regional endemic – refers to a species or community which is only found within south-west Western Australia.

The south-west Western Australia region was originally described as the South-West Botanical Province (Beard 1980; Beard *et al.* 2000; Myers *et al.* 2000) and has an area of 309 840 $km^2$. The area is also known as the South Western Australian Floristic Region (SWAFR; Hopper and Gioia in prep.; Hopper and Gioia 2004). The SWAFR is similar in extent to the Southwest Botanical Province, with an adjusted northern boundary to the south of Shark Bay in accordance with Gibson *et al.* (2000) and covers 302 627 $km^2$ (Hopper and Gioia in prep.). The area is described as relatively wet to semi-arid refuge that is very species rich with high levels of endemism. Beard *et al.* (2000) reported 5710 native species in the region, of which 53% are regionally endemic to the SWAFR (79% are Western Australian state endemics). Hopper and Gioia (2004) report a total of 7380 native plant taxa (approximately 6790 species) of which 49% are regionally endemic.

The high levels of species richness and endemism in south-west Western Australia are attributed to a number of factors, the dominant being the extreme poverty of the region's soils that promotes sclerophylly and habitat specialisation, thus resulting in many endemics (Beard *et al.* 2000). In addition, species have evolved over a long period of time with increasing aridity and have been predominantly isolated from the rest of Australia by desert (Coates and Atkins 2001). Cowling *et al.* (1996) attributes two main causes for the high speciation rate and low extinction rate of south-west Western Australia. These are high fire frequencies (resulting from low soil fertility and associated high flammability), and mild quaternary climates resulting from a strong maritime influence. The region has particularly high diversity among the angiosperms, especially woody families such as the Myrtaceae and Proteaceae (Hopper and Gioia 2004). Indeed, south-west Western Australia has been a major centre of diversity for both of these plant families since at least the Eocene (McLoughlin and Hill 1996), a period 55–33 million years ago.

## Floristic diversity of the GSS study area

There have been a number of major studies on the flora of the Swan Coastal Plain (Gibson *et al.* 1994; Keighery 1999; Marchant *et al.* 1987). Within the Perth metropolitan region alone, > 1200 native taxa have been identified (DEP 1996). Species richness in the Swan Coastal Plain is high and floristic quadrat (10 x 10 m) estimates range from 9 to 66 taxa (Government of Western Australia 2000b). The highest diversity has been recorded in woodlands of the Bassendean dunes and on the eastern side of the plain (Government of Western Australia 2000b). Woody native species from the Myrtaceae and Proteaceae families tend to dominate the flora of the GSS and the Swan Coastal Plain (Barrett and Pin Tay 2005). Prominent overstorey species in the GSS include tuart (*Eucalyptus gomphocephala*), jarrah (*E. marginata*), marri (*Corymbia calophylla*), the coastal blackbutt (*E. todtiana*), *Melaleuca* spp., as well as several *Banksia* species, including the slender banksia (*B. attenuata*), firewood banksia (*B. menziesii*), holly-leaved banksia (*B. ilicifolia*) and the swamp banksia (*B. littoralis*). Indeed, the *Banksia* woodlands typical of the GSS study area are floristically rich, taxonomically diverse, and exhibit a high degree of variability in the understorey (Dodd and Griffin 1989). In addition, there are a number of wetland associated plant species and vegetation communities, and these are discussed in greater detail in the wetlands chapter (Chapter 4).

To determine current species richness in the GSS study area, we used herbarium records to derive species location datasets to determine total species richness and dominant plant taxa. Within the GSS study area, the total number of plant taxa (species and infraspecies) that have been recorded is 1901, including 1337 native taxa and 564 introduced taxa. A total of 157 plant families are recorded from the GSS study area, comprising 649 genera, of which 316 genera include alien taxa. The ten dominant families include ~ 53% of the total native vascular taxa recorded in the GSS study area and include Myrtaceae, Proteaceae, Cyperaceae, Orchidaceae, Asteraceae, Papilionaceae, Stylidiaceae, Poaceae, Haemodoraceae and Mimosaceae. The twelve dominant genera within the GSS study area contribute ~ 22% of the total native vascular taxa recorded, and include *Melaleuca, Verticordia*, *Banksia* (including *Dryandra* taxa), *Hakea, Schoenus, Caladenia, Stylidium, Conostylis, Acacia*, *Leucopogon*, *Drosera* and *Hibbertia*. There are 17 *Banksia* species known from the GSS study area, including nine species formerly considered in the genus *Dryandra* (Mast and Thiele 2007).

### *Relative richness and representativeness of floristic diversity in the GSS study area*

To determine the relative importance of floristic diversity on the GSS study area, we compared species richness and representativeness of dominant plant taxa among the GSS study area, Swan Coastal Plain and the South Western Australian Floristic Region. Species richness and representation estimates for the Swan Coastal Plain were determined using herbarium records obtained in 2008 (for the Perth Coastal Plain SWA2 IBRA subregion), while species richness estimates for the SWAFR were obtained from Hopper and Gioia (2004), using herbarium records from 2004. The dominant plant taxa used are those identified for the SWAFR region (Hopper and Gioia 2004). In addition to species richness and representativeness, we used area estimates for each region to calculate a relative richness measure, expressed as the number of plant taxa per km$^2$.

In species richness terms, the GSS study area is floristically diverse in comparison to both the Swan Coastal Plain and the SWAFR (Table 2.1). The GSS study area encompasses less than one-sixth of the area of the Swan Coast Plain but contains ~ 50% of the native vascular plants (Table 2.1). In comparison to the SWAFR, the GSS study area contains ~ 17% of the plant species in an area 140 times smaller (Table 2.1). The relative richness measure of the GSS study area indicates that there is three times the number of species in each km$^2$ compared to the Swan Coastal Plain, and 25 times compared to the SWAFR. This may be attributed to the relatively larger area of remnant vegetation remaining within the GSS (~ 47%; see Chapter 6), as opposed to the Swan Coast Plain (~ 37%) and the SWAFRA (~ 30%).

Table 2.1: Native vascular plant richness in the GSS study area, Swan Coastal Plain (SWA2) subregion and the South West Australian Floristic Region

| | GSS study area | Swan Coastal Plain[a] | SWAFR[b] |
|---|---|---|---|
| Species | 1 196[a] | 2 372[a] | ~ 6 790[b] |
| Plant taxa | 1 337[a] | 2 740[a] | 7 380[b] |
| Area (km$^2$) | 2 149 | 13 339 | 302 627 |
| Richness index (taxa/km$^2$) | 0.622 | 0.205 | 0.024 |

[a] Data derived from Western Australian Herbarium (2008)

[b] Data obtained from Hopper and Gioia (2004)

The number of taxa from the ten dominant vascular plant families and genera within the SWAFR is compared to the number and proportion these contribute to the GSS study area and the Swan Coastal Plain in Table 2.2. The ten dominant families contribute 64% of the taxa found in the SWAFR region while the dominant genera contribute 29%. These 10 families account for more than half of the total native vascular flora of the GSS study area (53%) and Swan Coastal Plain (55%). The ten dominant genera account for 16% and 20% of the GSS and Swan Coastal Plain taxa, respectively.

The percentage of plant taxa of the Myrtaceae and Mimosaceae family found in the GSS study area is similar to that in the Swan Coastal Plain, but less than the number found in the South Western Australian Floristic Region (Table 2.2). Within the Myrtaceae family, the genera *Eucalyptus* and *Melaleuca* are under represented in the GSS and the Swan Coast Plain compared to the SWAFR, while the number of taxa of the *Verticordia* genus was similar in the three areas. The representation of taxa in the Proteaceae family was lower within the GSS than both the Swan Coast Plain and the SWAFR, and this is made evident by the lower number of *Grevillea* and *Dryandra* taxa. In contrast, the representation of taxa in the Asteraceae and Cyperaceae families is higher in the GSS study area than in the Swan Coast Plain and the SWAFR.

Table 2.2: The ten dominant families and genera of native vascular plant taxa recorded in the SWAFR bioregion (Hopper and Gioia 2004), with a comparison of their contribution to native flora of the GSS and Swan Coastal Plain(Western Australian Herbarium 2008). Percentages relate to the contribution of each taxa to the total plant taxa of each region.

| Plant Family | Genera | **GSS study area**[a] | | **Swan Coastal Plain**[a] | | **SWAFR**[b] | |
|---|---|---|---|---|---|---|---|
| | | **Number** | **%** | **Number** | **%** | **Number** | **%** |
| **Myrtaceae** | | **120** | **9.0** | **274** | **10.0** | **1283** | **17.4** |
| | *Eucalyptus* | 17 | 1.3 | 42 | 1.5 | 362 | 4.9 |
| | *Melaleuca* | 18 | 1.3 | 42 | 1.5 | 185 | 2.5 |
| | *Verticordia* | 19 | 1.4 | 36 | 1.3 | 138 | 1.9 |
| **Proteaceae** | | **91** | **6.8** | **277** | **10.1** | **859** | **11.6** |
| | *Grevillea* | 15 | 1.1 | 61 | 2.2 | 229 | 3.1 |
| | *Hakea* | 17 | 1.3 | 43 | 1.6 | 105 | 1.4 |

| Plant Family | Genera | **GSS study area**[a] | | **Swan Coastal Plain**[a] | | **SWAFR**[b] | |
|---|---|---|---|---|---|---|---|
| | | **Number** | **%** | **Number** | **%** | **Number** | **%** |
| | *Dryandra*[c] | 9 | 0.7 | 34 | 1.2 | 136 | 1.8 |
| **Fabaceae**[d] | | **80** | **6.0** | **180** | **6.6** | **540** | **7.3** |
| **Mimosaceae** | | **43** | **3.2** | **98** | **3.6** | **503** | **6.8** |
| | *Acacia* | 41 | 3.1 | 95 | 3.5 | 502 | 6.8 |
| **Orchidaceae** | | **83** | **6.2** | **173** | **6.3** | **374** | **5.1** |
| | *Caladenia* | 20 | 1.5 | 54 | 2.0 | 162 | 2.2 |
| **Ericaceae** | | **40** | **3.0** [e] | **93** | **3.4** [e] | **297** | **4.0** [f] |
| | *Leucopogon* | 18 | 1.3 | 40 | 1.5 | 165 | 2.2 |
| **Asteraceae** | | **80** | **6.0** | **116** | **4.2** | **280** | **3.8** |
| **Goodeniaceae** | | **32** | **2.4** | **68** | **2.5** | **207** | **2.8** |
| **Cyperaceae** | | **85** | **6.4** | **131** | **4.8** | **199** | **2.7** |
| **Stylidiaceae** | | **48** | **3.6** | **102** | **3.7** | **178** | **2.4** |
| | *Stylidium* | 45 | 3.4 | 97 | 3.5 | 170 | 2.3 |

[a] Data derived from Western Australian Herbarium specimen records (2008) of native vascular plants using familial groupings as per Hopper and Gioia (2004)

[b] Data obtained from Hopper and Gioia (2004)

[c] Records from former *Dryandra* spp. that are now included in the *Banksia* genus (Mast and Thiele 2007)

[d] Fabaceae grouping including Papillionaceae and Caesalpiniaceae

[e] Ericaceae count is for Epacridaceae only

[f] Ericaceae grouping includes Epacridaceae (Hopper and Gioia 2004)

# Threatened and priority listed flora

South-west Western Australia has undergone extensive habitat modification that has profoundly altered vegetation communities and the distribution of numerous plant species. The South Western Australian Floristic Region has approximately 2500 species of threatened plants, a figure that eclipses the number of threatened species in other Australian states (Coates and Atkins 2001; Hopper *et al.* 1990). Most threatened taxa in the SWAFR are woody perennials, a third of which are short-lived disturbance opportunists and obligate seeders after fire (Hopper *et al.* 1990). Perennial herbs (mostly orchids) also feature prominently among the threatened taxa (Hopper and Gioia 2004). Several processes threaten plant species, including habitat loss and fragmentation, risk of root-rot disease (*Phytophthora cinnamomi*), invasive plant species, and rising saline watertables (Hopper and Gioia 2004). Some of these threatening processes are explored in greater detail in later chapters.

On the Swan Coastal Plain, species assessments demonstrate that extinctions have occurred at population, regional and global scales (Keighery 1999). The Swan Coastal Plain contains 71 taxa of declared rare flora and 353 taxa that are listed as ‘priority’ (Western Australian Herbarium 1998-2009). Declared rare flora are taxa which have been adequately searched for and are deemed to be either rare or in danger of extinction under the *Wildlife Conservation Act 1950*. Taxa that are not declared rare flora but are considered to be rare or poorly known are listed on DEC’s priority flora list, with a priority ranking from 1 to 4.

Within the GSS study area, 47 threatened plant species were known to be extant in 2008 (Atkins 2008), (Table 2.3, Figure 2.1). Ten of these are gazetted declared rare flora (Figure 2.2), with the remaining 37 threatened taxa listed as priority flora (Table 2.3). Of these species, 19 are locally endemic, being restricted to the Swan Coastal Plain (Table 2.3). A total of 27 species are regional endemics, being restricted to south-west Western Australia, of which 3 are declared rare flora and 24 are priority species. Only one of these species (*Beyeria cygnorum*) is classified as not endemic (Table 2.3). The degree of endemism of declared rare flora may be considered as a method of prioritising threatened taxa. Within the GSS, management actions may rank GSS unique declared rare flora above those declared rare flora found throughout south-west Western Australia (region endemics). These concepts are discussed in Chapter 11.

An example of a GSS unique declared rare flora is the critically endangered *Grevillea curviloba* subsp. *curviloba*. This species, and the subspecies *G. c. incurva*, are listed as endangered under the Commonwealth’s *Environment Protection and Biodiversity Conservation Act 1999* (EPBC) (DEWHA 2009a; b). Populations of *G. c. curviloba* are restricted to near Bullsbrook (Atkins 2008; Brown *et al.* 1998) and are predominantly located on the Pinjarra Plain on the eastern side of the Swan Coastal Plain, within the GSS study area sub-areas of Lake Mungala, Ellenbrook Town, Bullsbrook and East Gnangara (DEC 2008b) (Figure 2.2).

The open heath habitats in which this species typically occurs include winter-wet areas on sand over limestone, winter-wet deep peaty sands, and ironstone sites with a high watertable, with some populations associated with the threatened ecological communities of Muchea limestone (shrubland and woodlands on Muchea limestone) and Northern ironstone (shrublands and woodlands on Perth to Gingin ironstone) (English and Phillimore 2000; Phillimore and English 2000). Threats such as invasive weeds, grazing by introduced and native animals, inappropriate fire regimes and recreational impacts (e.g. trampling by walkers) may diminish the species' resilience to hydrological threats such as reduced rainfall from climate change and alteration of the height of the local watertable.

Table 2.3: Conservation status and endemism of declared rare flora and priority flora in the GSS study area (Atkins 2008; Western Australian Herbarium 1998-2009). Endemic codes: GSS – unique to the GSS study area; LE – Locally endemic to Swan Coastal Plain; RE – regionally endemic to SWAFR; and NE – Not endemic, found elsewhere in Western Australia.

| Conservation status | Scientific name | Endemism |
|---|---|---|
| Declared rare flora | *Caladenia huegelii* | RE |
| | *Darwinia foetida* | LE |
| | *Drakaea elastica* | LE |
| | *Eleocharis keigheryi* | RE |
| | *Epiblema grandiflorum* var. *cyaneum* | GSS |
| | *Eucalyptus argutifolia* | LE |
| | *Grevillea curviloba* subsp. *curviloba* | GSS |
| | *Grevillea curviloba* subsp. *incurva* | RE |
| | *Marianthus paralius* | LE |
| | *Trithuria occidentalis* | GSS |
| Priority 1 flora | *Calectasia* sp. Pinjar (C. Tauss 557) | GSS |
| | *Carex tereticaulis* | RE |
| | *Eucalyptus x mundijongensis* | GSS |
| | *Grevillea evanescens* | LE |
| | *Lechenaultia magnifica* | RE |
| | *Tripterococcus paniculatus* | RE |
| Priority 2 flora | *Acacia benthamii* | LE |
| | *Anigozanthos humilis* subsp. *Badgingarra* (S.D. Hopper 7114) | RE |
| | *Fabronia hampeana* | RE |
| | *Isotropis cuneifolia* subsp. *glabra* | LE |
| Priority 3 flora | *Adenanthos cygnorum* subsp. *chamaephyton* | RE |
| | *Angianthus micropodioides* | RE |
| | *Aotus cordifolia* | RE |
| | *Beyeria cinerea subsp. cinerea* | NE |
| | *Blennospora doliiformis* | RE |
| | *Conostylis bracteata* | LE |
| | *Cyathochaeta teretifolia* | RE |
| | *Dillwynia dillwynioides* | LE |
| | *Haemodorum loratum* | RE |
| | *Hibbertia spicata* subsp. *leptotheca* | LE |
| | *Lasiopetalum membranaceum* | RE |
| | *Meionectes tenuifolia* | RE |
| | *Myriophyllum echinatum* | RE |
| | *Rhodanthe pyrethrum* | RE |
| | *Sarcozona bicarinata* | GSS |
| | *Stylidium longitubum* | RE |
| | *Stylidium maritimum* | RE |
| Priority 4 flora | *Anthotium junciforme* | RE |

| Conservation status | Scientific name | Endemism |
|---|---|---|
| | *Conostylis pauciflora* subsp. *euryrhipis* | LE |
| | *Conostylis pauciflora* subsp. *pauciflora* | LE |
| | *Dodonaea hackettiana* | RE |
| | *Drosera occidentalis* subsp. *occidentalis* | RE |
| | *Grevillea thelemanniana* | RE |
| | *Jacksonia sericea* | LE |
| | *Schoenus natans* | RE |
| | *Stachystemon axillaris* | RE |
| | *Verticordia lindleyi* subsp. *lindleyi* | RE |

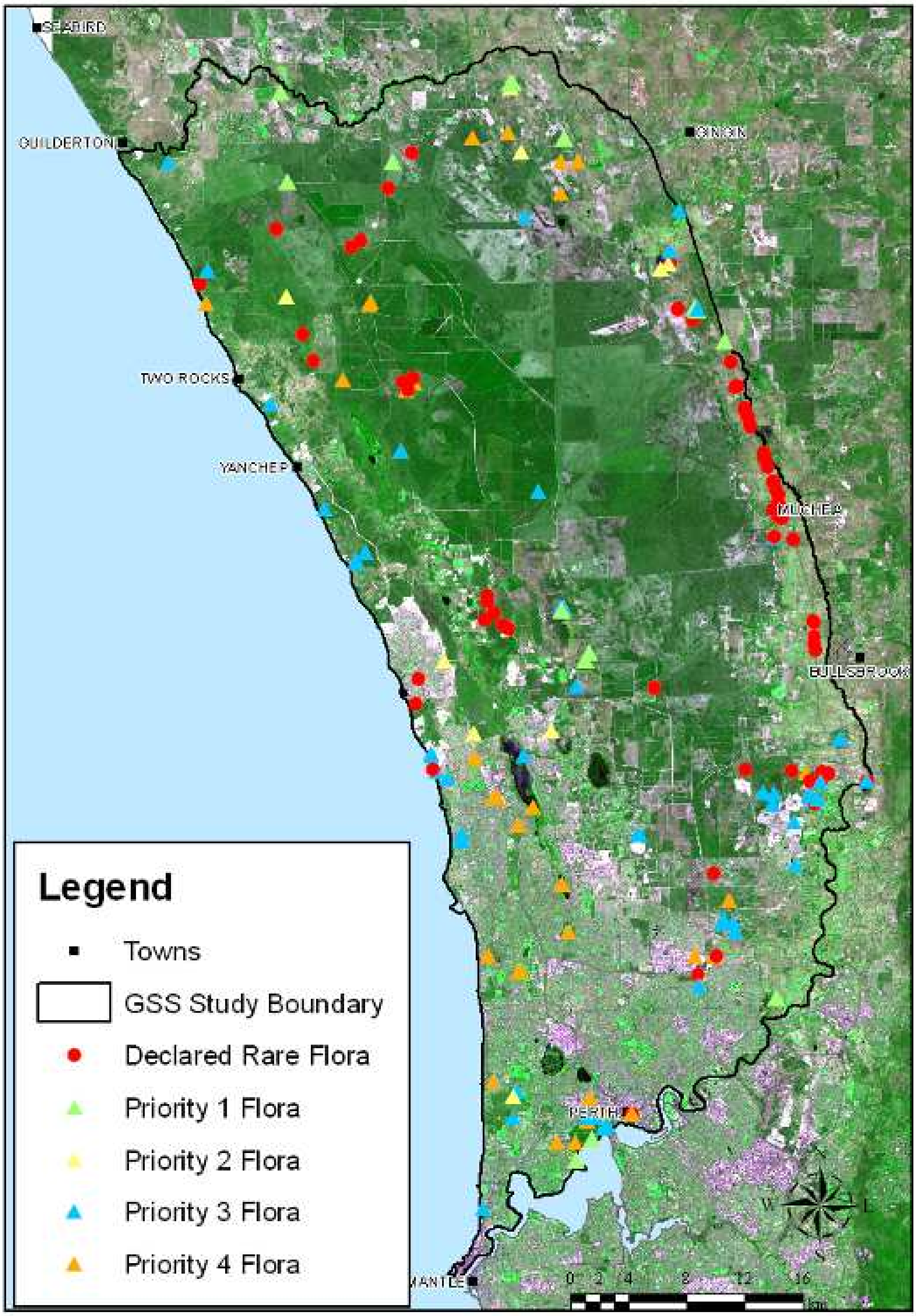


Figure 2.1: Locations of the populations of the 47 declared rare flora and priority flora in the GSS study area.

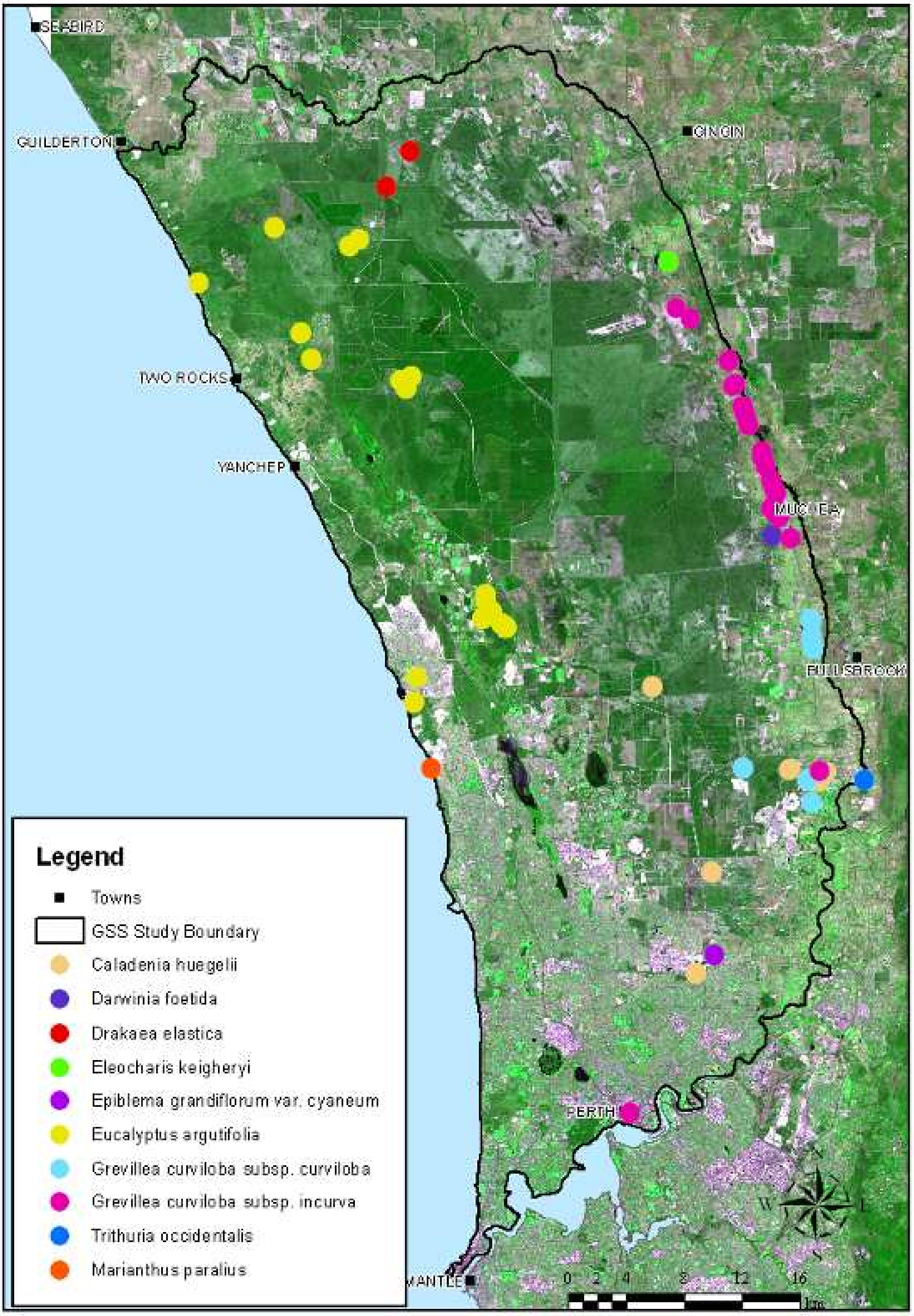


Figure 2.2: Locations of populations of each of the ten declared rare flora species within the GSS study area.

## Threatened ecological community

Ecological communities are typically defined as naturally occurring biological assemblages that occur in a particular type of habitat (Government of Western Australia 2000a). The extent of such communities may vary greatly, and the Threatened Species and Communities Branch of the Department of Environment and Conservation has developed a procedure for identifying which ecological communities are threatened (English and Blyth 1997; 1999). Threatened ecological communities are assessed based on the geographic extent and degree of threatening processes. Depending on the assessment, ecological communities may be defined as 'presumed totally destroyed', 'critically endangered', 'endangered', 'vulnerable', 'data deficient', 'lower risk' and 'not assessed'. A number of the threatened ecological communities in the GSS study area, such as Yanchep caves, are clearly groundwater-dependent ecosystems, and are covered in greater detail in Chapter 4. In this chapter we provide an overview of the occurrences of threatened ecological communities, a brief description of those unique to the GSS study area, and the conservation status of these communities in the GSS.

Floristic community types defined by floristic-based studies (e.g. Gibson *et al.* 1994) can be used as a basis to delineate ecological communities. At least 21 floristic community types (Gibson *et al.* 1994) occur within the GSS study area, and of these, 10 have been identified as threatened ecological communities, four are listed as critically endangered, three are endangered and three are vulnerable (Figure 2.3; Table 2.4). All of the threatened ecological communities that occur in the GSS are restricted to the Swan Coastal Plain. In addition, three of them – Yanchep caves, Organic mound springs and Northern ironstone – are classified as critically endangered and unique to the GSS study area. These threatened ecological communities are briefly described below.

### *Aquatic root mat community of caves of Swan Coastal Plain (Yanchep caves)*

The Yanchep caves, described as limestone karst caves, occur on the Spearwood dune system with groundwater-fed cave streams supporting both tuart root mats and rare Gondwanan relictual stygofauna that rely on the root mats for nutrients (Figure 2.3). The cave communities depend on the Gnangara groundwater system to feed the underwater cave streams that sustain the aquatic root mat community. In addition, the critically endangered crystal cave crangonyctid (*Hurleya* spp.) occurs in association with one of the caves. Persistence of aquatic root mat community depends on the presence of permanent water in the caves. Continued lowering of the groundwater and drying up of cave streams are the key threats to this community. For more detail on the ecological functions and unique fauna of the Yanchep caves, refer to the wetlands chapter (Chapter 4).

### *Communities of tumulus springs (Organic mound springs, Swan Coastal Plain)*

The Organic mound springs occur at the junction of the Bassendean dune system and Pinjarra plain where permanently moist peat mounds accumulate above areas of continuous groundwater discharge (Figure 2.3). These microclimates support discrete assemblages of vegetation and macroinvertebrates that are reliant on a permanent supply of fresh surface and groundwater, such as the Priority 3 species *Aotus cordifolia* which occurs in two of these threatened ecological community occurrences (DEC 2008b). The maintenance of hydrological processes in terms of delivery of both quality and quantity of water to the mounds is thus essential to sustain the tumulus spring communities. This includes the maintenance of the head of pressure to ensure continued flow of the springs and the retention of adjacent native vegetation in the subcatchment. The overall decline in groundwater levels in the Gnangara groundwater system as a result of combination of groundwater uses and low rainfall is a major threat to these communities. Decline in water flow of the mound springs will increase the likelihood of summer fires and destruction of peat mounds and wetland vegetation. A number of occurrences are threatened by clearing of adjacent vegetation and the subsequent changes to the hydrology that sustains these springs.

### *Perth to Gingin ironstone association (Northern ironstones)*

The threatened ecological communities of the Northern ironstone association are found on the Pinjarra plain on the eastern side of the Swan Coastal Plain (Figure 2.3). This community has formed over lateritic ironstone and heavy clay soils that impede drainage and result in a seasonally inundated freshwater habitat. The watertable may also come close to the surface during winter, contributing to the seasonal inundation. The herb layer is a major distinguishing characteristic of the community, and probably would not occur without winter inundation (Meissner and English 2005). A reduction in surface flow as a result of climate change and/or alteration of the height of the local watertable may change the duration or depth of ponding, which in turn may be detrimental to the daisy-dominated herb layer. Clearing, inappropriate fire regimes and weed invasion also threaten these communities (DEC 2008a; Meissner and English 2005). The declared rare flora *Grevillea curviloba* subsp. *incurva* occurs in this community as well as five priority listed taxa, including *Isotropis cuneifolia* subsp. g*labra*, *Grevillea evanescens*, *Meionectes tenuifolia*, *Myriophyllum echinatum* and *Stylidium longitubum* (Meissner and English 2005).

Table 2.4: Threatened ecological communities in the GSS study area (DEP 1996; Gibson *et al.* 1994). Conservation status is given as per the Western Australian *Wildlife Conservation Act 1950*, with EPBC listings provided in brackets. Endemism codes: GSS – unique to the GSS study area; LE – locally endemic to Swan Coastal Plain.

| **Threatened ecological community** | **Conservation status** | **Endemism** |
|---|---|---|
| Aquatic root mat community of caves of Swan Coastal Plain (Yanchep caves) | Critically endangered (endangered) | GSS |
| Communities of tumulus springs (Organic mound springs, Swan Coastal Plain) | Critically endangered (endangered) | GSS |
| Perth to Gingin ironstone association (Northern ironstones) | Critically endangered (endangered) | GSS |
| Woodlands over sedgelands in Holocene dune swales of the southern Swan Coastal Plain (community type 19b described by Gibson *et al.* 1994). | Critically endangered (vulnerable) | LE |
| *Melaleuca huegelii – Melaleuca systena* shrublands on limestone ridges (community type 26a described by Gibson *et al.* 1994) | Endangered | LE |
| *Banksia attenuata* woodland over species rich dense shrublands (community type 20a described by Gibson *et al.* 1994) | Endangered | LE |
| Shrublands and woodlands on Muchea limestone | Endangered (Endangered) | LE |
| Herb rich saline shrublands in clay pans (community type 7 described by Gibson *et al.* 1994) | Vulnerable | LE |
| Forests and woodlands of deep seasonal wetlands of Swan Coastal Plain (community type 15 described by Gibson *et al.* 1994) | Vulnerable | LE |
| *Callitris preissii* forests and woodlands, Swan Coastal Plain (community type 30a described by Gibson *et al.* 1994) | Vulnerable | LE |

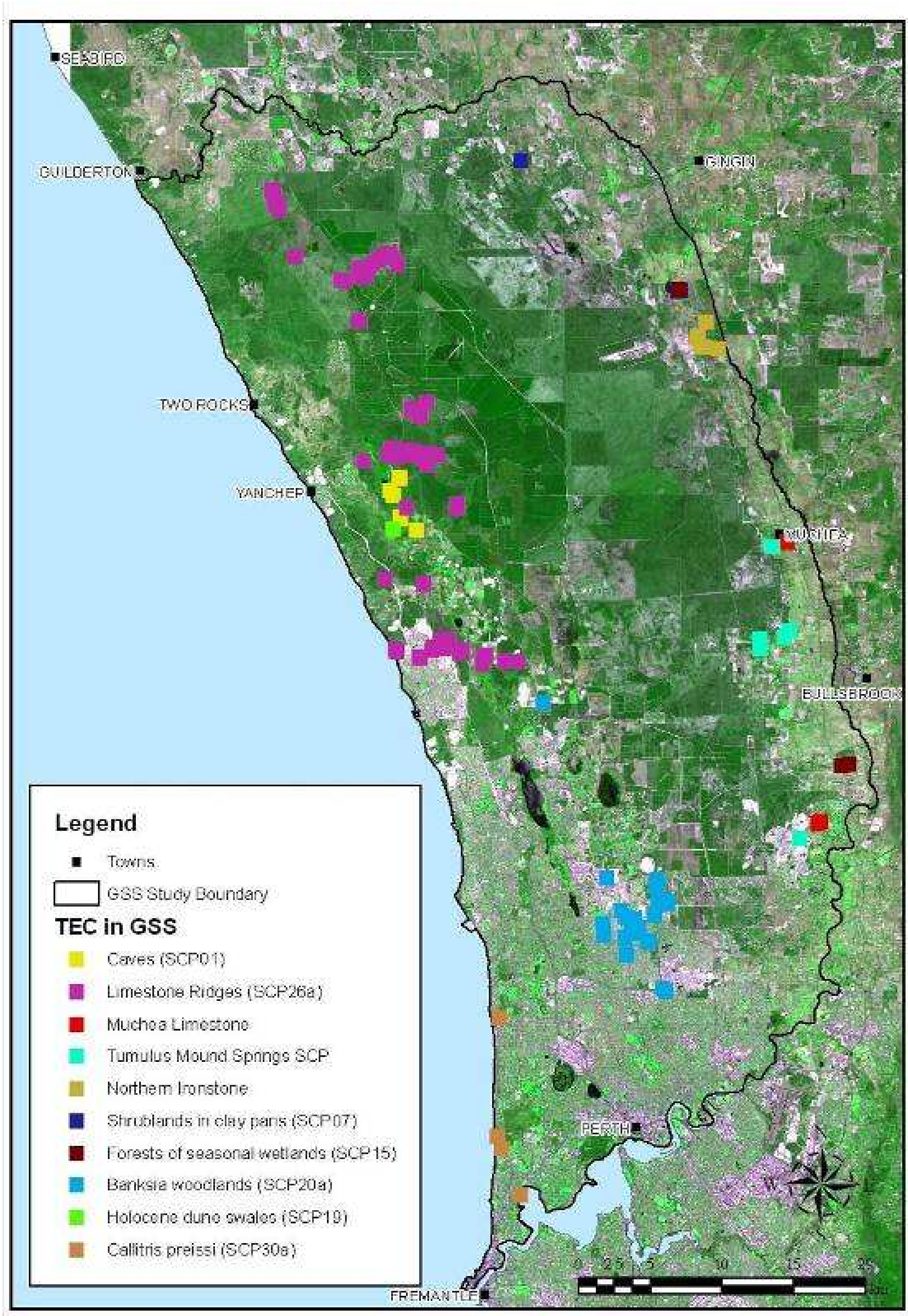


Figure 2.3: Map of threatened ecological communities in the GSS study area.

## Protection of threatened ecological communities within the GSS study area

While the protection of threatened ecological communities within conservation reserves is not guaranteed, their occurrence within conservation reserves should theoretically increase the likelihood of their future survival through active management of the area. We assessed the level of protection of threatened ecological communities in conservation reserves both inside and outside the GSS study area (Table 2.5). For the Yanchep caves threatened ecological community, 6 of the 7 occurrences are located within Yanchep National Park. The seventh occurrence is located on private property and has not yet been adequately surveyed. In lieu of this survey information, we attributed to this occurrence the same details as other occurrences.

Two of the GSS-unique threatened ecological communities, the Yanchep caves and Northern ironstones, occur predominantly within conservation reserves, while only 35.8% of the total area of Organic mound springs is located within conservation reserves (Table 2.5). The one occurrence of Woodlands over sedgelands in Holocene dune (community 19b) in the GSS is located entirely within a conservation reserve and the Forests and woodlands of deep seasonal wetlands (community type 15) have more than 80% of their area in conservation reserves (Table 2.5). However, a number of threatened ecological communities are under-represented in conservation reserves, including Shrublands and woodlands on Muchea limestone and *Melaleuca* shrublands on limestone ridges (community type 26a) both inside and outside the GSS boundary (Table 2.5). Furthermore, none of the occurrences of Herb rich saline shrublands in clay pans (community type 7), *Banksia attenuata* woodland over species rich dense shrublands (community type 20a) and *Callitris preissii* forests and woodlands (community type 30a) are located within conservation reserves in the GSS study area.

‘*Banksia attenuata* woodland over species rich dense shrublands’ (community 20a) is an endangered threatened ecological community occurring on both the Bassendean and Spearwood dune systems. There are 20 known occurrences within the GSS study area, but none are protected within conservation reserves (Table 2.5). In addition, very little of this community is protected across its known extent (Table 2.5). *Banksia* woodland over species rich dense shrublands (community 20a) is threatened by the compounding effects of both climate change and groundwater extraction – lowering of the groundwater level and reduction in soil moisture recharge. The exacerbation of summer drought stress is likely to result in tree deaths, particularly in the *Banksia* overstorey, leading to a change in the structural composition of the community and a subsequent change in species composition in all strata. This threatened ecological community is also considered to be at high risk of the effects of *Phytophthora cinnamomi* (Chapter 9). Other threats that could lead to severe modification and even total destruction of the community include clearing, inappropriate fire regimes and invasive weeds (DEC 2008a).

Table 2.5: Occurrences and amount of threatened ecological communities in conservation reserves within GSS study area and across their known extent. GSS specific data is highlighted.

| **Threatened ecological community** | **Number of occurrences** | | **Extent of community ha** | | **Area and proportion in conservation reserves** | | | |
|---|---|---|---|---|---|---|---|---|
| | Within GSS | Known extent | Within GSS | Known extent | Within GSS | | Known extent | |
| | | | | | ha | % | ha | % |
| Aquatic root mat community of caves of Swan Coastal Plain (Yanchep caves) | 7 | 7 | 0.07 | 0.07 | 0.06 | 85.7 | 0.06 | 85.7 |
| Communities of tumulus springs (Organic mound springs) | 6 | 6 | 17.1 | 17.1 | 6.1 | 35.8 | 4.3 | 25 |
| Perth to Gingin ironstone association (Northern ironstones) | 3 | 3 | 39.0 | 39 | 35.4 | 90.7 | 35.4 | 90.7 |
| Woodlands over sedgelands in Holocene dune swales (community type 19b) | 1 | 38 | 0.2 | 109.8 | 0.2 | 100 | 2.8 | 2.5 |
| *Melaleuca* shrublands on limestone ridges (community type 26a) | 73 | 79 | 138.3 | 148.0 | 33.5 | 24.2 | 33.8 | 22.8 |
| *Banksia attenuata* woodland over species rich dense shrublands (community type 20a) | 20 | 51 | 224.5 | 412.1 | 0 | 0 | 28.6 | 6.9 |
| Shrublands and woodlands on Muchea limestone | 2 | 9 | 28.1 | 175.5 | 0.06 | 0.2 | 65.0 | 37.0 |
| Herb rich saline shrublands in clay pans (community type 7) | 2 | 26 | 15.8 | 210.0 | 0 | 0 | 148.0 | 70.5 |
| Forests and woodlands of deep seasonal wetlands (community type 15) | 3 | 6 | 6.2 | 14.3 | 5.5 | 88.7 | 8.0 | 56.1 |
| *Callitris preissii* forests and woodlands (community type 30a) | 4 | 51 | 16.3 | 529.1 | 0 | 0 | 23.1 | 4.4 |

# Overview of vegetation classifications

## Vegetation complexes

The distribution of plant communities on the Swan Coastal Plain is strongly linked to the underlying landforms, soils, depth to watertable and climate (Cresswell and Bridgewater 1985; Heddle *et al.* 1980). Indeed, the landform units identified by Churchward and McArthur (1980) were used as a basis to define broad vegetation complexes across the Darling System of the South-West Botanical Province (Heddle *et al.* 1980) in five geomorphological provinces, including the Darling Plateau, Swan Coastal Plain, Dandaragan Plateau, Collie Basin and Blackwood Plateau (see Chapter 1). Vegetation complexes represent a combination of repeatedly occurring plant communities on a particular landform unit (as defined by Churchward and McArthur 1980) and, where relevant, the broad rainfall category (low, medium or high). Thus, each vegetation complex is defined in terms of distinctive shared characteristics, including growth-form dominance, species dominance, vertical structure, species composition, landform type and rainfall (Heddle *et al.* 1980). Vegetation complexes have been delineated from data available in the literature, small scale plots and transects, road traverses, aerial photography, previous vegetation interpretation and vegetation maps of Smith (1974) and Havel (1968; 1975). In the Bassendean and Spearwood dune systems, Havel (1968) defined relationships of plant species associations based on the underlying site conditions and local climatic conditions, and Heddle *et al.* (1980) used this work as the basis for describing vegetation complexes of the Swan Coastal Plain.

In the Swan Coastal Plain region, Heddle *et al*. (1980) highlighted several features of the vegetation of the area, including the importance of landforms, soils and climate in determining vegetation types, and the distinct floristics of the region. In addition, a rainfall gradient of increasing aridity from south to north of the Swan Coast Plain and associated changes in vegetation structure and floristics were identified (Heddle *et al.* 1980). In total, 29 vegetation complexes were described for the Swan Coastal Plain, of which 21 occur within the GSS study area (Figure 2.4).

A suite of plant species were used to describe different vegetation complexes (see Appendix 6 for a description of each vegetation complex). Typically, vegetation complexes are dominated by a *Banksia* overstorey and sporadic stands of *Eucalyptus*, *Corymbia*, *Melaleuca* and *Allocasuarina*. The genus *Banksia* is a dominant describing component of 14 of the 21 vegetation complexes in the GSS study area, especially *B. attenuata, B. grandis, B. ilicifolia, B. laricina, B. littoralis, B. menziesii and B. prionotes*. A number of *Melaleuca* species, including *M. cardiophylla, M. cuticularis, M. hamulosa, M. hueglii, M. lateritia, M. rhaphiophylla, M. preissiana* and *M. seriata* were used in the description of 13 vegetation complexes. Dominant overstorey species also include marri (*Corymbia calophylla*), jarrah (*Eucalyptus marginata*), coastal black butt or prickly bark (*E. todtiana*) and flooded gum (*E. rudis*), and to a lesser degree, tuart (*E. gomphocephala*) and wandoo (*E. wandoo*).

Vegetation complexes occur across seven landforms from the Quindalup dunes in the west to the Gingin scarp in the east, and prior to European clearing, accounted for 214 214 ha (Table 2.6). Their pre-European extent shows that three of these complexes occurred entirely within the GSS study area (Bassendean complex – central and south transition, Karrakatta complex – north transition, and, Pinjar complex) and a further two had > 60% of their pre-European extent within the GSS study area (Bassendean complex – north and Yanga complex). Prior to European settlement, the Bassendean complex – north was the most widely distributed complex, covering more than 50 000 ha within the GSS. In addition, the Karrakatta complex – central and south transition, Cottesloe complex – central and south transition and Cottesloe complex – north all covered more than 20 000 ha within the GSS study area. Since European arrival, the GSS study area has been cleared by ~ 50% and we examine the current extent and representation of vegetation complexes in protected areas in the chapter on habitat loss and fragmentation (Chapter 6).

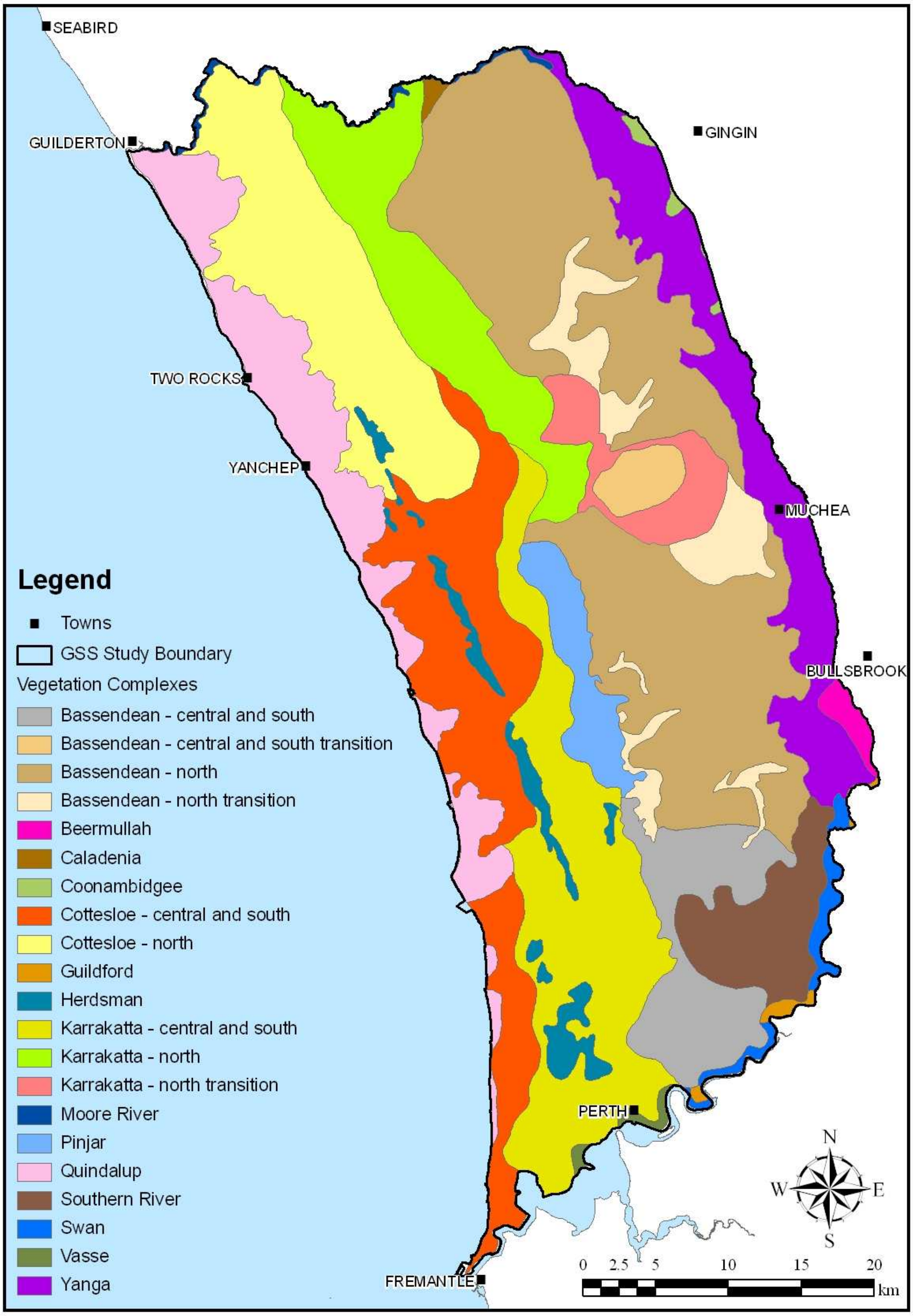


Figure 2.4: Pre-European vegetation complexes in the GSS study area (Heddle *et al.* 1980).

Table 2.6: Pre-European extent of vegetation complexes in the GSS study area. Vegetation complexes highlighted in grey indicate those that are unique to the GSS study area, while those in bold indicate that > 60% is located within the GSS study area.

| Landform | Vegetation complex | Pre-European extent in GSS ha | Proportion of complex in GSS % |
|---|---|---|---|
| Quindalup dunes | Quindalup | 15 843 | 30 |
| Spearwood dunes | Cottesloe – central and south | 21 593 | 48 |
| | Cottesloe – north | 21 399 | 49 |
| | Karrakatta – central and south | 24 284 | 49 |
| | Karrakatta – north | 15 365 | 35 |
| | **Karrakatta – north transition** | **5 260** | **100** |
| Marine (estuarine and lagoonal) deposits | Vasse | 549 | 5 |
| Wetlands | Herdsman | 4 144 | 43 |
| | **Pinjar** | **4 893** | **100** |
| Combinations of Quindalup/Spearwood / Bassendean dunes | Moore River | 797 | 9 |
| Bassendean dunes | Bassendean – central and south | 10 437 | 12 |
| | **Bassendean – central and south transition** | **2 178** | **100** |
| | **Bassendean – north** | **51 920** | **66** |
| | Bassendean – north transition | 7 789 | 37 |
| | Caladenia | 277 | 3 |
| Combinations of Bassendean dunes / Pinjarra plain | Southern River | 7 490 | 13 |
| Pinjarra plain | Beermullah | 1 000 | 15 |
| | Guildford | 486 | 1 |
| | Swan | 1 741 | 10 |
| | **Yanga** | **16 321** | **62** |
| Gingin Scarp | Coonambidgee | 448 | 7 |

The structural formations also vary among vegetation complexes, from tall open forest in the Guildford complex to closed heath in the Cottesloe complex – north. In addition, each vegetation complex may contain a number of structural formations. For example, Caladenia complex may contain a variety of structural formations and species associations, ranging from an open forest of tuart (*E. gomphocephala*) to sedgelands. The three vegetation complexes unique to the GSS study area contain a variety of structural formations. On the Bassendean dunes, the Bassendean complex – central and south transition range from woodlands of jarrah, sheoak and *Banksia* spp. on sand dunes to a closed scrub of *Melaleuca* spp. on low-lying depressions. On the Spearwood dunes, the

Karrakatta complex – north transition consists predominantly of low open forest and low woodland of *Banksia* and coastal blackbutt (*B. attenuata, B. menziesii, B. ilicifolia* and *Eucalyptus todtiana*). In contrast, the Pinjar complex is a wetland associated vegetation complex characterised by woodlands of *E. rudis – Melaleuca preissiana* and sedgelands in depressions, and woodlands of jarrah – banksia on upper dune slopes.

## Floristic community types

A major floristic survey of the patterning of plant communities on the Swan Coastal Plain (south of Gingin Brook) was undertaken by Gibson *et al.* (1994), in a joint project of the Department of Conservation and Land Management (now the Department of Environment and Conservation) and Conservation Council, funded by the Australian Heritage Commission. This work examined plant distribution on the Swan Coast Plain and classified floristic community types based on the presence or absence of individual species in 10 x 10 m quadrats. Quadrats were established on public lands throughout the plain and were located to sample the variety of geomorphic features and plant communities identified in previous studies (e.g. Churchward and McArthur 1980; Heddle *et al.* 1980). At 509 sites, the vascular plant species were recorded, as were information on vegetation structure, vegetation condition and a number of physical parameters. Multivariate analysis of quadrats identified a total of 43 floristic community types, belonging to four major ‘supergroups’ in the area surveyed on the Swan Coastal Plain (Gibson *et al.* 1994). Three of these supergroups were related to major landform elements, and one group was a ‘seasonal wetland supergroup’, that was not restricted to any particular landform elements.

The Gibson *et al.* (1994) classification system was used as the basis for further analysis of quadrats for the System 6 and Part System 1 Update Program (DEP 1996). For this work, an additional 613 quadrats located on the Swan Coastal Plain were examined from a variety of sources: 278 quadrats from the Department of Environmental Protection (1996), 13 quadrats from Trudgen and Keighery (1995), 32 quadrats from Keighery (1996), and 290 quadrats from Griffin (1994).Analysis of all these quadrats using the Gibson *et al.* (1994) methods identified a total of 66 floristic types in the Swan Coast Plain, belonging to the following supergroups:

- Supergroup 1: foothills/Pinjarra plain
- Supergroup 2: seasonal wetlands

- Supergroup 3: uplands centred on Bassendean dunes and Dandaragan Plateau
- Supergroup 4: uplands centred on Spearwood and Quindalup dunes.

One additional community type (S15) described weed groups not allied with any supergroup.

The grouping of floristic community types into supergroups tended to reflect underlying landform element and moisture regimes. The seasonal wetland supergroup contained the highest number of floristic community types and represented the most heterogeneous communities. At least 21 of the identified floristic community types occur within the GSS study area, and of these, 10 have been identified as threatened ecological communities (see the threatened ecological community section above). One limitation of the floristic community types is that the extent of each floristic community within the GSS or the Swan Coastal Plain has not been digitally mapped, limiting the potential use of spatial imagery to determine the extent of each community.

## Site-vegetation types

When assessed in conjunction with abiotic data, the vegetation complexes are sometimes regarded as only moderately successful surrogates for conservation assessments (Keighery *et al.* 2007). The Water and Rivers Commission and the Water Corporation commissioned Mattiske Consulting Pty Ltd (2003) to integrate data available on the flora and vegetation for the Gnangara groundwater system, and to digitally map the vegetation types for part of the Gnangara groundwater system.

Mattiske Consulting Pty Ltd (2003) established a total of 491 floristic survey quadrats (10 m x 10 m) within the Gnangara Mound area, including 298 conducted by Mattiske Consulting Pty Ltd (2003), and 193 collected in earlier studies in the northern Swan Coastal Plain (e.g. DEP 1996; Gibson *et al.* 1994; Griffin 1994; Keighery 1996). Using the quadrat data, Mattiske Consulting Pty Ltd described site-vegetation types based on the system developed by Havel (1968). The site-vegetation types are similar to the description of vegetation complexes (Heddle *et al.* 1980) in that they relate to the underlying soil and site conditions on the Spearwood and Bassendean dunes. In addition, the site-vegetation types incorporate site parameters, key indicator species and local soil preferences. Furthermore, as the site-vegetation types were defined based on 10 m x 10 m quadrats, they are comparable to floristic community types defined by Gibson *et al*. (1994). For example, the site-vegetation type K1 is similar to the floristic community type 4 (*Melaleuca preissiana* damplands) defined by Gibson *et al.* (1994) and occurs on the eastern edges of the Bassendean dune system.

A total of 32 site-vegetation types were defined in the surveyed area of the Gnangara groundwater system, not including open water bodies and pine plantations (Table 2.7 and Figure 2.5). Site-vegetation types ranged from closed heath communities to *Banksia* woodland, and open forests. Site-vegetation types were associated with a major dune system or a wetland element. Seven types of structural formation were also identified. These are:

- open forest
- woodland
- low woodland
- low open woodland
- shrubland
- closed heath
- sedgeland.

A current limitation of the site-vegetation types is that the digital mapping does not cover the entire GSS study area (Figure 2.5).

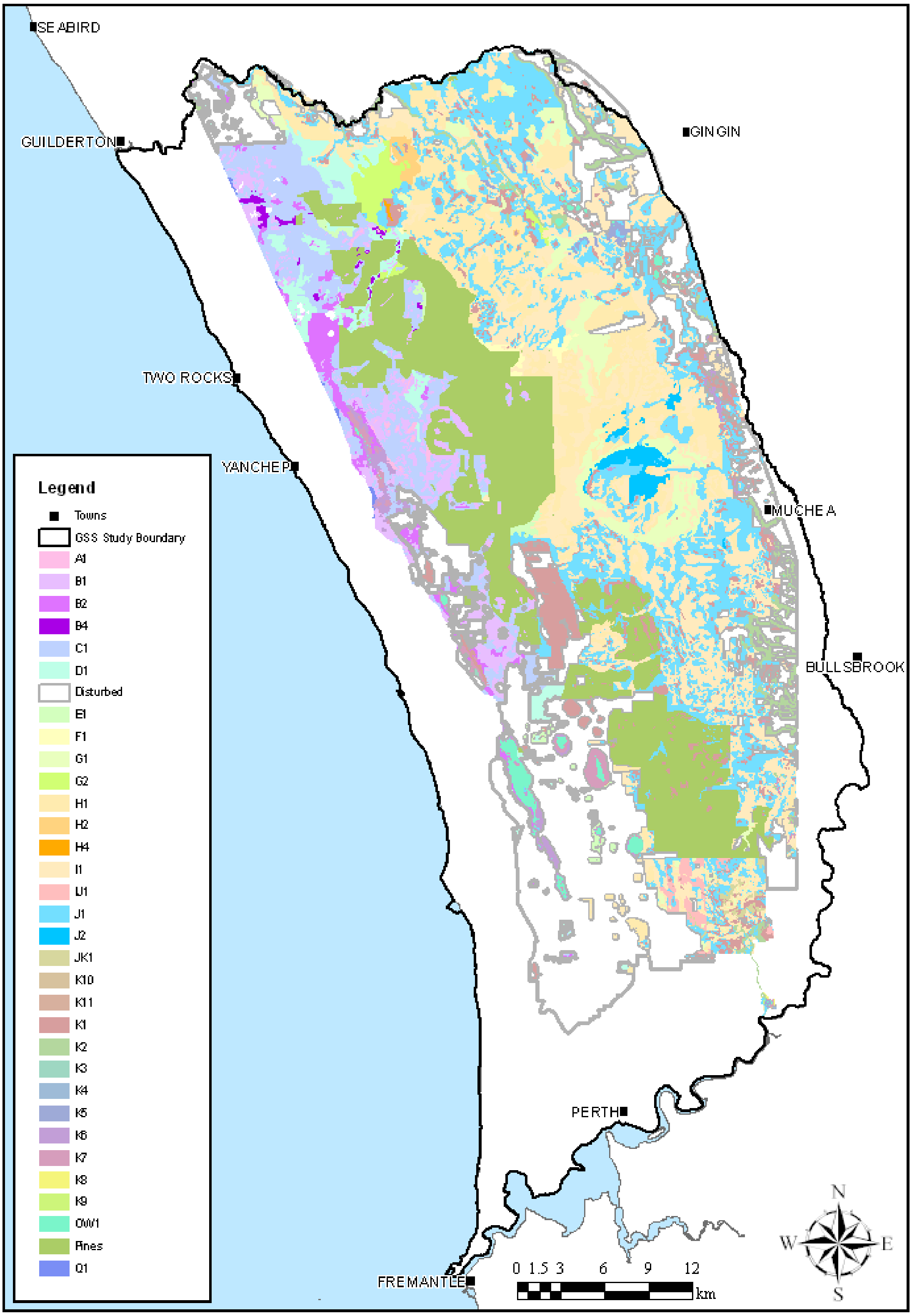


Figure 2.5: Site-vegetation types described within the GSS study area (mapped by Mattiske Consulting Pty Ltd (2003), refer to Table 2.7 for vegetation descriptions).

Table 2.7: Description of site-vegetation types in the GSS study area (Mattiske Consulting Pty Ltd 2003).

| Code | Vegetation description |
|---|---|
| A1 | Closed heath of *Melaleuca huegelii, Trymalium ledifolium, Grevillea preissii* subsp. *preissii, Grevillea vestita* and *Banksia sessilis* |
| B1 | Closed heath of *Jacksonia hakeoides, Conospermum triplinervium, Calothamnus quadrifidus, Melaleuca systena* and *Lechenaultia linarioides* |
| B2 | Woodland of *Eucalyptus gomphocephala – Banksia attenuata – Allocasuarina fraseriana* over *Hibbertia hypericoides, Hakea trifurcata, Conospermum canaliculatum* subsp. *canaliculatum, Melaleuca systena, Macrozamia riedlei, Acacia pulchella* var. *glaberrima* and *Stirlingia latifolia* |
| B3 | Low woodland of *Eucalyptus decipiens* subsp. *decipiens – Banksia attenuata – Allocasuarina fraseriana* with pockets of *Eucalyptus foecunda* over *Hakea prostrata, Hakea ruscifolia, Xanthorrhoea preissii, Daviesia divaricata* and *Scholtzia involucrata* |
| B4 | Shrubland of *Conospermum canaliculatum* subsp. *canaliculatum, Melaleuca systena, Xanthorrhoea preissii* and *Hibbertia hypericoides* |
| C1 | Low open woodland of *Banksia attenuata – Banksia menziesii – Allocasuarina fraseriana* over *Hibbertia hypericoides, Hibbertia racemosa, Hakea costata, Petrophile serruriae, Petrophile brevifolia, Jacksonia hakeoides, Jacksonia sternbergiana, Mesomelaena stygia, Xanthorrhoea preissii* and *Stirlingia latifolia* |
| D1 | Low open woodland of *Banksia attenuata – Banksia menziesii* over *Mesomelaena stygia, Calothamnus sanguineus, Eremaea pauciflora* var. *pauciflora* and *Melaleuca scabra* |
| E1 | Low open woodland of *Banksia attenuata – Banksia menziesii* over *Eremaea fimbriata, Xanthorrhoea preissii, Synaphea spinulosa* subsp. *spinulosa, Stirlingia latifolia* and *Melaleuca scabra* |
| F1 | Low open woodland of *Banksia attenuata – Banksia grandis* over *Conospermum stoechadis* subsp. *stoechadis, Xanthorrhoea preissii, Eremaea pauciflora* var. *pauciflora* and *Jacksonia calcicola* (ms) with pockets of *Banksia attenuata – Banksia menziesii – Nuytsia floribunda* over *Melaleuca systena, Allocasuarina humilis* and *Xanthorrhoea preissii* |
| G1 | Low woodland to low open woodland of *Banksia attenuata – Banksia menziesii – Eucalyptus todtiana – Nuytsia floribunda* with occasional *Allocasuarina fraseriana* and *Banksia grandis* (southern section only) over *Leucopogon conostephioides, Scholtzia involucrata, Eremaea pauciflora* var. *pauciflora, Melaleuca scabra, Boronia purdieana* subsp. *purdieana* and *Astroloma xerophyllum* |
| G2 | Low open woodland of *Banksia attenuata – Banksia menziesii – Allocasuarina fraseriana – Eucalyptus todtiana* over *Xanthorrhoea preissii, Lysinema ciliatum, Verticordia nitens, Hibbertia hypericoides, Philotheca spicata, Eremaea pauciflora* var. *pauciflora, Bossiaea eriocarpa, Daviesia nudiflora, Mesomelaena pseudostygia* and *Stirlingia latifolia* |
| H1 | Low woodland to low open woodland of *Banksia attenuata – Banksia menziesii – Banksia ilicifolia – Nuytsia floribunda* over *Beaufortia* |

| Code | Vegetation description |
|---|---|
| | *elegans, Leucopogon polymorphus, Melaleuca systena, Calytrix angulata, Calytrix flavescens, Stirlingia latifolia, Dasypogon bromeliifolius, Leucopogon conostephioides, Lyginia barbata, Macrozamia riedlei* and *Xanthorrhoea preissii* |
| H2 | Low woodland of *Banksia attenuata – Banksia menziesii* over *Melaleuca viminea, Dasypogon bromeliifolius, Kunzea ericifolia* subsp. *ericifolia, Xanthorrhoea preissii, Phlebocarya ciliata* and *Hibbertia subvaginata* |
| H4 | Low woodland of *Banksia prionotes* over *Adenanthos cygnorum, Calytrix angulata, Verticordia densiflora* var. *densiflora* and *Regelia ciliata* |
| I1 | Low open woodland of *Banksia attenuata – Banksia menziesii* over *Verticordia nitens, Dasypogon bromeliifolius, Melaleuca seriata* and *Patersonia occidentalis* |
| IJ1 | Woodland of *Corymbia calophylla – Banksia attenuata – Banksia menziesii* over *Xanthorrhoea preissii* and *Jacksonia furcellata* |
| J1 | Woodland of *Corymbia calophylla – Banksia attenuata – Banksia menziesii – Melaleuca preissiana* over *Xanthorrhoea preissii, Hypocalymma angustifolium, Pultenaea reticulata, Adenanthos obovatus, Regelia ciliata* and *Jacksonia furcellata* |
| J2 | Woodland of *Corymbia calophylla* over *Xanthorrhoea preissii, Hibbertia subvaginata* and *Gompholobium scabrum* |
| JK1 | Woodland of *Corymbia calophylla – Banksia attenuata – Banksia menziesii – Banksia ilicifolia – Melaleuca preissiana* over *Xanthorrhoea preissii, Hypocalymma angustifolium* and *Jacksonia furcellata* |
| K1 | Open forest of *Eucalyptus rudis* subsp. *rudis – Melaleuca preissiana – Banksia ilicifolia* with occasional *Banksia attenuata, Banksia menziesii, Nuytsia floribunda* and *Eucalyptus todtiana* over *Kennedia prostrata, Lyginia barbata, Xanthorrhoea preissii, Hypocalymma angustifolium, Dasypogon bromeliifolius, Pericalymma ellipticum* var. *ellipticum, Astartea scoparia, Lepidosperma tenue, Jacksonia furcellata, Kunzea ericifolia* subsp. *ericifolia* and *Bossiaea eriocarpa* |
| K2 | Open forest of *Eucalyptus rudis* subsp. *rudis – Melaleuca rhaphiophylla – Banksia ilicifolia* with occasional pockets of *Casuarina obesa, Melaleuca preissiana, Banksia littoralis* over *Baumea juncea, Lepidosperma longitudinale, Regelia ciliata, Hypolaena exsulca* and *Hakea varia* |
| K3 | Open forest to open woodland of *Eucalyptus rudis* subsp. *rudis – Melaleuca preissiana* over *Acacia saligna* and *Hypocalymma angustifolium* |
| K4 | Woodland of *Melaleuca preissiana – Banksia attenuata – Nuytsia floribunda* with the occasional *Banksia menziesii* and *Eucalyptus todtiana* over *Xanthorrhoea preissii, Dasypogon bromeliifolius* and *Jacksonia furcellata* |
| K5 | Closed heath to tall shrubland of Myrtaceae – Proteaceae species including *Acacia saligna, Melaleuca lateriflora, Kunzea ericifolia* subsp. *ericifolia, Astartea scoparia, Regelia ciliata, Kunzea recurva, Hypocalymma angustifolium* over *Drosera* species |
| K6 | Open forest of *Melaleuca rhaphiophylla – Eucalyptus rudis* subsp. *rudis* over *Baumea* and *Meeboldina* species |
| K7 | Sedgeland of *Typha, Baumea* and *Meeboldina* species |
| K8 | Low woodland of *Casuarina obesa* over *Juncus kraussii* subsp. *australiensis* |
| K9 | Low open woodland of *Melaleuca rhaphiophylla* over *Juncus subsecundus, Cyperus eragrostis* and *Centella asiatica* |

| Code | Vegetation description |
|---|---|
| K10 | Low open woodland of *Melaleuca rhaphiophylla – Casuarina obesa* over *Tecticornia* spp. |
| K11 | Woodland of *Eucalyptus rudis* subsp. *rudis, Melaleuca rhaphiophylla* and *Casuarina obesa* over dense *Cyperus eragrostis* with patches of *Juncus* spp. and *Tecticornia pergranulata.* The introduced species *Typha orientalis* occurred on the edges of the open water. |
| Q1 | Shrubland of *Acacia rostellifera, Acacia lasiocarpa, Calothamnus quadrifidus, Melaleuca systena, Phyllanthus calycinus* and *Leucopogon parviflorus* |
| OW | Open water |
| D | Disturbed or modified community |

## Bush Forever – Perth's Bushplan Project

One of the main aims of Perth's Bushland Project (Government of Western Australia 1998) was to develop a strategic plan for conserving representative bushland and associated wetland areas on the Swan Coastal Plain within the Perth metropolitan region. This aim was realised with the Bush Forever report (Government of Western Australia 2000a; b). Bush Forever is concerned with the protection of regionally significant bushland and associated wetlands that are representative of the range of ecological communities within the Swan Coastal Plain portion of the Perth metropolitan region. Bush Forever recognises that survival of bushland sites depends on their size, shape, condition and the management of threatening processes (see Chapter 6 for a review on the impacts of habitat loss and fragmentation).

Information on areas examined for Bush Forever sites was compiled from previous work within the area including the vegetation mapping conducted by Heddle *et al.* (1980), the System 6 and Part System I Update (DEP 1996), and floristic surveys conducted by Gibson *et al*. (1994). With consideration of the land tenure and land-use zoning, sites were then assessed against a number of criteria (Government of Western Australia 2000a). These are:

- representation of ecological communities (based on vegetation complex mapping by Heddle *et al.* (1980)) with a minimum of 10% or 400 ha of each vegetation complex in at least five areas
- level of diversity for flora and/or fauna within a site
- rarity of species within a site, with preference given to sites containing rare or threatened communities or species, or restricted (endemic) species

- maintenance of ecological processes or natural systems at a regional or national scale
- scientific or evolutionary importance, including the presence of fossilised material, relict species, important geomorphological or geological sites or areas recognised as important reference sites for scientific or educational purposes
- conservation of wetland and coastal habitat areas including wetlands, streamlines and estuarine fringing vegetation and coastal vegetation
- additional attributes that may increase a site's contribution to Bush Forever, such as landscape or historical values.

The Bush Forever plan identified some 287 sites in the Perth metropolitan region of the Swan Coastal Plain and represented at least 10% of each of the original 26 vegetation complexes in the area (where possible). Of the 51 200 ha identified as regionally significant vegetation, approximately 33 400 ha already had some level of protection (e.g. conservation reserves or land tenure agreements), with the majority of the remaining habitat (13 200 ha) owned by local governments or private property (Government of Western Australia 2000b).

## Assessing biodiversity values of remnant vegetation

Historically, one of the main requirements of conservation planning has been the need for information on the extent and type of remnant vegetation, as well as the depletion of this vegetation, in order to manage and protect biodiversity assets (Newell *et al.* 2006). More recently the condition (sometimes termed 'quality') of vegetation communities is recognised as an important element in conservation planning and there is increasing requirement to monitor changes in vegetation condition over time (Newell *et al.* 2006).

Vegetation type and condition are widely used as surrogates of biodiversity (e.g. State of environment reporting (Saunders *et al.* 1998)). For example, both vegetation type and condition are incorporated in the 'comprehensive, adequate and representative' system. Design of reserves based on this system depends on the ability of surrogates to reliably predict the distribution of various characteristics of biodiversity that are difficult or expensive to measure. Instead of detailed biological surveys, a particular species and biological community may act as surrogates for all other forms of biodiversity (Burgman and Lindenmayer 1998). However, it should be acknowledged that measures of vegetation type, extent and condition are not necessarily an accurate surrogate for biodiversity (Williams 2005).

Vegetation maps are perhaps the most frequently used surrogates for biodiversity because they may be the only useful information available for an area where few or no biological surveys have been undertaken (Williams 2005). Although vegetation maps are frequently used, the ability of these maps to act as surrogates for biodiversity is rarely validated as it is presumed that the protection of a proportion of each vegetation type will automatically protect sufficient proportions of other organisms (Burgman and Lindenmayer 1998). Vegetation maps may fail as surrogates because the distributions of flora assemblages may not be related to the distribution and abundance of other species (Burgman and Lindenmayer 1998). In addition, in some areas, such as parts of the GSS, detailed spatial mapping of vegetation types is not available.

Despite these shortcomings, several studies and toolkits have used vegetation data as a surrogate for the biodiversity status of remnant vegetation in Australia (e.g. Goldney and Wakefield 1997; Jenkins *et al.* 2000; Oliver *et al.* 2005). Earlier in this chapter we examined the extent of the occurrence of threatened ecological communities in conservation reserves as a surrogate for conservation status, and this may be used when ascribing ranks to biodiversity assets (see Chapter 11). In the habitat loss and fragmentation chapter (Chapter 6), we assess the type and extent of remnant vegetation protected in conservation reserves in the GSS study area as a way of identifying vegetation units (e.g. vegetation complexes) that are not well represented in conservation reserves. In this section, we discuss the use of monitoring vegetation condition for conservation management purposes.

## *Vegetation condition*

Vegetation communities are typically indefinite entities, with all habitat types undergoing some form of modification in species composition and structure. Change is often an integral component of a community. However, some disturbance forces, typically human-mediated (e.g. clearing, fragmentation, changes to fire regimes, introduction of non-native plant and animal species, dieback and other plant diseases, soil movement, changes to water regimes, rubbish dumping, mining, grazing and proliferation of tracks) can compromise the self-maintenance of vegetation (Government of Western Australia 2000a; Keighery 1994).

The concept and assessment of vegetation condition is very complex and can be difficult to measure because it is univariate in nature – that is, it varies on a single scale from ‘good’ to ‘poor’ (Keith and Gorrod 2006). However, vegetation condition that is commonly used to describe biodiversity is not univariate. Assessment of vegetation condition is also a context-dependent concept and includes aspects such as ecological function and biodiversity conservation (Oliver *et al.* 2002). Vegetation condition assessments can also be measured within the context of threatening processes that affect the quality and presence of sensitive and vulnerable plant species, or by using a reference state or ‘benchmark’ (Parkes and Lyon 2006). There are a number of vegetation condition surrogates that may be used to assess biodiversity (Table 2.8). However, all surrogate measures of biodiversity are imperfect, and no one surrogate can adequately capture all of the compositional, structural and functional attributes of biodiversity.

Table 2.8: Possible vegetation condition surrogates that may be used for assessing biodiversity (Jansen *et al.* 2004; Noss 1990; Oliver 2002a; b; Perkins 2002; Tongway and Hindley 2004).

| **Composition** | **Structure** | **Function** |
|---|---|---|
| • Native plant species richness<br>• Native plant species richness by life form<br>• Cover of exotic species<br>• Presence/abundance of problematic weed species<br>• Presence/abundance of threatened plant species<br>• Presence/abundance of increasers and/or decliners<br>• Presence/abundance of nectar or seed resources<br>• Mistletoe abundance<br>• Evidence of introduced animals (e.g. rabbits, foxes) | • Cover by plant life form<br>• Cover by vertical stratum<br>• Number of vegetation strata<br>• Tree diameter distribution<br>• Number of trees with hollows<br>• Volume (or other measure of abundance) of coarse woody debris<br>• Tree growth stage<br>• Basal area of overstorey stems<br>• Canopy height<br>• Abundance of large, dead trees<br>• Litter cover (or other measure of abundance)<br>• Rock cover | • Presence of regeneration<br>• Cover of bare ground<br>• Cryptogam cover<br>• Soil surface stability<br>• Rate of infiltration<br>• Soil compaction<br>• Adjacent land use<br>• Dieback<br>• Soil salinity<br>• Presence/abundance of salt-tolerant plant species<br>• Presence/abundance of plant functional types<br>• Grazing, fire, or logging regimes<br>• Time since clearing<br>• Degree of soil modification<br>• Mistletoe abundance<br>• Perennial plan basal cover<br>• Bioturbation |

### Measuring vegetation condition

In Australia, measuring the condition of vegetation has become an important component of conservation programs. Methods to assess vegetation condition have been developed on a national level (the 'Vegetation assets, states, and transitions or VAST classification) (Thackway and Lesslie 2005) as well as for several states and territories. These include 'Habitat complexity score' (Catling and Burt 1995), 'Habitat hectares' – Victoria (Parkes *et al.* 2003); 'Biodiversity benefits index' (Oliver and Parkes 2003), 'Rapid appraisal of riparian condition' (Jansen *et al.* 2004), 'BioMetric' – New South Wales (Gibbons *et al.* 2005) and 'BioCondition assessment toolkit' – Queensland (Eyre *et al.* 2006).

Vegetation condition can be assessed at a range of spatial scales depending on the desired conservation goals (Briggs and Freudenberger 2006), from site (e.g. stand or remnant), to relatively recent regional-scale assessments that often use remote sensing and spatial modelling (Parkes and Lyon 2006). Gibbons and Freudenberger (2006) considered two levels of vegetation condition assessment methods. They suggested site-scale assessment as a rapid, on-ground method based on easily measured biophysical attributes for an individual plant that can be used as an indicator of vegetation condition. These could be, for instance, plant height, canopy height, the number of trees with hollows, biomass and longevity. These site-scale assessments can be accurate at fine scales, but can be impractical for assessment and monitoring across broad scales (Gibbons *et al.* 2006). In contrast, landscape-scale assessments that use spatial modelling with on-ground assessments of vegetation condition undertaken at individual sites can be spatially interpolated at a coarse scale over large areas using expert knowledge, environmental predictors, or a combination of environmental predictors and data from remote sensing platforms (Gibbons *et al.* 2006).

In Western Australia, several methods to assess vegetation condition are used. These include the following methods (for more details, refer to the source documents listed).

- Keighery (1994) developed a vegetation condition scale that is widely used in rapid assessment techniques of vegetation condition, including projects such as Bush Forever (Government of Western Australia 2000a) and the Perth Biodiversity Project (see Appendix 7 for a description of the vegetation condition scale).

- The *Monitoring and evaluation biodiversity conservation project manual* (Coote 2001) was designed as a guide for managers and technical advisers to plan and design a monitoring and evaluation program for native vegetation and biodiversity management projects.
- The *Local government biodiversity planning guidelines for the Perth metropolitan region* (Del Marco *et al.* 2004) assess vegetation condition by using a rating given to vegetated natural areas to categorise disturbance related to human activities (adapted from Government of Western Australia 2000a).
- The Perth Biodiversity Project has developed a web-based ‘Natural area initial assessment database’ that was designed to collate and analyse data collected on their ‘Natural area initial assessment templates’. This has also been adopted by the South West Biodiversity Project. Data is in the early stages of compilation. This data will incorporate descriptions of benchmark reference sites for major vegetation units on the Swan Coastal plain. (see <www.walga.asn.au/about/policy/pbp/na_templates>).

### Remote sensing – a tool for measuring condition

Spatial modelling and remote sensing are tools that can increase the accuracy and hence the value of assessments of vegetation condition in conjunction with more commonly used site-scale, on-ground assessment methods (Briggs and Freudenberger 2006). An example is the use of Landsat imagery, which can be used to compare spectral data over time to assist with monitoring changes in vegetation condition (Wallace *et al.* 2006). Satellite platforms, such as Landsat, SPOT and IKONOS may be suitable for assessment of woody cover over large areas and for monitoring over time. Satellite imagery, primarily due to its synoptic views of landscapes and multi-temporal sensing, can be suited for monitoring vegetation condition. In addition, the historical archive of satellite imagery for studying landscape change continues to grow, and it now spans almost a third of a century (Skidmore and Prins 2002). However, these platforms can be limited in the characteristics of vegetation that they can detect. Instruments mounted on aircraft (such as multispectral and hyperspectral sensors and airborne laser scanners) can detect more features of vegetation than satellite-mounted sensors because of higher resolutions. However, they are often suited to smaller areas because they are expensive (Gibbons *et al.* 2006).

Over the last decade a Landsat-based remote-sensing monitoring program has been developed called Land Monitor – a multi-agency project producing information products for land management in Western Australia (Behn *et al.* in prep.). This program produces information on two types of vegetation change:

- the extent of perennial or woody vegetation cover and its change through time
- vegetation trends in cover over time, which summarise vegetation history from multiple images.

The Remote Sensing and Image Integration Group (CSIRO) use multispectral Landsat TM imagery to detect changes in vegetation density or cover over time (Wallace and Thomas 1999). They have developed a 'Vegetation index', which is related to vegetation cover and has the capacity to detect not just changes in woody vegetation cover, but also to identify areas of woody vegetation where there is a permanent or long-term decrease in vegetation cover density. To date, only isolated monitoring or knowledge of the whereabouts of vegetation changes has occurred in the open woodlands of the GSS study area (Behn *et al.* in prep.).

Within DEC, the Resource Condition Monitoring – Native Vegetation Integrity Project was developed in 2007 to provide state-wide surveillance monitoring of native vegetation condition, and is part of the resource condition monitoring required under the bilateral agreements between the Australian and Western Australian governments. The aim of this project is to develop the basis for a long-term, large-scale, strategic approach to monitoring and evaluation of native vegetation integrity in Western Australia, by providing suitable evaluation tools and establishing a suite of reference areas. This project is currently in progress and the final methodology for the assessment and monitoring of native vegetation condition in Western Australia has yet to be finalised.

# Summary of DEC–GSS projects (2007–09)

## Project 1. Patterns of floristic diversity in the GSS study area

Technical report reference: Mickle, D and Swinburn, M (2009) *Patterns of floristic diversity in the GSS study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

This project aimed to improve understanding of how floristic diversity varies across landforms, vegetation types and time-since-fire in the GSS study area. Here we present a brief summary of this project. For more detail refer to Mickle and Swinburn (in prep.). Vascular plant species richness and composition were recorded (using methods described in Keighery (1994)) at 36 sites in conservation reserves in the GSS study area. Sites represented the major landform units (Spearwood and Bassendean dunes), vegetation communities (*Banksia* woodland, *Melaleuca* wet or dampland, jarah and tuart woodland; Figure 2.6) and time-since-fire (< 11 years and > 16 years), and were located at sites used for a concurrent fauna survey (see Chapter 3).

Species richness varied among the four vegetation types examined, with the average highest species richness recorded in *Banksia* woodlands and the lowest in *Melaleuca* damplands (Figure 2.7). *Banksia* woodlands are renowned for having high species richness (Dodd and Griffin 1989). Several of the *Melaleuca* sites in our study conformed to Community Type 17, a seasonal wetland vegetation type that typically has low species number (Gibson *et al.* 1994). There was no significant difference in species richness between the different landform types. Nor did plant species richness vary significantly among time-since-fire categories for the four vegetation types, although we further explore patterns in floristic diversity as it relates to time-since-fire in Chapter 7.


Figure 2.6: The different vegetation associations studied during floristic surveys: a) tuart, b) *Melaleuca*, c) jarrah, and d) *Banksia*.

Most sites, especially the *Banksia* woodland sites, had a rich shrub layer below 1 m, and this layer of vegetation was positively correlated with species numbers, indicating that sites containing a dense shrub layer also contained high species richness. These observations were consistent in the jarrah and tuart communities, where the number of over-storey and mid-storey species was lower than the number of taxa in the lower strata. Much of the floristic diversity in the GSS study area is due to the plant species richness on the ground and in the lower, and may be susceptible to disturbances that influence these layers. Patterns of species richness in relation to time-since-fire and the presence of *Phytophthora cinnamomi* are explored in more detail in Chapters 7 and 9 respectively.

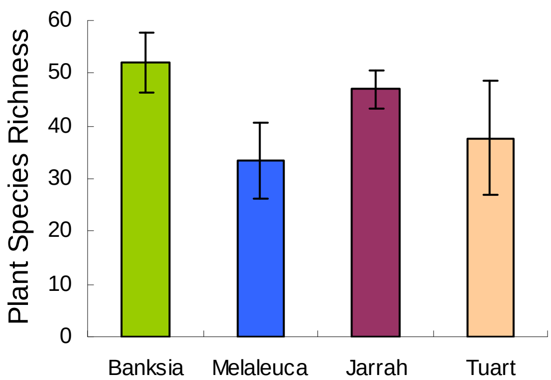


Figure 2.7: Mean (average) plant species richness (± 95% CI) across vegetation types, using untransformed data for ease of interpretation.

# Project 2. Monitoring vegetation condition in *Banksia* woodlands in the GSS study area: the role of remote sensing tools

Technical report references: Kinloch, J, Zdunic, K, Behn, G and Wilson, B (2009) *Monitoring vegetation condition in the* Banksia *woodland of the Gnangara mound: the role of remote sensing tools*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

The aim of this project was to gain a greater understanding of how remote sensing tools can be used in monitoring *Banksia* woodland vegetation in the GSS study area. These woodlands are being affected by a number of threatening processes such as *Phytophthora cinnamomi*, fragmentation, decreasing rainfall due to climate change, groundwater extraction and altered fire regimes. For more informed management actions, there is an urgent need to develop methods that adequately monitor native vegetation condition. The objectives of the project were thus to:

- employ Landsat data to assess and characterise vegetation cover changes over a 36-year time period
- use Landsat and time-since-fire data to determine characteristics of recovery of vegetation after fire
- evaluate Landsat vegetation trends at a landscape scale between 1972 and 2008.

### Vegetation cover dynamics at a small scale

Vegetation cover indexes were obtained using Landsat satellite (MSS, TM and ETM+) imagery for 16 dates between 1972 and 2008. A spectral image index, which was calibrated with ground measurement of projected foliage cover, was used to provide a ‘projected foliage cover’ (PFC) index. The PFC index ranges from 0 to 33, with low values relating to sparse crown cover and high values to dense crown cover.

Study areas were identified within the H1 and I1 vegetation types (Mattiske Consulting Pty Ltd 2003), both of which have a dominant over storey of *Banksia attenuata* and cover a large proportion of the study area (Figure 2.5, Table 2.7). Study areas were described as either ‘undisturbed’ or in ‘variable’ or ‘poor’ condition (e.g. areas mapped as impacted by

*Phytophthora cinnamomi* (Project Dieback 2008a; b)). The fire history of each site, from 1972 to 2007, was then reconstructed using the cover index information and DEC fire database information (i.e. year since last burnt, YSLB).

Fire had the greatest impact on PFC index values. Once sites had regenerated after fire, the PFC index values stabilised for ‘undisturbed’ sites to values ranging from 17 to 25, with small fluctuations in cover index values most likely related to seasonal rainfall differences (Figure 2.8a). The PFC index for ‘poor’ condition sites stabilised to a much lower value, $<= 15$, when an area remained unburnt for a long period (Figure 2.8b). The vegetation cover within ‘poor’ condition sites were more variable as shown by the higher standard deviations of PFC index values compared to ‘undisturbed’ sites. The vegetation dynamics at ‘variable’ condition sites were similar to ‘undisturbed’ sites in that the PFC index showed a similar degree of stability (both between years and within the site) once sites had recovered from fire. However, stabilised PFC index values were generally not as high ($\leq 22$ and usually $< 20$; Figure 2.8c). At the ‘variable’ condition Yeal North study area the PFC index in more recent years has been low, indicating that the condition of the vegetation has deteriorated, most likely due to the combination of two threatening processes – *P. cinnamomi* and recurrent fire (Figure 2.8c).

The lower PFC index values at the ‘poor’ and some ‘variable’ condition sites indicate that the vegetation structure has been altered resulting in less vegetation cover and increased amounts of bare soil compared to ‘undisturbed’ sites (Kaesehagen 1994; Keighery 1994). The higher variability in PFC index values across ‘poor’ condition sites could also indicate that vegetation is more patchily distributed once a disturbing process has altered the vegetation structure.

Recovery of vegetation cover after fire on ‘undisturbed’ and ‘poor’ condition sites was also examined. Only sites that were burnt and then remained unburnt for a minimum of 12 years were included in this analysis. At ‘undisturbed’ sites there was a rapid increase in PFC index values in the first 7 years after fire (Figure 2.9a). At ‘poor’ condition sites there was no consistent relationship between YSLB and PFC index (Figure 2.9b).

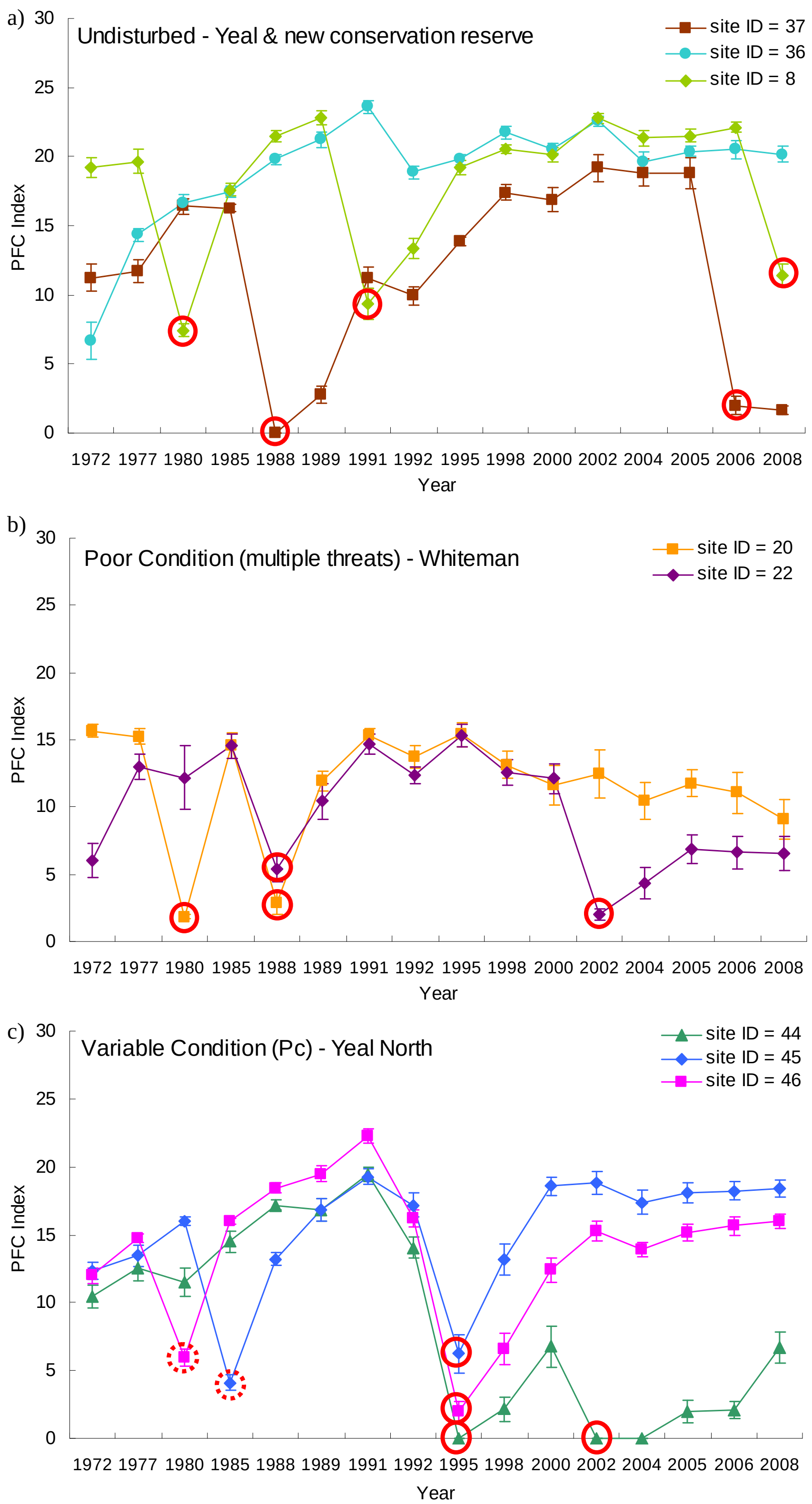


Figure 2.8: Mean (± SD) of PFC index values for 36 years, showing a) 'undisturbed' sites at Yeal and New Conservation Park; (b) 'poor' condition sites at Whiteman Park, with 'poor' condition due to multiple threatening processes including past livestock grazing, *Phytophthora cinnamomi* and high fire frequency, and c) 'variable' condition sites at Yeal

North, with 'variable' condition due to presence of *Phytophthora cinnamomi*. Years with fire are indicated with a red circle, dashed circle indicates uncertainty of year of fire event due to the time between image dates.

The study indicated that two surrogate measures may be useful for monitoring condition: (1) PFC index with thresholds applied to discriminate broad condition classes, and (2) variability of PFC index. These two surrogate measures, however, require testing at a landscape scale, and should be assessed in a broader range of vegetation types, ideally in conjunction with on-ground assessments. In addition, when applying surrogates at the landscape scale, detailed mapping of fire history, vegetation type and *Phytophthora cinnamomi* infection will be required.

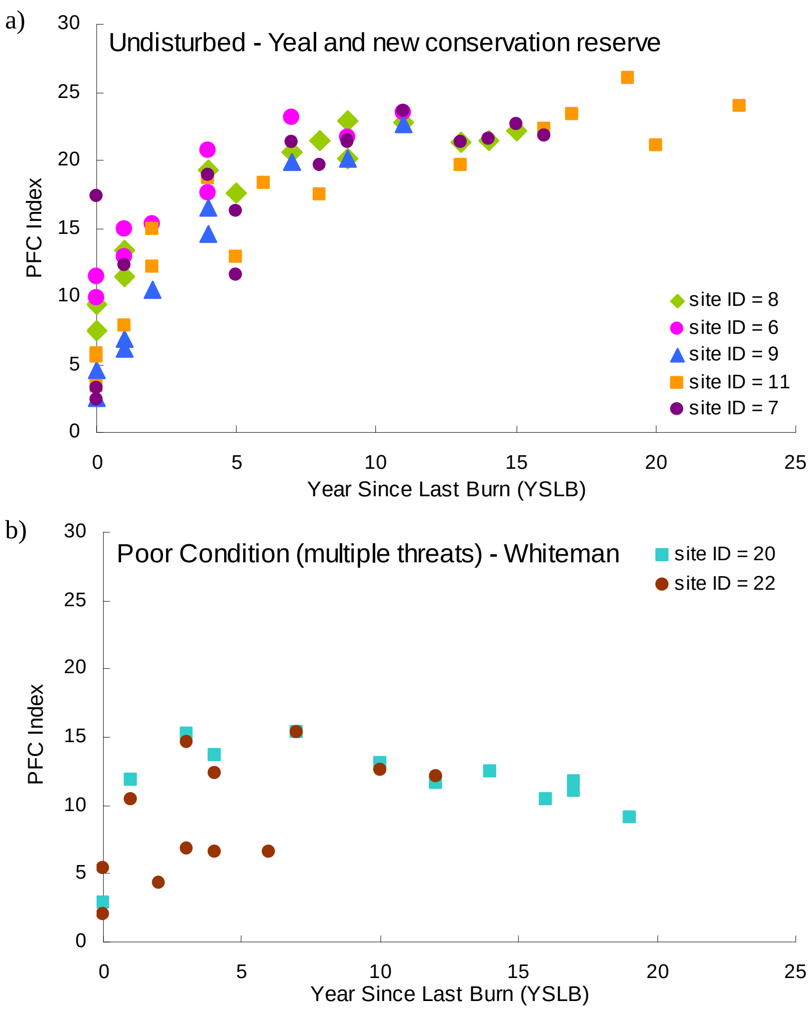


Figure 2.9: Mean PFC index values in relation to year since last burn for (a) undisturbed (new Conservation Park/Yeal), and (b) poor condition (Whiteman Park) sites in vegetation type H1.

### Vegetation cover trends at a landscape scale

Multi-spectral Landsat data is particularly suited to monitoring vegetation changes at the regional-landscape scale as it can assess vegetation cover of large areas. A 'linear vegetation trends' procedure that summarises changes in reflectance of vegetation over a specified time period has been employed to assess vegetation changes over large areas (see Behn *et al.* in prep.; Furby *et al.* 2007; Wallace and Thomas 1999 for a full description of methods). This method classes remnant vegetation into one of five trend categories: major positive, positive, no major change, negative, and major negative. We calculated linear vegetation trends over the GSS study area for three time periods. These were:

- 1972–2008, representing the full range of image dates available in the Landsat archive
- 1972–1992, comparatively wetter years
- 1992–2008, comparatively drier years.

The total area of remnant vegetation (using the 2005 extent of remnant vegetation) in each linear trend class was then calculated for the time periods.

The long-term 1972 to 2008 vegetation trends reveal that the majority of remnant vegetation across the Gnangara groundwater system has either increased in vegetation cover or experienced no major change in vegetation cover since 1972 (Figure 2.10). Patches of remnant vegetation with negative trends near Burns Beach are associated with clearing that occurred. An assessment of linear trends for the period 1972 to 1992 revealed that the area in each trend category was similar to the 1972 to 2008 long-term trend, and they are not displayed here. For the 1992 to 2008 period the area of vegetation in each category differed substantially from those in the long-term period (1972 to 2008). Although 80% of the remnant vegetation was stable, there was a major decline in area exhibiting increased vegetation cover, and an increase in area exhibiting declines in cover, particularly in the north-east of the GSS study area. The decrease in positive trends during the 1992 to 2008 period may have been caused by declining rainfall or increased prescribed burning since 2002 (P. Brown pers. comm.), and requires further analyses.

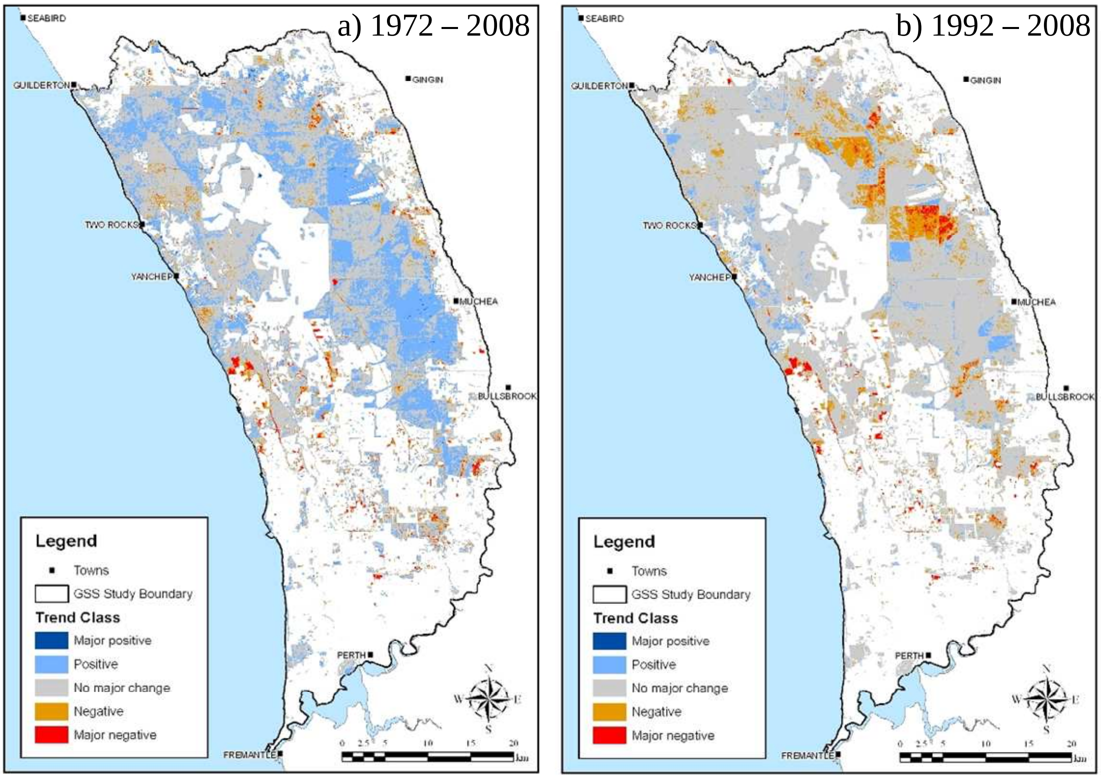


Figure 2.10: Linear trends of the PFC index for (a) 1972–2008, (b) 1972–1992 and (c) 1992–2008, for the extent of remnant vegetation across the GSS study area.

The comparison of vegetation trends over three time periods across the Gnangara groundwater system has revealed that this type of analysis does provide a broad indication of vegetation cover gain and loss at a regional scale. In situations where there are multiple threatening processes, it is difficult to separate out the effects of each processes at a regional scale. However, Landsat trends analysis is very informative at a smaller scale, and may be applied within the boundaries of a conservation or national park, such as Yeal Nature Reserve. At this smaller scale, the analysis outcomes can be assessed in terms of on-ground information known for the locality regarding the impacts from threatening processes or management interventions. Further development of trend analyses and its applicability is required.

# Project 3. Remnant vegetation in the pine plantations and Lake Pinjar

Technical report reference:
1) Brown, PH, Sonneman, T and Kinloch, J (2009) *Ecological linkages within the pine plantations on the Gnangara groundwater system.* Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

2) Sonneman, T (in prep.) *Remnant vegetation and ecological linkages for Lake Pinjar*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

This project aimed to identify the ecological values of patches of remnant vegetation, based on remnant patch condition and size, in the pine plantations of the GSS study area and the Lake Pinjar area. The creation of ecological linkages has been proposed as one land-use option following the removal of the Gnangara, Pinjar and Yanchep pine plantations (Government of Western Australia 2009). Located within each pine plantation are several patches of remnant vegetation that could provide a focal point for rehabilitation efforts. To aid in the delineation of ecological linkages within the pine plantations of the GSS study area and in the Lake Pinjar area, a rapid assessment of vegetation condition was undertaken using the Keighery (1994) scale. The assessment was used in conjunction with other environmental attributes (e.g. patch size and perimeter to area ratio) to determine which patches of vegetation had the highest conservation value.

The pine plantations are located within four subareas in the GSS study area (Gnangara, West Gnangara, West Pinjar, East Yanchep). Overall, there were a greater number of bushland patches rated as 'excellent' (67%) than any other class, followed by the 'excellent–very good class' (15%). The vegetation condition rating was not evenly distributed amongst the four subareas (Figure 2.11), with the Gnangara subarea containing the least amount of remnant vegetation in very good to excellent condition. In addition, position in the landscape, namely uplands versus wetlands, influenced vegetation condition rating. Of the remnant patches located in uplands, 89% were classed as excellent or excellent–very good, but only 37% of the wetlands were considered to be in these condition classes.

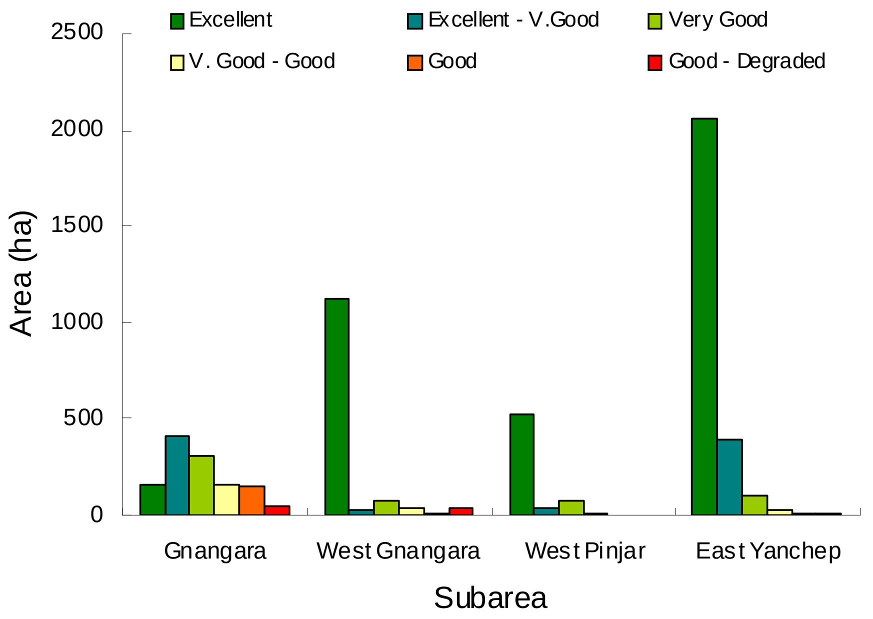


Figure 2.11: Vegetation condition attributes for bushland remnants in each of the Gnangara Sustainability Strategy subareas containing pine plantation.

Over 1500 ha of remnant vegetation was considered bushland in the Lake Pinjar area, with ~ 53% considered to be in excellent condition (Figure 2.12). In addition, the majority of upland remnant vegetation was considered to be in excellent condition. However, the wetland remnant vegetation contained a mixture of condition classes, with substantially less habitat categorised as in excellent condition. This data was incorporated in analyses to define the boundaries of potential ecological linkages that may be established post-pine removal (see Chapter 6).

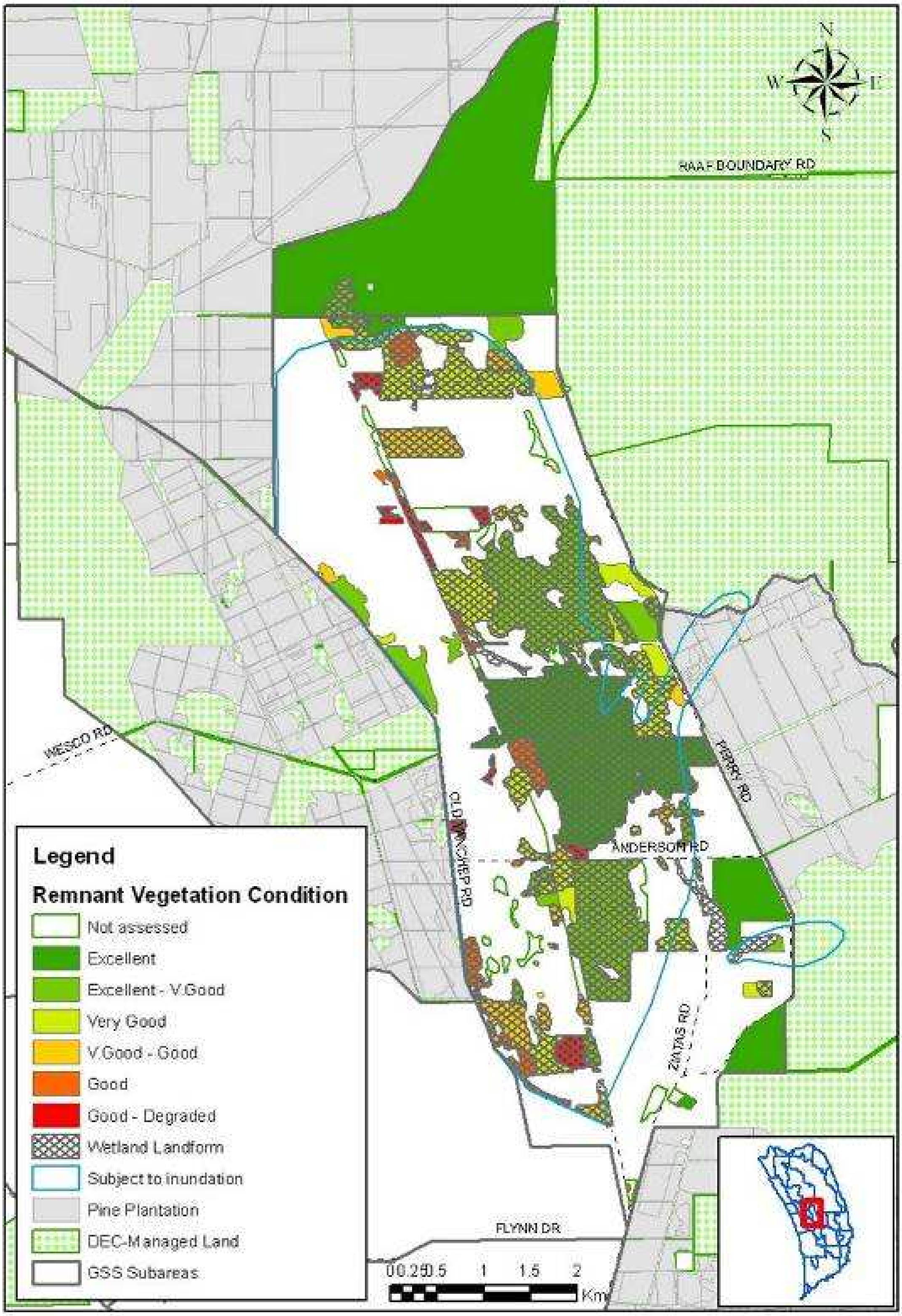


Figure 2.12: Vegetation condition ranking for the remnant vegetation within the Lake Pinjar area.

# Discussion

The GSS study area is located on Perth's doorstep and within the south-west Western Australian biodiversity hotspot (Mittermeier *et al.* 2004). However, the floristic biodiversity values for this area have not recently been collated in one document. The aims of this chapter were to compile information on the floristic biodiversity and vegetation units, identify rare and priority species, ecological communities and vegetation units, and to assess the role of remote sensing in measuring vegetation condition in the GSS study area.

This review confirmed that the GSS study area is floristically diverse, with several endemic or threatened plants, ecological communities and vegetation units. Although the size of the GSS study area is only one-sixth that of the Swan Coastal Plain, it contains nearly 50% of the native vascular plants recorded in the Swan Coast Plain. The GSS study area contains 17% of the floristic diversity found in the South Western Australian Floristic Region, which has an area 140 times greater. This indicates clearly just how important the GSS study area is for the conservation of floristic diversity.

As a number of floristic surveys have previously been conducted in the GSS study area, this review originally aimed to identify patterns in floristic diversity amongst landform types and vegetation units. However, a major constraint of our review was access to these data sets. Floristic surveys that were conducted as part of the DEC–GSS project (2007–09) identified that plant species richness varied among vegetation types (Mickle and Swinburn in prep.), and confirmed that *Banksia* woodlands contained most plant species (Dodd and Griffin 1989), typically within the lower strata of vegetation. Future work should include further analyses of the floristic dataset, particularly with regard to species composition between vegetation types.

A number of rare and priority plant taxa were identified within the GSS study area, including 10 declared rare flora and 37 priority flora, of which, six are unique to the GSS study area. In addition, ten threatened ecological communities were recorded in the GSS study area, including the critically endangered, and GSS unique, aquatic root mat community of caves of Swan Coastal Plain (Yanchep caves), community of tumulus springs (Organic mound springs) and Perth to Gingin ironstone association (Northern ironstone). Of these, both the Northern ironstone and Yanchep caves are predominantly located in conservation reserves. However, only 35% of the extent of Organic mound springs is located within conservation reserves, indicating an inadequate level of protection for this unique threatened ecological community. In addition, more than half of the known extent of *Banksia attenuata* woodland over species rich dense shrublands (community 20a) occurs within the GSS study area.  However, none of this species rich threatened ecological community is located in conservation reserves within the GSS study area and only ~ 7% is conserved throughout its known extent. These threatened ecological communities could be regarded as priorities for inclusion in conservation reserves within the GSS study area.

There are numerous ways to describe vegetation units within the GSS study area, including vegetation complexes (Heddle *et al.* 1980), floristic community types (Gibson *et al.* 1994), and site-vegetation types (Mattiske Consulting Pty Ltd 2003). The extent of digital mapping of vegetation units varies considerably, and only the broad vegetation complexes have been mapped across the entire GSS study area, limiting the potential use of spatial imagery to determine the current extent and representation of vegetation types within conservation reserves. However, within the GSS study area we identified 21 vegetation complexes, at least 21 floristic community types, and 32 site-vegetation types. Of the vegetation complexes, three are unique to the GSS study area, and a further two had greater than 60% of their pre-European extent within the GSS study area

Vegetation type, extent and condition are often used in conservation monitoring programs (Saunders *et al.* 1998), and for identifying priority habitat for conservation management. Remote sensing has the potential to be a very informative tool for measuring vegetation condition, and in the GSS study area, remote sensing techniques, developed by Behn *et al.* (in prep.), detected at small scales, changes in vegetation condition due to the combination of *P. cinnamomi* and recurrent fire frequency (Kinloch *et al.* 2009).

In addition to utilising remote sensing tools, the ecological values of remnant habitat in pine plantations and the Lake Pinjar area of the GSS study area was assessed based on remnant patch condition (assessed using on-ground methods) and size. This work established that remnant vegetation was predominantly in ‘excellent’ condition, although the amount varied between subareas and was somewhat dependent on position in the landscape (Brown *et al.* in prep.; Sonneman in prep.). Compared to uplands, the wetland remnant vegetation was mostly in poorer condition. Vegetation condition provides an index to the health of remnant habitat, and can be incorporated as a value to rank bushland patches and, consequently, may provide a method of identifying priority habitat.

## Recommendations

In order to compile knowledge on the flora and vegetation units of the GSS study area, it is recommended that:

- floristic survey analyses be completed, particularly those of species composition in relation to time-since-fire, and different vegetation types
- a centralised database be created for floristic surveys conducted within the GSS study area, and the Swan Coastal Plain
- spatial mapping of finer-scale vegetation units be further developed (e.g. site-vegetation types or floristic types) across the GSS study area and the Swan Coastal Plain.

In order to ensure adequate conservation of flora and vegetation units of the GSS study area, it is recommended that:

- conservation priorities be determined for flora and vegetation units of the GSS study area (possible priorities are discussed in Chapter 10), possibly incorporating threatened ecological communities currently under-represented in protected area (e.g. Organic mound springs and *Banksia attenuata* woodland over species rich dense shrublands (community 20a) in future conservation planning
- the applicability of remote sensing tools for monitoring vegetation condition be further explored. This would best be done by studying conservation reserves where there is existing knowledge of threatening processes.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter Three: Fauna Biodiversity

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Department of Environment and Conservation

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Our environment, our future

September 2009

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# 3. FAUNA BIODIVERSITY

## Key points

- The diversity of terrestrial invertebrate fauna of the GSS is expected to be high, and currently includes three threatened taxa, one of which is the graceful sun moth (*Synemon gratiosa*).
- Historically, the GSS study area had high vertebrate species richness, with at least 13 species of frogs, 64 species of reptiles, 176 species of birds (excluding seabirds), and 33 native species of mammals recorded in the area. Of the extant fauna in the GSS study area, 12 frog, 27 reptile, 9 bird and 2 mammal taxa are considered regional endemics to south-west Western Australia. Of these taxa, one frog and seven reptiles are restricted to the Swan Coastal Plain.
- The loss of wetland and dampland habitats is expected to have reduced the range and abundance of frog species. In addition, the critically endangered western swamp tortoise (*Pseudemydura umbrina*), which is restricted to the GSS study area, has undergone a significant reduction in range due to habitat loss.
- Populations of nearly 80 species of birds have declined since European settlement, including 13 species now considered locally extinct or very rare. Of these, 18 are listed as threatened or priority taxa. Wetland birds have been particularly affected by large-scale losses of coastal plain wetlands. The GSS study area is an important foraging area for the endangered Carnaby's black-cockatoo (*Calyptorhynchus latirostris*).
- The mammal taxa in south-west Western Australia have undergone a large number of species extinctions and declines, due to habitat loss and fragmentation, and predation by the introduced European fox and feral cat. Of the 28 non-bat mammals historically recorded, 10 species are still persisting in the GSS study area, including four threatened or priority listed taxa.
- At least 88 vertebrate taxa have been identified as likely to decline in abundance or to experience range contractions due to their high or very high dependence upon groundwater. Of these, 68 species were considered to have a very high dependence on groundwater including 5 frogs, 2 reptiles, 60 birds and 1 mammal.

- A DEC–GSS vertebrate fauna survey was undertaken in 2007-08 to assess the occurrence and distribution of ground-dwelling vertebrate fauna in the relatively undisturbed bushland in the northern half of the GSS study area. Skinks were the most commonly caught taxa. Mammal captures were low, with the exception of the native honey possum (*Tarsipes rostratus*) and the introduced house mouse (*Mus musculus*).
- A DEC–GSS targeted survey was undertaken in 2008 to locate populations of the southern brown bandicoot (*Isoodon obesulus fusciventor*; quenda) and water rat (*Hydromys chrysogaster*; rakali). The highest numbers of quenda were recorded at Twin Swamps Nature Reserve, the only site that is fenced and baited against introduced predators. Rakali were captured at three lakes within the study area.
- A DEC–GSS survey was conducted to assess the status of frogs in the GSS study area, and to review the biology and current distribution of these frogs. Frog species that are reliant on permanent or near-permanent wetlands may have already suffered some local contractions in range, while frog species that rely on seasonal wetlands will most likely be affected by slight or moderate hydrological changes.

# Introduction

As part of the south-west Western Australian biodiversity hotspot, the GSS study area is located in a region that is globally recognised for its diversity values, endemism rates and loss of habitat (Mittermeier *et al.* 2004). South-west Western Australia provides habitat for > 500 vertebrates, of which nearly one-sixth are endemic ((Mittermeier *et al.* 2004). The GSS study area is located on the IBRA Perth Coastal Plain subregion SWA2 (Mitchell *et al.* 2003), and the IBRA region is hereafter referred to as the Swan Coastal Plain. This area is know for its richness of fauna (Government of Western Australia 2000; Western Australian Museum 1978), with a particularly diverse reptilian fauna (How and Dell 1993; 1994; 2000; Storr *et al.* 1978a) and historically diverse mammal fauna (Abbott 2006; Kitchener *et al.* 1978).

Mitchell *et al*. (2003) identified gaps in the data on faunal biodiversity values for the Swan Coastal Plain. These included a lack of uniform coverage of surveys across the subregion and the absence of any long-term survey data for most reserves. A comprehensive fauna survey has not been conducted since the late 1970s (Western Australian Museum 1978), and the current status, including persistence, of fauna in the GSS study area is unclear.

There are also limited data on the habitat requirements of persisting mammals within the critical weight range, uncommon vertebrate species and all invertebrate species. Another significant gap identified is the paucity of quantitative data on the impacts of exotic predators, fragmentation and fire on fauna, and the impacts of these threats at landscape scales. In addition, although there is a substantial amount of information on some terrestrial vertebrates (e.g. *Pseudemydura umbrina*, western swamp tortoise), the current overall understanding of biodiversity values, particularly at a landscape scale, is limited.

The need to collate existing data on the occurrence of species on the Gnangara groundwater system and to assess the spatial distribution of taxa and communities in much greater detail was acknowledged in the Gnangara Sustainability Strategy Situation Statement (Government of Western Australia 2009). It was recognised that the lack of data on fauna habitats and their distribution at a landscape scale would limit the ability to ascertain important areas for retention and protection of fauna, and to determine microhabitat requirements essential for restoring landscapes and restoring ecological linkages across the fragmented landscape (Government of Western Australia 2009).

In this chapter, we discuss species that are principally terrestrial, including reptiles, frogs, mammals, birds and invertebrates. The chapter covers the species richness, endemicity, conservation status, and patterns of decline and local extinction of the specific fauna groups in the GSS study area. Due to a lack of information, bats are not reviewed in this chapter. In addition, marine bird species are not reviewed in this chapter. Wetland birds and turtles are discussed in this chapter, and they are also mentioned, along with fish, in Chapters 4 and 5. Threats and landscape-scale issues (e.g. fragmentation, introduced predators, inappropriate fire regimes and climate change) relating to these fauna groups are discussed in Chapters 6 to 10.

The aims of this chapter are to:

- collate and review the available information on terrestrial fauna (reptiles, frogs, mammals, birds and invertebrates) in the GSS study area
- address knowledge gaps in the occurrence, distribution and conservation status of fauna in the GSS study area
- identify threatening processes.

These aims were addressed by:

- reviewing databases and literature on the historical occurrence and distribution of selected terrestrial fauna groups (invertebrates, frogs, reptiles, birds and mammals) in the GSS study area
- identifying the conservation status of terrestrial fauna taxa in the GSS study area
- reviewing the patterns of decline of terrestrial fauna on the Swan Coastal Plain
- identifying taxa that may be susceptible to declining rainfall and groundwater levels.

In addition, a number of DEC–GSS projects were undertaken to address identified knowledge gaps for the GSS study area. These projects are outlined below.

1) **Patterns of ground-dwelling vertebrate biodiversity in the GSS study area**
   This project assessed the current occurrence and distribution of terrestrial vertebrate fauna across the GSS study area. It examined patterns in biodiversity to broad landscape features including landforms and vegetation types. It examined patterns in biodiversity and selected species to site characteristics including habitat and microhabitat structure and floristics, and it assessed the susceptibility of taxa and communities to threatening processes such as declining groundwater levels, habitat fragmentation, fire, and plant pathogens.

2) **Targeted surveys for two mammal species in the GSS study area**
   Targeted trapping surveys for quenda (*Isoodon obesulus fusciventor*) and rakali (water rats; *Hydromys chrysogaster*) were undertaken in May 2008. Quenda were targeted because of the lack of captures during the GSS terrestrial vertebrate survey, and to confirm reports indicating that the species appears to be linked to wetland- associated vegetation. Rakali were studied to determine if the species is persisting within the study area and, if so, their distribution. As the only known aquatic mammal of inland waters in south-west Western Australia, their presence in a wetland or waterway could be a useful indicator of the health of wetland systems.

3) **Assessing the occurrence and status of frogs in the GSS study area**
   The focus of this project was to assess the status of frogs in the GSS study area, and to review the biology and distribution of these frogs. The current distribution of frog species across the GSS study area was examined in relation to environmental parameters such as soil, vegetation, presence of water, seasonal variations in water

levels, presence of other species and water quality. The biology of each species was reviewed in order to assess its likely sensitivity to hydrological change.

The major findings of these projects have been summarised in this document. The full reports are provided as a list of technical reviews in Appendix 2. In addition, several reviews have been completed as part of this project (see Bamford and Huang 2009; Davis 2009b; Huang 2009; Reaveley 2009) that provide information regarding the occurrence and distribution of ground-dwelling vertebrate groups on the Swan Coastal Plain.

## Species richness, endemicity and threatened taxa

### The faunal diversity of south-west Western Australia

The south-west Western Australia biodiversity hotspot is species rich and provides habitat for some 573 vertebrates, of which ~ 15% are endemic to the region (Mittermeier *et al.* 2004). The high degree of endemicity is particularly well illustrated in the freshwater fish and frog taxa, with approximately 20 species of freshwater fish and 32 species of frogs, of which ~ 50% and 80% respectively are endemic to south-west Western Australia (Mittermeier *et al.* 2004). Reptile fauna is particularly diverse, with an estimated 177 taxa, of which ~ 15% are considered endemic to the south-west Western Australia region. The bird taxa is also diverse with ~ 285 species (excluding marine birds), including 13 species endemic to south-west Western Australia. Similarly, the mammal fauna is historically diverse with 59 species recorded, including 12 endemics (Mittermeier *et al.* 2004). In our report we use the following categories of endemism:

- GSS unique – a species which is only found within the Gnangara Sustainability Strategy boundary
- local endemic – refers to a species which is only found within the Swan Coastal Plain SWA2 IBRA region
- regional endemic – refers to species which is only found within south-west Western Australia (see Chapter 2).

The terrestrial invertebrate fauna of south-west Western Australia also has both high species diversity and high levels of endemicity (Hopper *et al.* 1996), although few studies have been conducted on the invertebrates in the region. In forests of south-west Western

Australia, Abbott (1995) identified a total of 1747 described terrestrial insect species, from 235 different Families and 24 Orders. Invertebrate species richness estimates range from 12 000 to 25 000 species, with only an estimated 10% of invertebrates taxonomically described (Abbott 1995). Of insects, the butterfly taxa are typically well-known, and in south-west Western Australia ~ 60 species are expected to occur (Williams 2009).

Some of the fauna in south-west Western Australia have undergone a number of species extinctions and declines (Kitchener *et al.* 1978). The Western Australian *Wildlife Conservation Act 1950* lists fauna species under schedules based on their threatened status. Schedule 1 fauna are those that are likely to become extinct; Schedule 2 fauna are those that are presumed extinct; Schedule 3 fauna are birds that are protected under an international agreement and Schedule 4 fauna are those in need of species protection (and not covered under the other schedules). The Department of Environment and Conservation maintains a priority list which applies priority management to species that are poorly known and/or conservation dependent. In addition to these state-level classifications, the Commonwealth *Environmental Protection and Biodiversity Conservation Act 1999* (EPBC Act). Threatened fauna and flora may be listed in any one of the following categories under the EPBC Act: extinct, extinct in the wild, critically endangered, endangered, vulnerable and conservation dependent.

In addition to recognised threatened taxa, several species have undergone range contractions. The patterns of decline for specific taxa are explored in this chapter. The occurrence and distribution of fauna is often influenced by landscape features, including structural complexity and succession age, and are thus likely to respond to disturbance induced changes in habitat (e.g. Pianka 1989; Rosenzweig and Winakur 1969; Wilson *et al.* 1986). Reptiles are strongly dependent on habitat structure (Pianka 1989), typically having small home ranges, and are therefore often used as a surrogate measure of faunal diversity in response to disturbances (e.g. Cunningham *et al.* 2002; Valentine and Schwarzkopf 2009). Although there is scant information about the relative importance of habitat factors for herpetofauna assemblages, it is likely that the importance of disturbance factors varies between reptiles and amphibians (as well as among species of these two groups), and the life history attributes of a species will determine its response to a disturbance (e.g. How and Dell 2000; Jellinek *et al.* 2004). The high number of extinctions and range contractions by mammals in south-west Western Australia certainly indicates a susceptibility to human-mediated disturbances (Kitchener *et al.* 1978).

## Faunal diversity in the GSS Study Area

### *Invertebrates*

#### Species richness and endemicity

The assessment of the species richness and endemicity of invertebrates on the Gnangara groundwater system is limited as little is known of the taxonomy, species richness or endemicity of most invertebrate taxa on the Swan Coastal Plain. As part of the GSS–DEC biodiversity project, a review of the known biodiversity of invertebrate fauna for the Gnangara groundwater system was conducted and should be referred to for further information (Durrant 2009). This review highlights the paucity of major studies on invertebrates on the Swan Coastal Plain, and identifies large gaps in available databases. Furthermore, the outcomes of the review suggest that terrestrial invertebrate fauna is the most understudied taxa in the GSS study area. In this chapter, the focus is on terrestrial invertebrates, as wetland invertebrates are examined in Chapters 4 and 5.

A major biological study on the Swan Coastal Plain that surveyed terrestrial invertebrates was conducted by the Urban Bushland Study (Harvey *et al.* 1997; How *et al.* 1996). As part of this project, the invertebrate fauna of a number of urban bushland remnants was surveyed, with a diverse range of species identified, including 148 spiders, 13 other arachnids, 8 centipedes, 10 millipedes, 1 symphylan, 33 cockroaches and 25 baeine (parasitic) wasps. Of these taxa, several families and species had not been previously recorded on the coastal plain, with some species providing new records for Western Australia. Other studies on the plain have tended to focus on specific taxa, and these provide an indication of the potentially diverse invertebrate fauna of the GSS study area.

At least 21 native earthworms were recently recorded from the metropolitan region of the Swan Coastal Plain (Abbott and Wills 2002), of which 13 species were new records for the region. In addition, several species had restricted distribution on the plain, with both the Swan and Canning Rivers acting as physical barriers for several species. A survey of ground beetles (Carabidae) on the plain identified ~ 37 species (Guthrie 2001). Seventeen species were restricted to a single geological system, with most recorded only on either the Quindalup or Bassendean dunes. However, overall carabid richness tended to be higher towards the centre of the plain. In addition, terrestrial isopods are considered to be very diverse in south-west Western Australia, with the Swan Coastal Plain being one of the most species rich regions (Judd 2005).

A recent study examined butterfly and day-flying moth richness in urban reserves in the Swan Coastal Plain (Williams 2009). Of the 46 bushland remnants surveyed, 18 were located within the GSS study area. A total of 35 butterfly and 5 day-flying moth species were recorded, most of which were observed in at least one bush reserve in the GSS study area (33 butterfly and 4 day-flying moth species). One site in the GSS study area (Koondoola, Bush Forever site number 201) contained 27 of the 40 lepidotperan taxa, and represents a regionally important location for butterflies and moths. In a comprehensive survey of the ant fauna of the south-west Western Australian, Heterick (in prep.) recorded ~ 218 species of ants from the coastal plain. In terms of endemicity, Heterick (in prep.) suggests that ~ 7% of the taxa are locally endemic to the Swan Coastal Plain.

### Threatened taxa

There are three terrestrial invertebrate species that are listed as Schedule 1 fauna currently found in the GSS area. These are *Synemon gratiosa* (graceful sun moth) and two native colletid bees, *Leioproctus douglasiellus* and *Neopasiphae simplicior.* The graceful sun moth (listed as endangered under the EPBC Act) is a day-flying moth species that is threatened from clearing associated with urban development. This species is restricted to the Swan Coastal Plain, between Wanneroo in the north and Mandurah in the south. A recent survey identified six sites in the GSS study area where populations of this species occur (Williams 2009). The two native bees *Leioproctus douglasiellus* and *Neopasiphae simplicior* (listed as critically endangered under the EPBC Act) are threatened by clearing, draining of winter-wet depressions, fire and competition from introduced honeybees (Houston 1994). In addition, the wetland-restricted amphipod *Crangonyctid* sp. (crystal cave crangonyctid) is listed as Schedule 1 fauna and is unique to the GSS study area. Currently, there are three threatened ecological communities within the GSS study area that contain stygofauna communities that include Crangonyctids (see Chapter 2 and Chapter 4 for more information). As part of the GSS–DEC biodiversity project, reviews into the occurrences and status of these taxa were undertaken (Halse 2008; Tang and Knott 2008a; b).

## *Frogs*

### Species richness and endemicity

The frog fauna of the GSS study area is fairly well documented (Bamford and Huang 2009; Storr *et al.* 1978a; Tyler *et al.* 2000). The Western Australian Museum (1978) study remains the most comprehensive fauna survey of the Swan Coastal Plain, examining fauna occurrences between the Moore and Swan Rivers, a region largely overlapping with the GSS study area boundaries. Of the 13 species historically known to occur in the GSS study area (Storr *et al.* 1978a), all are considered extant (Bamford and Huang 2009). Regional endemicity of frog species is very high, with 12 of the 13 species considered regional endemics. However, only one species, *Crinia insignifera*, is locally endemic (restricted to the Swan Coastal Plain), and no species are unique to the GSS study area (Table 3.1).

Of the species that occur in the GSS study area, six are restricted to wetlands, including *Litoria moorei* (motorbike frog), *Litoria adelaidensis* (slender tree frog), *Crinia georgiana, Crinia pseudinsignifera*, *Crinia glauerti* and possibly *Pseudophryne guentheri* (Gunther's toadlet) (Table 3.1). Other species include the terrestrial *Myobatrachus gouldii* (turtle frog), *Limnodynastes dorsalis* (pobblebonk frog) and *Heleioporus eyrei* (moaning frog), which are dependent on upland woodland habitats during the non-breeding season (Bamford and Huang 2009). *Heleioporus barycragus* may only occur on the south-eastern edge, and the burrowing frog *Neobatrachus pelobatoides* was formerly widespread but is now only known from populations south of Perth (How and Dell 2000). Surveys conducted as part of the GSS–DEC biodiversity project recorded 9 of the 13 species calling during winter 2008 (Bamford and Huang 2009), although only 6 species were captured during pit-fall trapping (Valentine *et al.* 2009). On the Swan Coastal Plain, Storr *et al.* (1978a) reported that the eastern side had higher numbers of frog species than the western, a pattern that was confirmed by How and Dell (2000). However, the local patterns of distribution of frogs are generally unclear.

### Threatened taxa and patterns of decline

Although all species of frogs known to historically occur in the GSS study area are considered extant and none are listed as threatened taxa, it is likely that for a number of species there has been a widespread decrease in range, abundance, and loss of local populations. The large-scale loss and alteration of wetland habitats in the Swan Coastal Plain upon which several species depend, is likely to have had an impact on distribution and abundance. However, there are no data on historical changes in distribution or abundance and there has been little effort to assess recent changes (Bamford and Everard 2008; Bamford and Roberts 2003; Bancroft and Bamford 2008). The terrestrial breeding *Myobatrachus gouldii* (turtle frog) may have declined from clearing of the sandy *Banksia* woodland habitats that it occupied throughout the metropolitan region. Species such as *Limnodynastes dorsalis* (pobblebonk frog) and *Heleioporus eyrei* (moaning frog) depend on upland woodland habitats during the non-breeding season and may also have declined as a result of habitat loss.

In addition to habitat loss, Australia's native amphibians are threatened by a pathogenic fungus (*Batrachochytrium dendrobatidis*), which is known as the amphibian chytrid or amphibian chytrid fungus. This fungus causes the infection known as chytridiomycosis, an infection that affects amphibians worldwide. Infection of amphibians with the fungus is listed as a key threatening process under the EPBC Act *1999* (DEH 2006). Chytridiomycosis was the cause of extinction of one species of threatened frog and was suspected to have caused the extinction of three other species. The fungus has been found in 12 frog species in south-west Western Australia, and is expected to occur north of Perth (DEH 2006), in the GSS study area. Of the 12 species that have been recorded with chytridiomycosis, 8 species are known to occur in the GSS study area, including *Litoria moorei*, *Helioporus eyrie* and the locally endemic *Crinia insignifera* (Speare and Berger 2005).

Frog species are influenced by habitat factors and processes at various spatial scales (e.g. availability of suitable habitat, disturbance regimes, capacity for effective dispersal). However, they are affected primarily by changes in water cycling and quality, which can have a strong influence on an animal's water balance physiology and reproduction (White and Burgin 2004). Because of their biology, frogs are likely to be sensitive to changes in landscape hydrology. Frog species differ in their biology to the extent that they will almost certainly vary in their distribution across a region, with some wetlands being suitable for some frog species and not others, and will vary in their responses to environmental change. These characteristics potentially make frogs important indicators to changing hydrological conditions on the Gnangara groundwater system. Therefore, a study was undertaken by Bamford and Huang (2009) to assess the status of frogs in the GSS study area. This study is discussed in more detail in the GSS–DEC projects section below, and in the full technical report of the study (Bamford and Huang 2009).

## *Reptiles*

### Species richness and endemicity

The reptile fauna of the Gnangara groundwater system is diverse and abundant (How and Dell 1993; 1994; 2000). The Western Australian Museum reptile survey (Storr *et al.* 1978a) is one of the most comprehensive reptile surveys of the Swan Coastal Plain, and recorded 57 reptile species. Subsequent studies have focused on remnant vegetation patches, targeted specific taxa or have been conducted over short time frames (How 1998; How and Dell 1993; 1994; 2000). Key reptile studies (n = 23) conducted in the GSS study area have been reviewed by Huang (2009). Species richness is currently estimated at 64 reptile species recorded from the GSS study area, including 2 turtles, 8 geckos, 8 pygopods, 2 dragons, 3 goannas, 21 skinks, 4 blind snakes, 2 pythons and 14 elapid snakes (Table 3.1). All of these species are native to the area.

In terms of endemicity, nearly 50% (27 taxa) are considered regional endemics, being mostly restricted to south-west Western Australia (Table 3.1). Of these, seven taxa are local endemics to the Swan Coastal Plain, including *Pseudemydura umbrina*, *Delma concinna concinna*, *Pletholax gracilis gracilis*, *Hemiergis quadrilineata*, *Lerista christinae*, *Rankinia adelaidensis adelaidensis* and *Neelaps calonotos*. Only one species, the critically endangered western swamp tortoise (*Pseudemydura umbrina*), is unique to the GSS study area, with populations restricted to Ellen Brook Nature Reserve and Twin Swamps Nature Reserve on the eastern boundary of the GSS study area (Burbidge and Kuchling 2004). In addition, the GSS study area has 17 taxa at, or close to, their geographical limit. Several species (e.g. *Delma concinna concinna* and *Elapognathus coronatus*) are at the northern, southern or western limits of their distribution and only occur on the Swan Coastal Plain in low numbers.

### Threatened taxa and patterns of decline

The western swamp tortoise, *Pseudemydura umbrina,* is the only reptile species unique to the GSS study area and is Australia's most threatened reptile (Cogger 2000). This species is listed as Schedule 1 and is considered critically endangered under the EPBC Act. The western swamp tortoise has suffered a significant reduction in its range as a result of clearing for agriculture and housing development (Burbidge and Kuchling 2004). In addition, the carpet python *Morelia spilota imbricata* is considered Schedule 4 (in need of protection). The small elapid *Neelaps calonotos* is listed as a Priority 3 species and the skink *Ctenotus gemmula* is listed as Priority 4 (Table 3.1).

In comparison to the mammalian taxa, reptiles appear resilient to threatening processes, as evidenced by a lack of extinctions in Australia since 1788 (Cogger *et al.* 1993). Although the reptile communities of the GSS study area do not appear to have changed drastically in the past 180 years since European settlement, there has been a marked decline within the urbanised metropolitan Perth area (Storr *et al.* 1978a). Further, How and Dell (1994; 2000) recorded declines in most reptile species, both in abundance and local distribution across the bushland remnants of urban Perth, possibly indicating a time-lag in response to habitat loss for reptiles.

Several species have probably experienced contractions in their range and abundance since European settlement, with two species considered to be locally extinct on the GSS study area, including *Morelia stimsoni* (Stimson's python) and *Pseudonaja modesta* (ringed brown snake) (How and Dell 1994). In addition, the larger reptile predators (e.g. *Varanus* spp.(goannas) and *Morelia* spp. (pythons) appear to have been most dramatically affected, possibly due to predation by foxes and feral cats, combined with declines in their prey, which include large native mammals, other reptiles and frogs (How and Dell 1993).

The impacts of habitat loss and fragmentation, and fire on reptiles are discussed in Chapters 6 and 7. Several species, including the critically endangered *Pseudemydura umbrina* (western swamp tortoise), and the skink *Egernia luctuosa*, have been identified as species that are likely to respond unfavourably to the decline in wetland habitats (Bamford and Bamford 2003). These will be discussed in more detail in the next section.

Some taxa appear to have adapted to urbanisation. For example, *Pseudonoja affinis affinis* (dugite) are now common in urban bushland remnants and fringes of metropolitan areas, possibly due to the availability of prey such as the house mouse (How and Dell 1993). *Tiliqua rugosa* (bobtail) are also frequently observed in urbanised areas, possibly due to their ability to consume introduced plants (How and Dell 1993). *Christinus marmoratus* (marbled gecko), *Cryptoblepharus buchananii* (fence skink) and *Hemiergis quadrilineata* (two-toed earless skink) also appear to be prevalent in urbanised areas.

The distribution of reptile species across habitats is likely to be determined by species' life history attributes. Species that are adapted to open spaces, such as dragons, are likely to occur more frequently in open *Banksia* woodlands than in more closed tuart forest. Although regional biogeographical patterns have been recorded for reptiles on the Swan Coastal Plain, they mostly relate to assemblage differences between south and north of the Swan River and assemblages on the Darling escarpment (How and Dell 2000). Previous studies identified only marginal differences in the reptile assemblages across the landform units of the coastal plain – Quindalup, Spearwood, and Bassendean (How and Dell 2000; Storr *et al.* 1978a). These included higher gecko diversity to the west, which was considered to be related to the frequent limestone outcropping in the area (Storr *et al.* 1978a). Similarly, skink diversity was found to be lower in the eastern Bassendean dunes, compared to Quindalup and Spearwood dunes (Storr *et al.* 1978a) possibly due to a reduction in leaf litter in the more open *Banksia* woodlands.

The occurrence and distribution of reptiles are influenced by processes occurring at various spatial scales. Factors such as the availability of suitable habitat, disturbance regimes (e.g. fire regimes), predation and the capacity for effective dispersal also influence survival and therefore the occurrence and distribution of species (Cogger and Heatwole 1981; Fischer *et al.* 2004; Welsh *et al.* 2005). At a smaller spatial scale, the finer microhabitat level, vegetation condition and structural complexity also influence the distribution of many species (Brown 2001; Downey and Dickman 1993; Fischer *et al.* 2003; Greer 1997; Hutchinson 1993; Kanowski *et al.* 2006; Shine 1987). However, there is little information regarding the relative importance of habitat factors for herpetofauna assemblages in the GSS study area. To improve understanding of reptile diversity patterns, the GSS–DEC biodiversity project conducted fauna surveys throughout the contiguous remnant vegetation in 2007 and 2008. A summary of this project is provided in the GSS–DEC projects section below, or see the technical report for full details (Valentine *et al.* 2009).

Table 3.1: Reptile and frog species recorded on the Swan Coastal Plain and their endemicity within Australia. Taxa that are restricted to the Swan Coastal Plain are highlighted.

| Family | Species | Common name | Endemicity[1] |
|---|---|---|---|
| | ***Frogs: Order Anura*** | | |
| Hylidae | *Litoria adelaidensis* | slender tree frog | RE |
| | *Litoria moorei* | moore's frog; motorbike frog | RE |
| Myobatrachidae | *Crinia georgiana* | Tschudi's froglet | RE |
| | *Crinia glauerti* | Glauert's froglet | RE |
| | *Crinia insignifera* | sign-bearing froglet | LE |
| | *Heleioporus eyrei* | moaning frog | RE |
| | *Heleioporus albopunctatus* | western spotted frog | RE |
| | *Heleioporus psammophilus* | sand frog | RE |
| | *Heleioporus barycragus* | western marsh frog | RE |
| | *Limnodynastes dorsalis* | bullfrog | AUS |
| | *Myobatrachus gouldii* | turtle frog | RE |
| | *Pseudophryne guentheri* | Gunther's toadlet | RE |
| | *Neobatrachus pelobatoides* | humming frog | RE† |
| | ***Reptiles: Class Reptilia*** | | |
| ***Turtles and tortoises: Class Reptilia, Order Testudines*** | | | |
| Chelidae | *Chelodina oblonga* | oblong turtle | RE |
| | *Pseudemydura umbrina*[2] (CE, S1) | sestern swamp tortoise | GSS-unique |
| ***Lizards: Order Squamata, Suborder Sauria*** | | | |
| Gekkonidae | *Christinus marmoratus* | marbled gecko | AUS |
| | *Crenadactylus ocellatus ocellatus* | clawless gecko | RE |
| | *Diplodactylus alboguttatus* | white-spotted ground gecko | RE |
| | *Diplodactylus polyophthalmus* | speckled stone gecko | RE |
| | *Diplodactylus granariensis granariensis* | wheatbelt stone gecko | AUS |
| | *Strophurus spinigerus inornatus* | orange-eyed southwestern spiny-tailed gecko | RE |
| | *Strophurus spinigerus spinigerus* | southwestern spiny-tailed gecko | RE |
| | *Underwoodisaurus milii* | barking gecko | AUS |
| Pygopodidae | *Delma concinna concinna* | west coast javelin lizard | LE |
| | *Aprasia pulchella* | granite worm lizard | RE |
| | *Aprasia repens* | sandplain worm lizard | RE |
| | *Delma fraseri fraseri* | Fraser’s legless lizard | AUS |
| | *Delma grayii* | Gray’s legless lizard | RE |
| | *Lialis burtonis* | Burton’s legless lizard | AUS |
| | *Pletholax gracilis gracilis* | keeled legless lizard | LE |
| | *Pygopus lepidopodus* | common scaly foot | AUS |
| Scincidae | *Acritoscincus trilineatum* | south-western cool skink | AUS |
| | *Cryptoblepharus buchananii* | fence skink | AUS |
| | *Ctenotus australis* | west coast long-tailed ctenotus | RE |
| | *Ctenotus fallens* | west coast ctenotus | WA |
| | *Ctenotus gemmula* [2](P4) | jewelled ctenotus | RE |
| | *Ctenotus impar* | odd-striped skink | RE |
| | *Cyclodomorphus celatus* | coastal slender bluetongue | WA |
| | *Egernia kingii* | King’s skink | RE |
| | *Egernia luctuosa* | glossy swamp skink | RE |
| | *Egernia multiscutata* | bull-headed skink | AUS |
| | *Egernia napoleonis* | southwestern crevice skink | RE |
| | *Hemiergis quadrilieata* | two-toed earless skink | LE |
| | *Lerista christinae* | bold-striped four-toed lerista | LE |

| Family | Species | Common name | Endemicity[1] |
|---|---|---|---|
| | *Lerista elegans* | west coast four-toed lerista | WA |
| | *Lerista lineopunctulata* | line-spotted robust lerista | WA |
| | *Lerista praepedita* | west coast worm lerista | WA |
| | *Menetia greyii* | common dwarf skink | AUS |
| | *Morethia lineoocellata* | west coast pale-flecked morethia | WA |
| | *Morethia obscura* | southern pale-flecked morethia | AUS |
| | *Tiliqua occipitalis* | western bluetongue | AUS |
| | *Tiliqua rugosa rugosa* | bobtail | AUS |
| Agamidae | *Pogona minor minor* | western bearded dragon | AUS |
| | *Rankinia adelaidensis adelaidensis* | western heath dragon | LE |
| Varanidae | *Varanus gouldii* | Gould's monitor | AUS |
| | *Varanus rosenbergi* | southern heath monitor | AUS |
| | *Varanus tristis tristis* | black-tailed monitor | WA |
| ***Snakes: Suborder Serpentes*** | | | |
| Typhlopidae | *Ramphotyphlops australis* | southern blind snake | WA |
| | *Ramphotyphlops bituberculatus* | prong-snouted blind snake | AUS |
| | *Ramphotyphlops pinguis* | fat blind snake | RE |
| | *Ramphotyphlops waitii* | beaked blind snake | WA |
| Boidae | *Antaresia stimsoni stimsoni* | Stimson's python | WA |
| | *Morelia spilota imbricata* [2] (S4, P4) | carpet python | RE |
| Elapidae | *Brachyurophis fasciolata fasciolata* | narrow-banded shovel-nosed snake | WA |
| | *Brachyurophis semifasciata* | southern shovel-nosed snake | AUS |
| | *Demansia psammophis reticulata* | yellow-faced whip snake | RE |
| | *Echiopsis curta* | bardick | AUS |
| | *Elapognathus coronatus* | crowned snake | RE |
| | *Neelaps bimaculatus* | black-naped snake | AUS |
| | *Neelaps calonotos* [2] (P3) | black-striped snake | LE |
| | *Notechis scutatus* | tiger snake | AUS |
| | *Parasuta gouldii* | Gould's snake | RE |
| | *Parasuta nigriceps* | black-backed snake | AUS |
| | *Pseudechis australis* | mulga snake | AUS |
| | *Pseudonaja nuchalis* | gwardar | AUS |
| | *Pseudonaja affinis affinis* | dugite | WA |
| | *Simoselaps bertholdi* | Jan's banded snake | AUS |

[1] The endemicity within Australia (at the taxa level) for each species is provided: GSS-unique (restricted to GSS study area), LE (locally endemic to the Swan Coastal Plain), RE (regionally endemic to south-west Western Australia), WA (restricted to Western Australia), and, AUS (occurring within and outside Western Australia).

[2] Threatened species; threat status is in brackets (CE – Critically Endangered on IUCN red List and EPBC Act; S1 – Schedule 1 of WA Wildlife Conservation Act; S4 – Schedule 4 of WA Wildlife Conservation Act; P3 – Priority 3 fauna on DEC Priority List; P4 – Priority 4 fauna on DEC Priority List).

# *Birds*

## Species richness and endemicity

The term 'seabirds' collectively describes birds that spend a considerable part of their life foraging or breeding in the marine environment (e.g. gulls, terns and petrels). As seabirds are predominantly marine species, we do not include them in detailed discussions in this chapter. For our purposes, we have defined seabirds as being predominantly associated with a marine environment, although we recognise that some of these species will also utilise terrestrial habitats. Wetland bird species are categorised as being strongly associated with wetlands, freshwater lakes, or swamps and include some species that also utilise estuarine habitats or shores (e.g. curlew sandpiper, *Calidris ferruginea*).

The Western Australian Museum review of bird species (excluding seabirds) occurring in the GSS study area recorded 233 species (Storr *et al.* 1978b), including 7 introduced taxa. Of the native bird fauna, 100 species were considered residents, 102 species breeding or non-breeding visitors, and 14 were vagrants. More recently, a review of birds on the Swan Coastal Plain and adjacent seas (Storr and Johnstone 1988) considered 174 species (excluding sea birds) that are (or were) resident or regular visitors to the Perth region, including 34 species that annually migrate from the northern hemisphere. In addition, there are ~ 12 species that are either occasional visitors or have recently colonised the Swan Coastal Plain from other parts of Western Australia.

Records from Birds Australia Atlas surveys (1997–early 2008; see Appendix 8) indicate that there are records for a total of 217 bird species within the GSS study area (Birds Australia Atlas Database). Of these, 8 species are considered introduced (or domestic stock), with 209 species considered native to Australia, including some species that have been introduced to south-west Western Australia (e.g. rainbow lorikeets). The EPBC Act lists 83 species as being 'marine'. Of these, 35 species are also listed under the bilateral Japan–Australia Migratory Birds Agreement (JAMBA) and/or China–Australia Migratory Birds Agreement (CAMBA), which list terrestrial, water or shorebird species that migrate between Australian and Japan. Of the 83 species considered marine under the EPBC Act, only 34 species are considered seabirds (e.g. petrels, shearwater, gulls and terns) under our definition. Excluding seabirds, there are 163 species of native (non-seabird) birds recorded in the GSS study area since 1998. Of these, 63 species can be considered wetland birds, including several ducks, grebes, darters, cormorants, egrets, herons, ibis, rails, crakes and snipe. In addition to these species, there are historical records for at least 13 other species (see Table 3.2) that are now considered very rare or locally extinct in the GSS study area, including two wetland bird species, the black bittern (*Ixobrychus flavicollis*) and Australasian bittern (*Botaurus poicilloptilus*).

There are 13 bird taxa regionally endemic to south-west Western Australia. Recent records for at least 9 of these species (Birds Australia Atlas Database, 199 –early 2008) occur in the GSS study area, including Baudin's black-cockatoo (*Calyptorhynchus baudinii*), Carnaby's black-cockatoo (*Calyptorhynchus latirostris*), red-capped parrot (*Purpureicephalus spurius*), western rosella (*Platycercus icterotis*), western thornbill (*Acanthiza inornata*), western wattlebird (*Anthochaera chrysoptera lunulata*), western spinebill (*Acanthorhynchus superciliosus*) and the white-breasted robin (*Eopsaltria georgiana*). However, several of these species have undergone declines (see below) or are only observed infrequently. For example, Baudin's black-cockatoo is infrequently recorded in the GSS study area, and is mostly observed in the Perth Hills region, or south of the Swan River. In contrast, Carnaby's black-cockatoo is observed frequently in the GSS study area but is a declining species. There are no bird species locally endemic to the Swan Coastal Plain or the GSS study area.

### Threatened taxa and patterns of decline

Eighteen bird taxa in the GSS study area have declined and are considered threatened or priority taxa (Table 3.2). Five species are gazetted as Schedule 1, including the locally extinct or very rare Australasian bittern. Two are gazetted as Schedule 4, including the peregrine falcon. Of these, Carnaby's black-cockatoo is listed as endangered under the EPBC Act and three species are considered vulnerable (Table 3.2). In addition, there are nine priority listed species, including the freckled duck, little and black bittern and the painted snipe.

The West Australian government's habitat conservation program Bush Forever recognises species that are declining or have become locally extinct on the Swan Coastal Plain. Nearly 50% of the 71 naturally occurring passerines (perching birds) and 40% of the non-passerines have declined in abundance since European settlement (Government of Western Australia 2000). In total, nearly 80 species (excluding seabirds) are identified as having declined on  the Swan Coastal Plain and GSS study area (Table 3.2). Since the early 1900s, 17 species have been considered locally extinct on the coastal plain at some stage. In addition, 13 species have not been recorded from Birds Australia Atlas surveys since 1997, and can be considered locally extinct or very rare in the GSS study area (Table 3.2). These include species identified in the early 1900s as locally extinct (e.g. ground parrot, *Pezoporus wallicus*) and others that may still occur on the Swan Coastal Plain, south of the Swan River (e.g. southern emu-wren, *Stipiturus malachurus*).

Nearly all of the insectivorous and nectarivorous species have suffered a decline, most likely as a result of habitat clearing (Government of Western Australia 2000). Of the 70 species of naturally occurring passerines on the Swan Coastal Plain, How and Dell (1993) identified 46 as having decreased as a result of habitat loss and fragmentation, and only 8 as increasing. A number of studies have been conducted in the urban remnants in the Perth region. For examples, Recher and Serventy (1991) documented the local extinction of 9 species from Kings Park, and noted the decline of a further 14 species. Species that have become locally extinct in Kings Park are mostly insectivores, including the western yellow robin (*Eopsaltria griseogularis*), scarlet robin (*Petroica multicolour*), western thornbill (*Acanthiza inornata*) and golden whistler (*Pacycephala pectoralis*). Many species in the GSS study area are highly dispersive, especially nectar-dependent honeyeaters that follow flowering resources throughout the state (Davis 2009a). The connection of the *Banksia* woodlands in the GSS study area with the northern sandplains is likely to be important in ensuring annual bird movements are maintained. There is scant data on the broad-scale movement patterns of these species in Western Australia, but maintaining landscape-level connectivity is likely to be very important for these species.

Wetland birds have been particularly affected by large-scale loss of coastal plain wetlands (How and Dell 1993). Species such as black bittern (*Ixobrychus flavicollis*), Australasian bittern (*Botaurus poiciloptilus*) and whistling kite (*Haliastur sphenurus*) have suffered major declines. Indeed, both of the bittern species have not been recently observed in the GSS study area (Table 3.2). Wetland bird species such as freckled duck (*Stictonella naevosa*), Australasian shoveler (*Anas rhynchotis*) and hardhead (*Aythya australis*) have seriously declined (Government of Western Australia 2000). How and Dell (1993) also noted the decline of most raptor species due to the loss of coastal plain woodlands. The status of wetlands birds in the GSS study area is now the focus of a review (Bamford and Bamford Consultancy Pty Ltd). The status of terrestrial avifauna in the GSS study area, including threatening processes to specific species, and patterns of decline was reviewed by Davis (2009b). Factors that affect the abundance and diversity of bird populations in the GSS area include fire regime, dieback, clearing, fragmentation and degradation of habitat, wetland loss and drought. These are discussed in more detail in Chapters 6 to 10.

Table 3.2: Bird species (excluding seabirds) identified as having declined in distribution or population, or have become locally extinct in the GSS study area since the early 1900s. EX = locally extinct, DE = decline (either in distribution or population). Species in bold are those considered locally extinct at some stage on the Swan Coastal Plain. Species highlighted are those that have not been recorded in the GSS study area by Birds Australia Atlas database (1997–early 2008).

| Species | Common Name | Early 1900s[a] | 1978-1988[b] | 2000[c] |
|---|---|---|---|---|
| *Dromaius novaehollandiae* | emu | DE | DE | DE |
| *Stictonetta naevosa* | freckled duck*(P4) | | DE | DE |
| *Oxyura australis* | blue-billed duck | | | DE |
| *Biziura lobata* | musk duck | | DE | DE |
| *Anas rhynchotis* | Australasian shoveler | | DE | DE |
| *Malacorhynchus membranaceus* | pink-eared duck | | | DE |
| *Aythya australis* | hardhead | | | DE |
| *Nycticorax caledonicus* | nankeen night heron | | | DE |
| *Ixobrychus minutus* | little bittern*(P4) | | | DE |
| ***Ixobrychus flavicollis*** | **black bittern*(P2)** | | **DE/EX** | **DE** |
| *Botaurus poicilloptilus* | Australasian bittern*(S1) | | DE | DE |
| *Lophoictinia isura* | square-tailed kite*(P4) | | | DE |
| *Haliastur sphenurus* | whistling kite | | DE | DE |
| *Accipiter fasciatus* | brown goshawk | | | DE |
| *Accipiter cirrocephalus* | collared sparrowhawk | | | DE |
| *Hieraaetus morphnoides* | little eagle | | | DE |
| *Aquila audax* | wedge-tailed eagle | DE | DE | DE |
| *Circus approximans* | marsh harrier | | DE | |
| *Falco berigora* | brown falcon | | DE | DE |
| *Falco peregrinus* | peregrine falcon*(S4) | | | DE |
| *Gallinula tenebrosa* | dusky moorhen | | | DE |
| *Ardeotis australis* | Australian bustard | | | DE |
| *Turnix varia* | painted button-quail | | DE | DE |
| ***Turnix velox*** | **little button-quail** | **DE/EX** | | |
| ***Burhinus grallarius*** | **bush stone-curlew*(P4)** | **EX** | **DE/EX** | **EX** |
| ***Rostratula benghalensis*** | **painted snipe*(P3)** | | **EX** | **DE** |
| *Phaps chalcoptera* | common bronzewing | DE | DE | DE |
| ***Phaps elegans+*** | **brush bronzewing+** | **DE/EX** | **EX** | **DE** |
| ***Cacatua pastinator pastinator*** | **western long-billed corella*(V,S1)** | **EX** | **EX** | **EX** |
| ***Calyptorhynchus banksii naso*** | **forest red-tailed black cockatoo*(V,P3)** | **DE** | **EX** | **EX** |
| *Calyptorhynchus latirostris* | Carnaby's black-cockatoo *(E,S1) | DE | DE | DE |
| *Calyptorhynchus baudinii* | Baudin's black-cockatoo *(V,S1) | | | DE |
| ***Glossopsitta porphyrocephala*** | **purple-crowned lorikeet** | **DE/EX** | **EX** | |
| *Purpureicephalus spurious* | red-capped parrot | DE | | |
| *Platycercus icterotis* | western rosella | DE | | DE |
| *Neophema petrophila* | rock parrot | | | DE |
| ***Pezoporus wallicus+*** | **ground parrot+*(CE)** | **EX** | **EX** | |
| *Cuculus pallidus* | pallid cuckoo | DE | DE | |
| ***Ninnox connivens connivens*** | **barking owl*(P2)** | | **DE/EX** | **EX** |
| *Tyto novaehollandiae* | masked owl*(S4) | | | DE |
| *Todiramphus sactus* | sacred kingfisher | DE | DE | |
| *Climacteris rufa* | rufous treecreeper | | DE | DE |
| *Malurus elegans* | red-winged fairy-wren*(P1) | DE | DE | DE |

| Species | Common Name | Early 1900s[a] | 1978-1988[b] | 2000[c] |
|---|---|---|---|---|
| *Malurus splendens* | splendid fairy-wren | | | DE |
| *Malurus lamberti* | variegated fairy-wren | | | DE |
| *Malurus leucopterus* | white-winged fairy-wren | | | DE |
| *Stipiturus malachurus* | southern emu-wren | | DE | DE |
| *Sericornis frontalis* | white-browed scrub-wren | | | DE |
| *Smircornis brevirostris* | weebill | | | DE |
| *Acanthiza apicalis* | inland thornbill | DE | DE | DE |
| *Acanthiza inornata* | western thornbill | DE | DE | DE |
| *Acanthiza chrysorrhoa* | yellow-rumped thornbill | | | DE |
| *Melithreptus chloropsis* | white-naped honeyeater | DE | DE | DE |
| ***Melithreptus brevirostris*** | **brown-headed honeyeater** | **EX** | | |
| *Phylidonyris novaehollandiae* | New Holland honeyeater | | | DE |
| *Phylidonyris nigra* | white-cheeked honeyeater | | | DE |
| *Phylidonyris melanops* | tawny-crowned honeyeater | | | DE |
| *Acanthorhynchus superciliosus* | western spinebill | DE | DE | DE |
| ***Lichenostomus ornatus*** | **yellow-plumed honeyeater** | **DE/EX** | **DE** | **DE** |
| *Anthochaera lunulata* | western wattlebird | DE | | DE |
| *Manorina flavigula* | yellow-throated miner | DE | DE | DE |
| *Epthianura tricolor* | crimson chat | DE | | |
| *Epthianura albifrons* | white-fronted chat | DE | | |
| ***Petroica multicolor*** | **scarlet robin** | **DE** | **EX** | **DE** |
| *Melanodryas cucullata* | hooded robin | | | DE |
| *Eopsaltria griseogularis* | western yellow robin | DE | DE | DE |
| ***Eopsaltria Georgiana*** | **white-breasted robin** | | **DE/EX** | **DE** |
| ***Psophodes nigrogularis+*** | **western whipbird+*(S1)** | | **EX** | **EX** |
| *Daphoenositta chrysoptera* | varied sittella | DE | | DE |
| ***Falcunculus frontatus leucogaster*** | **western crested shrike-tit *(P4)** | | **EX** | **EX** |
| *Pachycephala pectoralis* | golden whistler | DE | DE | DE |
| *Colluricincla harmonica* | grey shrike-thrush | | | DE |
| *Myiagra inquieta* | restless flycatcher | | DE | DE |
| *Artamus cinereus* | black-faced woodswallow | DE | | DE |
| *Artamus cyanopterus* | dusky woodswallow | DE | | DE |
| *Cracticus torquatus* | grey butcherbird | DE | | |
| *Strepera versicolor* | grey currawong | DE | DE | DE |
| ***Stagonopleura oculata**** | **red-eared firetail*** | | **EX** | |
| *Cincloramphus cruralis* | brown songlark | DE | | |

[a] source: Alexander (1921)

[b] sources: Storr *et al* (1978b) and Storr and Johnstone (1988)

[c] source: Government of Western Australia (2000)

* Threatened species; threat status is in brackets (CE – Critically Endangered on EPBC Act; E – Endangered on EPBC Act; V – Vulnerable on EPBC Act; S1 – Schedule 1 of WA Wildlife Conservation Act; S4 – Schedule 4 of WA Wildlife Conservation Act; P3 – Priority 3 fauna on DEC Priority List; P4 – Priority 4 fauna on DEC Priority List).

+ Indicates that there are no definite records for this species in the GSS study area (Storr *et al.* 1978b)

## *Mammals*

### Species richness and endemicity

The Western Australian Museum survey in the late 1970s reviewed historical mammal species richness since settlement of the Swan River Colony in 1829. This review concluded that 33 native mammal species (including bat species), and 9 introduced species, have been recorded in the GSS study area from previous surveys (Kitchener *et al.* 1978)(Table 3.3). However, following a more recent comprehensive survey, only 12 of the original 33 species were recorded as persisting (Table 3.3), including 9 ground-dwelling mammals and 3 bat species (Kitchener *et al.* 1978). Reviews of subsequent surveys in the GSS study area indicate that of the 28 non-bat mammals historically recorded, 10 are still persisting in the GSS study area (Reaveley 2009) (Table 3.3). In addition, during the DEC–GSS fauna survey, a juvenile of an unidentified *Sminthopsis* species was captured, indicating that at least one *Sminthopsis* taxa is still extant in the GSS study area, albeit in small numbers.

There is a difference in the species composition derived from the Western Australian Museum records and that found subsequently in later surveys (Reaveley 2009). The Western Australian Museum recorded *Pseudomys albocinereus* but did not record *Cercartetus conncinnus* or *Dasyurus geoffroii*. The current mammal suite does not include *P. albocinereus*, as it has not been recorded since 1987, but as a result of recent observations, it now includes *C. concinnus* (R Davis, pers comm.: Lexia wetlands, 2006) and *D. geoffroii* (M Bamford, pers. comm.). These two species (and possibly still *P. albocinereus)* are predicted to only persist in isolated pockets of habitat in the GSS study area. The remaining 17 non-bat mammal species recorded prior to the museum surveys in the late 1970s (see Table 3.3) are considered unlikely to still occur in the GSS study area.

Five bat species have been recorded historically in the GSS study area (Table 3.3). However, the Western Australian Museum survey (Kitchener *et al.* 1978) identified only three species of bat in the GSS study area: *Nyctophilus geoffroyi*, *Vespadelus regulus* and *Chalinolobus gouldii*. Subsequent to Kitchener *et al*. (1978), few surveys have targeted bats and there have not been any systematic or comprehensive studies to assess their distribution, richness and abundance across the GSS study area.

Of the species historically recorded in the GSS study area, five taxa are regionally endemic to south-west Western Australia (Table 3.3). In addition, three taxa, *Bettongia penicillata ogilbyi, Dasyurus geoffroii* and *Myrmecobius fasciatus* are now considered restricted to south-west Western Australia as they have become extinct throughout the rest of their range. Of the 11 non-bat mammal taxa currently considered extant in the GSS study area, three species are considered regionally endemic (including the formerly widespread *D. geoffroii*).

### Threatened taxa and patterns of decline

Of the mammal species that have declined or disappeared from the GSS study area during the last century, 10 species are gazetted under the *Wildlife Conservation Act 1950* as Schedule 1 (including the extant *Dasyurus geoffroii*), and one species (*Onychogalea lunata*) as Schedule 2, presumed extinct (Table 3.3). Nine of the 11 species are also listed as vulnerable under the EPBC Act *1999* (Reaveley 2009). For the remaining species that currently persist within the GSS study area, several are considered priority fauna, including *Hydomys chrysogaster*, *Macropus irma*, and *Isoodon obesulus* (Table 3.3).

The record of extinction of mammals since Europeans arrived in Australia is significantly higher than elsewhere in the world (Burbidge and McKenzie 1989; Maxwell *et al.* 1996; Morton 1990; Short and Smith 1994; Smith and Quin 1996). In addition to these extinctions, the ranges of many Australian mammals have decreased significantly. A total of 27 mammals have become extinct with a further 39 being critically endangered or endangered (Environment Australia 2009). In Western Australia, 11 mammal species are considered extinct (Wildlife Conservation – Specially Protected Fauna Notice 2006). The dramatic decline of mammals is startlingly clear in the GSS study area, with a loss of 17 species following European settlement. In addition, a number of remaining species are now most likely restricted to isolated remnants. Surprisingly, few studies on mammal persistence have been conducted in the GSS study area. In comparison with reptiles, mammals are considered to lack resilience to threatening processes, particularly introduced predators, disease and changed fire regimes (Burbidge and McKenzie 1989; Maxwell *et al.* 1996; Morton 1990; Short and Smith 1994; Smith and Quin 1996). The impacts of these threatening processes on mammal fauna are further explored in Chapters 6 to 10, and in Reaveley (2009).

Table 3.3: Occurrence of mammal species in the GSS study area, endemicity within Australia and the broad periods of time in which they have been recorded. Taxa that are considered restricted to south-west Western Australia, including species once widespread, are in bold. Non-bat mammals that are considered to be extant in the GSS study area are highlighted.

| Species | Common Name | Endemicity[1] | 1800-1900 | 1900-1950 | 1950-1977 | 1977-2000 | 2000-2008 |
|---|---|---|---|---|---|---|---|
| *Macropus fuliginosus* | western grey kangaroo | AUS | ✓ | ✓ | ✓ | ✓ | ✓ |
| ***Macropus irma*** | **brush wallaby** * (P4) | **RE** | ✓ | ✓ | ✓ | ✓ | ✓ |
| *Macropus eugenii* | tammar wallaby | AUS | ✓ | ✓ | | ✓ | |
| *Petrogale lateralis lateralis* | black-footed rock-wallaby * (V, S1) | WA | ✓ | | | | |
| *Onychogalea lunata* | crescent nailtail wallaby * (E, S2) | AUS | ✓ | | ✓ | | |
| ***Bettongia penicillata ogilbyi*** | **woylie or brushtail bettong** * (S1) | **AUS** | ✓ | ✓ | | | |
| *Bettongia lesueur lesueur* | boodie or burrowing bettong * (V, S1) | AUS | ✓ | | ✓ | | |
| ***Setonix brachyurus*** | **quokka** * (V, S1) | **RE** | ✓ | ✓ | ✓ | | |
| *Lagostrophus fasciatus fasciatus* | banded hare-wallaby * (V, S1) | WA | ✓ | | | | |
| *Trichosurus vulpeca* | brush-tailed possum | AUS | | ✓ | ✓ | ✓ | ✓ |
| ***Pseudocheirus occidentalis*** | **western ringtail possum *(V,S1)** | **RE** | ✓ | ✓ | ✓ | | |
| *Cercartetus concinnus* | western pygmy possum | AUS | ✓ | ✓ | ✓ | | ✓ |
| ***Tarsipes rostratus*** | **honey possum** | **RE** | ✓ | ✓ | ✓ | ✓ | ✓ |
| *Isoodon obesulus fusciventor* | southern brown bandicoot or quenda * (P5) | WA | | ✓ | ✓ | ✓ | ✓ |
| *Macrotis lagotis* | greater bilby | AUS | ✓ | ✓ | | | |
| ***Dasyurus geoffroii*** | **chuditch or quoll *(V,S1)** | **AUS** | ✓ | ✓ | ✓ | ✓ | ✓ |
| *Antechinus flavipes leucaster* | mardo | WA | ✓ | ✓ | | | |
| ***Parantechinus apicalis*** | **dibbler*(V,S1)** | **RE** | | | ✓ # | | |
| *Phascogale tapoatafa* | brush-tailed phascogale *(S1) | AUS | ✓ | | | | |
| *Sminthopsis crassicaudata* | fat-tailed dunnart | AUS | | ✓ | ✓ | | |
| *Sminthopsis murina*† | common dunnart | AUS | ✓ | | | | |
| *Sminthopsis granulipes* | white-tailed dunnart | WA | | ✓ | | | |
| ***Myrmecobius fasciatus*** | **numbat*(V,S1)** | **AUS** | ✓ | ✓ | | | |
| *Rattus fuscipes* | southern bush rat | AUS | | | ✓ | ✓ | ✓ |
| *Rattus tunneyi* | Tunney's rat | AUS | | | ✓ ? | | |
| *Pseudomys albocinereus* | ash grey mouse | WA | ✓ | | ✓ | ✓ | |
| *Hydromys chrysogaster* | water rat or rakali *(P4) | AUS | | ✓ | ✓ | ✓ | |
| *Tachyglossus aculeatus* | echidna | AUS | | ✓ | ✓ | ✓ | ✓ |
| *Nyctophilus geoffroyi* | lesser long-eared bat | AUS | | ✓ | ✓ | ✓ | |
| *Vespadelus regulus* | southern forest bat | AUS | | ✓ | ✓ | ✓ | |
| *Chalinolobus gouldii* | Gould's wattle bat | AUS | ✓ | | ✓ | ✓ | ✓ |
| *Chalinolobus morio* | chocolate wattled bat | AUS | ✓ | | ✓ | | |
| *Tadarida australis* | white-striped freetail-bat | AUS | | ✓ | ✓ | | |
| *Canis lupus* | dingo | AUS | ✓ | ✓ | ✓ | | |
| *Vulpes vulpes* | red fox | I | | ✓ | ✓ | ✓ | ✓ |
| *Mustela putorius* | ferret | I | | | | ✓ | |
| *Felis catus* | feral cat | I | ✓ | | ✓ | ✓ | ✓ |
| *Rattus rattus* | black rat | I | | | | ✓ | ✓ |
| *Mus musculus* | domestic house mouse | I | | | | ✓ | ✓ |
| *Oryctolagus cuniuclus* | European rabbit | I | | ✓ | ✓ | ✓ | ✓ |

| Species | Common Name | Endemicity[1] | 1800-1900 | 1900-1950 | 1950-1977 | 1977-2000 | 2000-2008 |
|---|---|---|---|---|---|---|---|
| *Sus scrofa* | feral pig | I | | | | | ✓ |
| *Capra hircus* | goat | I | | | | | ✓ |

Sources: (Arnold *et al.* 1991; Bamford 1986; Bamford and Bamford 1990; 1994; 1999; Burbidge *et al.* 1996; DEC 1993; 2008; Drew 1998; Ecologia Environmental Consultants 1990; 1997; Friend 1996; How and Dell 2000; How *et al.* 1996; Kinhill Pty Ltd 1997; Kitchener *et al.* 1978).

[1] LE (locally endemic to the Swan Coastal Plain), RE (regionally endemic to south-west Western Australia), WA (restricted to Western Australia), AUS (occurring within and outside Western Australia), and I (Introduced – not endemic to Australia)

# cave deposit

† *Sminthopsis giseoventer* and *S. dolichura* were once considered part of the *S. murina* complex but are now considered separate species

? indicates uncertainty regarding species record

* threatened species; threat status is in brackets (CE – Critically Endangered on IUCN Red List and EPBC Act; S1 – Schedule 1 of WA Wildlife Conservation Act; S4 – Schedule 4 of WA Wildlife Conservation Act; P3 – Priority 3 fauna on DEC Priority List; P4 – Priority 4 fauna on DEC Priority List).

A number of surveys conducted near the GSS study area, but beyond its boundaries, have recent observations of species that are no longer considered extant in the GSS study area. A survey of Boonanarring Nature Reserve (15 km north of Gingin) in 1996 yielded a mammal suite of 11 species, which included *Pseudomys albocinereus* and *Sminthopsis griseoventer*, two ground-dwelling species that have not been recorded from the GSS study area for many years (Burbidge *et al.* 1996). A more recent survey in 2006 (R. Davis, pers comm.) in habitat near Ioppollo Road to the east of the GSS study area also recorded *P. albocinereus* and *S. griseoventer*. Two additional *Sminthopsis* taxa were recorded by Bamford (1986) from Mooliabeenee (private property), 10 km south-east of Boonanarring, during trapping surveys from 1983 to 1985. These species, *S. dolichura* and *S. granulipes*, historically occurred in the GSS study area, but have not been recorded for some time (Table 3.3).

These observations indicate that species originally extant in the GSS study area are still present close to its boundaries, and possibly indicate a preference of these species for transitional habitat from the coastal dunes to the Darling Scarp and Dandaragan Plateau. Furthermore, several species in the GSS study area, such as *Rattus fuscipes*, *Hydromys chrysogaster* and *Macropus irma*, only occur in restricted or isolated populations (How and Dell 2000; Kitchener *et al.* 1978). The mammal fauna of the GSS study area have experienced both high local extinction rates, and all remaining species are considered to have declined in distribution and abundance (Kitchener *et al.* 1978).

The reasons for mammal decline are many and varied. Their decline can be attributed to no single factor, and it is more likely that each species has been affected differently by various combinations of factors, some of which commenced before the arrival of the first European settlers (Abbott 2008). However, following European colonisation, habitat loss and fragmentation from land clearing, altered fire regimes, predation by foxes and cats, habitat modification resulting from declines in wetland, rainfall and groundwater levels, and *Phytophthora* dieback, disease and introduced herbivores have significantly contributed to their decline (Reaveley 2009). The threatening processes affecting mammals in the northern Swan Coastal Plain are discussed in more detail in Reaveley (2009), and in Chapters 6 to 10.

## Threats to terrestrial fauna in the GSS study area

A number of threatening processes have been implicated in the extinctions and declines of vertebrate fauna in the GSS study area. These include habitat clearance and fragmentation as a result of agriculture and urbanisation (How and Dell 2000; Kitchener *et al.* 1980), changed fire regimes (Calver and Dell 1998; How *et al.* 1987) and predation by introduced foxes and cats (Algar and Smith 1998; Christensen 1980; Christensen and Burrows 1994; Hayward *et al.* 2003; Kinnear *et al.* 2002; Morris *et al.* 1998; Short *et al.* 2002). The impact of wildlife disease has also been implicated in the decline of mammal species in Western Australia (Abbott 2008)). More recently, climate change (declining rainfall and groundwater levels), and the impacts of the plant pathogen *Phytophthora cinnamomi* on flora, fauna and ecosystems have been identified (Froend *et al.* 2004a; Froend *et al.* 2004b; Froend *et al.* 2004c; Government of Western Australia 2009; Groom *et al.* 2000a; Groom *et al.* 2001; Groom *et al.* 2000b; Horwitz *et al.* 2008; How and Dell 1993; Pettit *et al.* 2007). Importantly, individual threatening processes may have synergistic effects, resulting in compounded impacts. Reviews of these threats and assessments of their impacts in GSS study area are discussed further in Chapters 6 to 10. In this section we focus specifically on identifying taxa threatened by declining groundwater levels.

# Declining groundwater levels

One of the main processes threatening biodiversity identified by the GSS is declining groundwater levels due to climate change, decreased rainfall and increasing groundwater abstraction. In the GSS study area, the dependence of fauna upon groundwater is largely indirect, with the exception of primarily aquatic species that occur in wetlands (e.g. freshwater fish or frogs) (Bamford and Bamford 2003). Fauna indirectly dependent on groundwater are mainly dependent upon vegetation that may be groundwater dependent (Bamford and Bamford 2003). Therefore, declining groundwater levels and reduced rainfall have the potential to affect all fauna within the GSS study area to some degree. However, the impacts can be expected to occur most rapidly where habitats are most sensitive to changes in groundwater levels, such as wetlands and damplands.

As part of an investigation by the Waters and Rivers Commission and the Water Corporation on the environmental effects of groundwater abstraction, Bamford and Bamford (2003) assessed the importance of wetlands and phreatophylic vegetation for vertebrates in the GSS study area. In addition, this study identified species that are likely to be sensitive to reductions or change in groundwater levels. Bamford and Bamford (2003) classified fauna into one of four categories depending on their perceived reliance on wetland habitats:

- low dependence upon groundwater; changes to groundwater levels unlikely to alter species directly
- moderate dependence upon groundwater; changes to groundwater levels likely to affect fauna depending on how the vegetation assemblages they occur in respond
- high dependence upon groundwater; changes to groundwater likely to cause a reduction in habitat area, and associated fauna species would therefore decline in abundance or disappear
- very high dependence upon groundwater; species that rely on aquatic habitats in wetlands and that are therefore likely to become locally extinct if a fall in the groundwater level leads to the disappearance of surface water in wetlands.

Of the taxa we have identified as occurring in the GSS study area, Bamford and Bamford (2003) suggest 88 species are likely to decline in abundance or experience range contractions due to their high to very high dependence upon groundwater (Table 3.4). Of these, 68 species were considered to have a very high dependence upon groundwater, including five frogs, two reptiles, 60 birds and one mammal.

Most of the frog species occurring in the GSS study area are considered sensitive or very sensitive to declining groundwater levels, with some species (e.g. *Heleioporus eyrei*) likely to become locally extinct with reduction in groundwater levels (Bamford and Bamford 2003). The majority of reptile species in the GSS study area are expected to occur in upland habitats and are therefore only likely to be indirectly susceptible to declining groundwater levels via subsequent changes to vegetation type or flow-on effects (e.g. altered fire regimes). However, both turtles that occur in the GSS study area (including the endangered western swamp tortoise) are threatened by declining groundwater.

Not surprisingly, the bird fauna included several taxa that are very reliant on wetlands and wetland habitats (Table 3.4). Wetland birds (e.g. bitterns, ducks) and birds associated with wetland habitat (e.g. sacred kingfisher) are likely to be affected by declining groundwater. There is only one semi-aquatic mammal species in south-west Western Australia, the priority-listed *Hydromys chrysogaster* (rakali). This species is classified as highly dependent upon groundwater levels as it is largely restricted to permanent wetlands. Other mammal species that rely on the dense vegetation associated with wetlands include the priority listed *Isoodon obesulus* (quenda).

In addition to species that are likely to be directly affected by declining groundwater, there are a number of species that are likely to be indirectly affected. Species such as the honey possum are not considered to have a high reliance upon groundwater levels as they mostly occur in upland *Banksia* woodlands. However, declining groundwater levels and reduced rainfall rates are predicted to alter vegetation communities (Chapter 10) and changes in vegetation cover, floristics composition and productivity may be expected. Changes to vegetation cover and composition may result in altered microhabitats (e.g. thermal properties) and resource availability (e.g. food resources, protection from predators) for fauna, and well as potentially modified fire regimes. Consequently, declining groundwater levels may indirectly affect all fauna in the GSS study area. Further work is required to identify how vegetation will respond to declining groundwater and reduced rainfall so that potential flow-on effects can be predicted.

Table 3.4. Vertebrate taxa predicted to decline in abundance or range due to a high or very high dependence on groundwater levels. Adapted from Bamford and Bamford (2003).

| **Taxa** | **Species** | **Common name** |
|---|---|---|
| | **High dependence upon groundwater** | |
| Frogs | *Heleioporus eyrei* | moaning frog |
| | *Heleioporus psammophilus* | marbled frog |
| | *Limnodynastes dorsalis* | pobblebonk |
| | *Pseudophryne guentheri* | Guenther's toadlet |
| Reptiles | *Acritoscincus trilineatum* | |
| | *Elapognathus coronatus* | crowned snake |
| | *Notechis scutatus* | tiger snake |
| Birds | *Acrocephalus stentoreus* | clamorous reed-warbler |
| | *Epthianura albifrons* | white-fronted chat |
| | *Malurus elegans* | red-winged fairy-wren |
| | *Megalurus gramineus* | little grassbird |
| | *Ninox connivens connivens* | barking owl |
| | *Stagonopleura oculata* | red-eared firetail |
| | *Threskiornis molucca* | Australian white ibis |
| | *Threskiornis spinicollis* | straw-necked ibis |
| | *Todiramphus sanctus* | sacred kingfisher |
| | *Vanellus tricolor* | banded lapwing |
| Mammals | *Isoodon obesulus* | southern brown bandicoot or quenda |
| | *Macropus irma* | brush wallaby |
| | *Rattus fuscipes* | bush rat or moodit |
| | **Very high dependence upon groundwater** | |
| Frogs | *Crinia georgiana* | quacking frog |
| | *Crinia glauerti* | Glauert's froglet |
| | *Crinia insignifera* | sandplain froglet |
| | *Litoria adelaidensis* | slender tree frog |
| | *Litoria moorei* | motorbike frog |
| Reptiles | *Chelodina oblonga* | long-necked tortoise |
| | *Pseudemydura umbrina* | short-necked tortoise |
| Birds | *Anas castanea* | chestnut teal |
| | *Anas gibberifrons* | grey teal |
| | *Anas platyrhynchos* | mallard |
| | *Anas rhynchotis* | Australasian shoveler |
| | *Anas superciliosus* | Pacific black duck |
| | *Ardea alba* | great egret |
| | *Ardea ibis* | cattle egret |
| | *Ardea pacifica* | white-necked heron |
| | *Aythya australis* | hardhead |
| | *Biziura lobata* | musk duck |
| | *Botaurus poiciloptilus* | Australasian bittern |
| | *Calidris acuminata* | sharp-tailed sandpiper |
| | *Calidris ferruginea* | curlew sandpiper |
| | *Calidris ruficollis* | red-necked stint |
| | *Charadrius ruficapillus* | red-capped plover |
| | *Chenonetta jubata* | Australian wood duck |
| | *Chlidonias hybrida* | whiskered tern |
| | *Circus approximans* | swamp harrier |
| | *Cladorhynchus leucocephalus* | banded stilt |

| Taxa | Species | Common name |
|---|---|---|
| | *Cygnus atratus* | black swan |
| | *Dendrocygna eytoni* | plumed whistling-duck |
| | *Egretta garzetta* | little egret |
| | *Egretta novaehollandiae* | white-faced heron |
| | *Elseyornis melanops* | black-fronted dotterel |
| | *Erythrogonys cinctus* | red-kneed dotterel |
| | *Fulica atra* | Eurasian coot |
| | *Gallinula tenebrosa* | dusky moorhen |
| | *Gallinula ventralis* | black-tailed native-hen |
| | *Gallirallus philippensis* | buff-banded rail |
| | *Haliaeetus leucogaster* | white-bellied sea-eagle |
| | *Himantopus himantopus* | black-winged stilt |
| | *Ixobrychus flavicollis* | black bittern |
| | *Ixobrychus minutus* | little bittern |
| | *Limosa lapponica* | bar-tailed godwit |
| | *Malacorhynchus membranaceus* | pink-eared duck |
| | *Nycticorax caledonicus* | nankeen night heron |
| | *Oxyura australis* | blue-billed duck |
| | *Pandion haliaetus* | osprey |
| | *Phalacrocorax carbo* | great cormorant |
| | *Phalacrocorax melanoleucos* | little pied cormorant |
| | *Phalacrocorax sulcirostris* | little black cormorant |
| | *Phalacrocorax varius* | pied cormorant |
| | *Platalea flavipes* | yellow-billed spoonbill |
| | *Platelea regia* | royal spoonbill |
| | *Plegadis falcinellus* | glossy ibis |
| | *Podiceps cristatus* | great crested grebe |
| | *Poliocephalus poliocephalus* | hoary-headed grebe |
| | *Porphyrio porphyrio* | purple swamphen |
| | *Porzana fluminea* | Australian spotted crake |
| | *Porzana pusilla* | Baillon's crake |
| | *Porzana tabuensis* | spotless crake |
| | *Recurvirostra novaehollandiae* | red-necked avocet |
| | *Rostratula benghalensis australis* | Australian painted snipe |
| | *Stictonetta naevosa* | freckled duck |
| | *Tachybaptus novaehollandiae* | Australasian grebe |
| | *Tadorna tadornoides* | Australian shelduck |
| | *Tringa glareola* | wood sandpiper |
| | *Tringa hypoleucos* | common sandpiper |
| | *Tringa nebularia* | common greenshank |
| | *Tringa stagnatalis* | marsh sandpiper |
| Mammals | *Hydromys chrysogaster* | rakali or water rat |

# Summary of DEC–GSS projects (2007–09)

## Project 1. Patterns of ground-dwelling vertebrate biodiversity in the GSS study area

Technical report reference: Valentine L. E., Wilson B. A., Reaveley A., Huang N., Johnson B. & Brown P. H. (2009). *Patterns of ground-dwelling vertebrate biodiversity in the Gnangara Sustainability Strategy study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

The aim of the GSS ground-dwelling vertebrate fauna survey was to assess the occurrence and distribution of terrestrial vertebrate fauna in the relatively undisturbed bushland in the northern half of the GSS study area (40 sites, using pitfall traps and Elliott and Sheffield traps), and to examine patterns in biodiversity according to landform and vegetation type. Trapping was conducted in spring 2007, autumn 2008 and spring 2008.

The fauna survey results indicate a diverse and rich assemblage of fauna within the GSS study area. In total, 38 reptile, 16 mammal and 6 frog species were trapped and/or recorded during surveys. Of the 64 reptile species expected to occur in the GSS region (Huang 2009), the fauna surveys recorded nearly 60% of these species. Total species richness varied among sites, ranging from 2 to 18 species, with the most species recorded at two Bassendean sites in long unburnt *Banksia* woodland (> 20 years since fire). The lowest species richness was recorded at a Spearwood site in *Melaleuca* dampland with young fuel age (three years since fire).

Skinks were the most commonly caught taxon during surveys (65%), including the widespread or commonly occurring *Lerista elegans*, *Cryptoblepharus buchananii* and *Menetia greyii*. The most widely distributed and abundant frog species were *Limnodynastes dorsalis* (pobblebonk) and *Heleioporus eyrei* (moaning frog). In general, mammal capture rates were low, with the exception of the introduced *Mus musculus* (house mouse) and the native *Tarsipes rostratus* (honey possum) trapped at 30% of sites. In addition, one *Rattus fuscipes* (bush rat) and three *Tachyglossuss aculeatus* (echidna) were captured during the general fauna survey. Honey possums occurred only on Bassendean and Spearwood soils and were mostly associated with *Banksia* and *Melaleuca* vegetation types.

Habitat structure and composition strongly influence faunal assemblage, as differences in these attributes invariably lead to differences in the resources available for fauna. Reptiles in particular depend on habitat structure for their survival (Pianka 1989). Of the three factors examined (landform, vegetation type and fuel category), vegetation type influenced the greatest number of response variables, including reptile abundance, species richness, diversity, evenness, plant species number and a number of individual taxa. Mean reptile abundance was similar between both landform units (Spearwood and Bassendean dune systems), but varied between vegetation types. The greatest abundance of reptiles was observed within tuart-dominated vegetation (Figure 3.1). Landform types did not influence patterns in species composition, and where differences in abundance between landform types were detected, they were generally reflecting a preference for a specific habitat type.

Reptile species richness was greatest in *Banksia* woodland sites (Figure 3.1), perhaps due to the floristic diversity of *Banksia* woodland reflecting a structural diversity that provides a variety of habitat resources for a range of reptile species. Although only two sites in the coastal scrub vegetation were surveyed (and hence this vegetation type was omitted from analysis), both mean abundance and species richness of reptiles in coastal scrub appears high (Figure 3.1). A number of individual species responded to vegetation type, with several species, including the frog *Heleioporus eyrei*, and the skinks *Hemiergis quadrilineata*, *Lerista elegans*, *Menetia greyii* and *Morethia obscura* observed in higher abundances in tuart vegetation (Figure 3.1). In contrast, the heath dragon *Rankinia adelaidensis* was most abundant in *Banksia* woodlands.

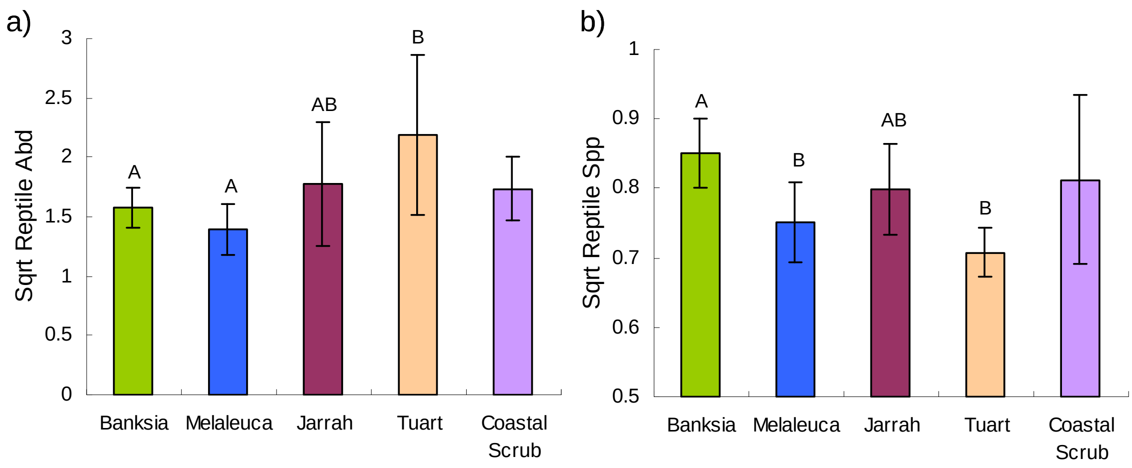


Figure 3.1: Mean (± 95%CI) a) reptile abundance and b) species richness in relation to vegetation type. Letters above error bars indicate significant differences of means between vegetation types (Tukey HSD, $\alpha < 0.05$). Coastal scrub vegetation type was not included in the statistical comparisons.

# Project 2. Targeted trapping of quenda and rakali in the GSS study area

Technical report reference: Valentine L. E., Wilson B. A., Reaveley A., Huang N., Johnson B. & Brown P. H. (2009). *Patterns of ground-dwelling vertebrate biodiversity in the Gnangara Sustainability Strategy study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

In May 2008 a targeted survey was undertaken to locate populations of *Isoodon obesulus* (quenda or southern brown bandicoot) and *Hydromys chrygsogaster* (rakali or water rat) within the GSS study area. Quenda were not recorded during the GSS terrestrial vertebrate survey (Valentine *et al.* 2009) and yet they are one of the few medium-sized native mammals known to persist in the study area. Quenda were also targeted as they are associated with wetland-associated vegetation and may be susceptible to declining groundwater (Bamford and Bamford 1994).

Rakali were targeted to determine if any persisted within the GSS study area, and if so, their distribution. As the only known aquatic carnivorous mammal of inland waterways in south-west Western Australia, its presence in a wetland or waterway could be a useful indicator of the ecological integrity of wetland systems. As permanent water bodies were believed to be suitable rakali habitat (Olsen in Van Dyck and Strahan 2008), three lakes were selected for trapping: Lake Joondalup in the suburb of Joondalup, Lake Goolellal in the suburb of Kingsley and Lake Loch McNess in Yanchep National Park.

Quenda were recorded from 5 of the 9 selected survey sites (Figure 3.2). Most quenda were recorded at Twin Swamps Nature Reserve, the only site fenced and baited against introduced foxes. The combined results of the targeted field survey in May 2008, a 2008 community survey, and recent data from Whiteman Park and Ellenbrook Nature Reserve (Reaveley 2009), indicate that quenda are widespread in their distribution from west to east across the GSS study area. The association of quenda with wetland-associated vegetation is possibly linked to the refuge that this habitat provides from fox predation (Bamford and Bamford 1994). In our study, quenda were observed only in high densities at the fenced and baited Twin Swamps Nature Reserve, indicating that fox predation has suppressed quenda populations in unbaited habitat. The persistence of quenda in unbaited areas on the Swan Coastal Plain relies on dense wetland-associated vegetation.

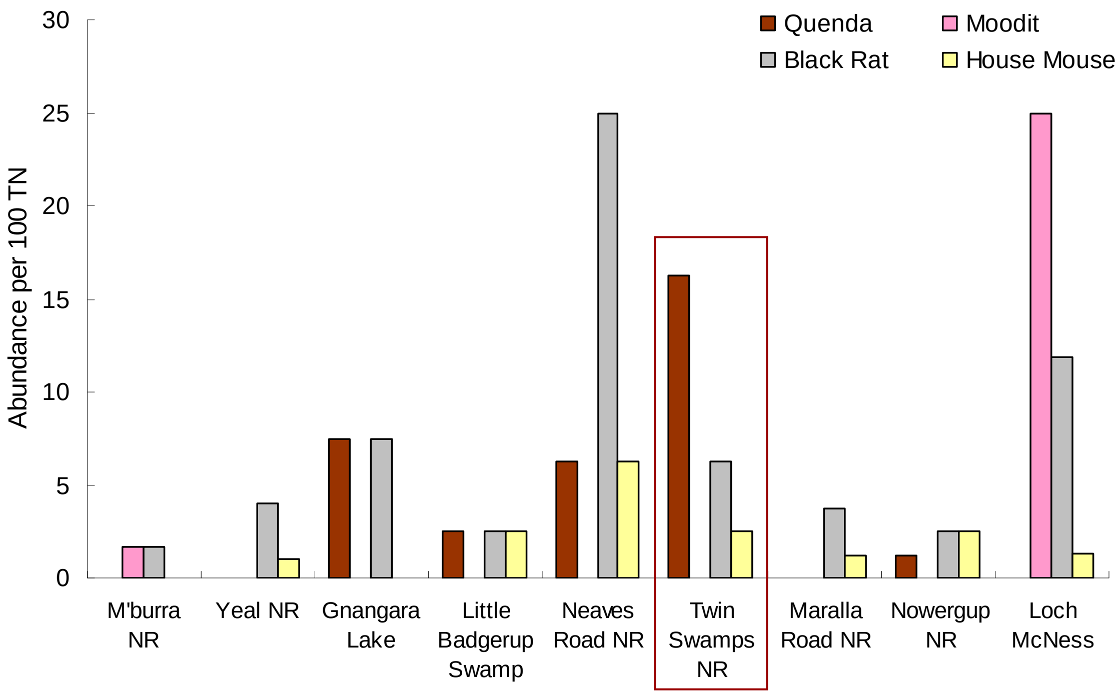


Figure 3.2: Abundance (per 100 trap nights) of four mammal species at sites targeted for trapping quenda. The abundance of quenda was highest at Twin Swamps Nature Reserve, which is fenced and baited to reduce predators. (M’burra NR = Muckenburra Nature Reserve).

Rakali were captured at all three of the selected sites. Although Lake Goolellal had the least survey effort, this site recorded the highest abundance of rakali (6 individuals from 30 trap nights), indicating that a sizable population may reside at this lake. The abundance of rakali recorded at Lake Goolellal and Loch McNess with minimal survey effort was surprising, indicating that these two lakes support reasonable populations of this species. Typically, rakali tend to be easily captured in areas where they are numerous (McNally 1960). Further surveys may establish that rakali persist in most of the permanent wetlands within the GSS study area. The survival of rakali is critically linked to the persistence of wetland ecosystems. This species will require careful management as climate change and decline in groundwater levels impairs the viability of wetlands.

## Project 3. Assessing the occurrence and status of frogs in the GSS study area

Technical report reference: Bamford M. & Huang N. (2009) *Status and occurrence of frog species in the Gnangara Sustainability Strategy study area*. Unpublished report prepared by M.J. and A.R. Bamford Consulting Ecologists and the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

The frog fauna of the GSS study area is well documented (Storr *et al.* 1978a; Tyler *et al.* 2000), with 13 species expected to occur. However, local patterns of distribution are not well understood. The focus of the study conducted by Bamford and Huang (2009) was to assess the status of frogs in the GSS study area (based on field surveys), and to review the biology and current distribution of these frogs. The study also reviewed the biology of each species, to investigate its potential for predicting sensitivity to hydrological change.

Frog species were broadly classified as follows:

- Robust in the face of hydrological decline and sensitive only to almost catastrophic change, due to their reliance on permanent or near-permanent wetlands. Such wetlands tend to be large systems with considerable capacity for contraction. Species in this category include *C. georgiana, L. dorsalis, L. adelaidensis* and *L. moorei*.
- Robust in the face of hydrological decline due to longevity and persistence even in the face of failed breeding in successive years. Longevity effectively masks impacts so that other facets of the wetland ecosystem could be profoundly affected before a change

would be detected in the frog species' population. Species in this category include *H. eyrei*.

- Sensitive to hydrological change, with populations likely to decline rapidly due to reliance on small, shallow wetlands and near-annual recruitment. Species in this category include *C. glauerti* and *C. insignifera*. C*. glauerti).*
- Very sensitive to hydrological change due to a specific and inflexible breeding biology that relies on early winter rains and very shallow wetlands. Species in this category include *P. guentheri).*

Species reliant on permanent or near-permanent wetlands (*C. georgiana*, *L. adelaidensis, L. dorsalis* and *L. moorei*) may already have suffered some local contractions in range. However, the wetlands on which they depend are likely to maintain some water levels other than in the event of catastrophic hydrological change. Species that rely on seasonal wetlands (*C. glauerti*, *C. insignifera, H. eyrei* and *P. guentheri*) are most likely to react to slight or moderate hydrological changes, as slight changes may lead to the rapid drying of the seasonal wetland. In the face of declining rainfall and groundwater levels, *H. eyrei* will probably persist for many years, which could mask effects on its breeding success. *C. insignifera* and particularly *C. glauerti* would be more sensitive to change and their presence could indicate wetland systems that have not yet been adversely affected. They could also be expected to disappear within a few years due to hydrological declines and the reliance of their populations on annual or near-annual recruitment. *P. guentheri* may already have declined across much of the GSS area and appears to be especially sensitive to hydrological declines (Bamford and Huang 2009).

Nine species were recorded during the Bamford and Huang (2009) surveys. Wetland type, the presence of sedges and rushes and the presence of surface water in winter strongly influenced the occurrence of species at sites (Figure 3.3). Most species occurred more frequently at lakes or watercourses and at sites where surface water was present in winter, or sites with sedges. *C. georgiana* occurred only along watercourses that were permanent or at least contained water for long periods, whereas *P. guentheri* occurred widely on sumplands, including sites with no surface water or sedges and rushes . The two froglets, *C. glauerti* and *C. insignifera*, were both found to be widespread. *C. glauerti* was recorded only at sites with surface water in winter and with sedges and rushes. *L. dorsalis* and *L.*

*adelaidensis* both showed a preference for wetlands in the Spearwood system, typically in wetlands with permanent or near-permanent water and abundant sedges and rushes.

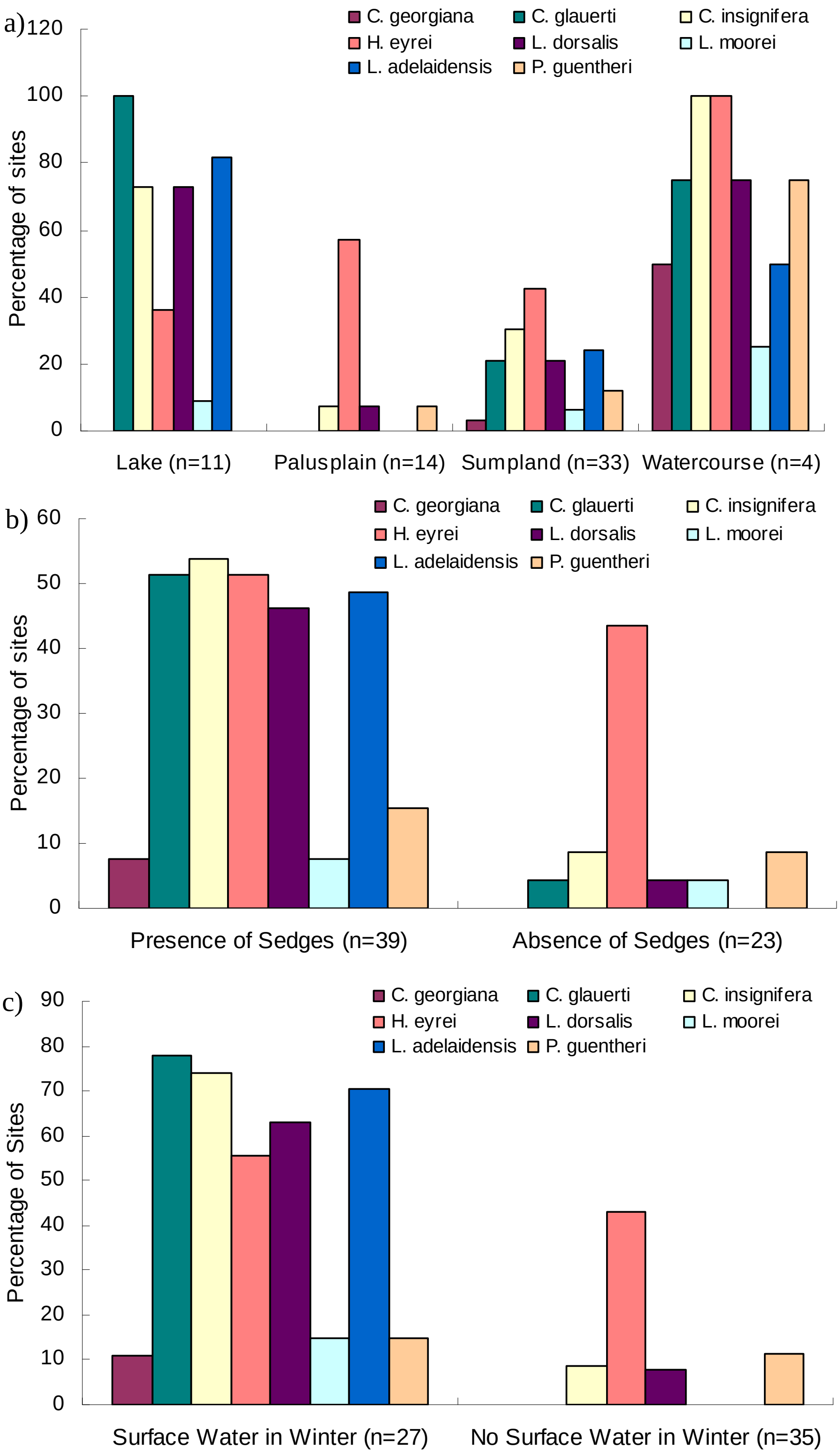


Figure 3.3. Occurrence of each species within a) wetland type, b) sites with and without sedges or rushes, and c) sites with and without surface water in winter.

# Discussion

The aims of this chapter were to collate and review information available on terrestrial fauna (reptiles, frogs, mammals, birds and invertebrates) in the GSS study area, address knowledge gaps in the occurrence, distribution and conservation status of fauna in the GSS study area, and identify threatening processes.

The review confirmed that the GSS study area historically had high species richness, with at least 13 frogs, 64 reptiles, 176 species of birds (excluding seabirds) and 33 native species of mammals recorded in the area. Several taxa, particularly amongst the frogs and reptiles are endemic to south-west Western Australia and the Swan Coastal Plain. One reptile, the critically endangered western swamp tortoise (*Pseudemydura umbrina*), which is restricted to the GSS study area, has undergone a significant reduction in range due to habitat loss. A major limitation encountered in this study was access to existing databases for several fauna taxa. Data were often held in disparate databases and in many cases did not contain spatial information – hindering database comparisons. Furthermore, this review highlighted the lack of invertebrate fauna data for the GSS study area.

The GSS study area also contains a number of threatened or priority taxa. There are three threatened terrestrial invertebrate species currently recorded in the GSS study area. These are *Synemon gratiosa* (graceful sun moth) and two native bees, *Leioproctus douglasiellus* and *Neopasiphae simplicior*. The GSS study area is particularly important for the graceful sun moth, with six populations of this species recently recorded (Williams 2009). The critically endangered western swamp tortoise, *Pseudemydura umbrina*, is restricted to the GSS study area and is Australia's most threatened reptile taxon. In addition, the carpet python *Morelia spilota imbricata*) is gazetted as Schedule 4 (in need of special protection) and there are two other priority-listed reptiles. The GSS study area has records for 18 threatened and priority bird taxa, 8 of which have not been recorded since before 1997, and can be considered either locally extinct or very rare. The GSS study area provides vital feeding habitat for the endangered Carnaby's black-cockatoo (*Calyptorhynchus latirostris*), which feed in the extensive areas of *Banksia* woodland as well as in the pine plantations during the non-breeding season. Of the 10 non-bat mammals considered extant on the GSS, four are listed as threatened or priority taxa, including *Dasyurus geoffroii* (chuditch), *Macropus irma* (brush wallaby), *Isoodon obesulus fusciventor* (quenda) and *Hydromys chrysogaster* (rakali).

The patterns of decline of terrestrial fauna in the GSS study area were reviewed in this chapter and indicate that the GSS study area has suffered substantial loss of species, particular in the mammal fauna. Although all species of frogs known to occur on the Swan Coastal Plain are considered extant, it is likely that there has been a widespread decrease in range, abundance, and local populations of some frogs, due to threatening processes such as declining rainfall and groundwater levels. Most reptile species have declined in abundance and local distribution across the urban bushland remnants, and two species are considered locally extinct.

Since European settlement, nearly 80 species of birds have been identified as having declined in abundance or range, including 13 species now considered locally extinct or very rare. Wetland birds have been particularly affected by the large-scale loss of wetlands on the Swan Coastal Plain, and two species of bitterns, the black bittern (*Ixobrychus flavicollis*) and Australasian bittern (*Botaurus poicilloptilus*), were not recorded by Birds Australia surveys since before 1997. The mammal taxa in south-west Western Australia have undergone a large number of species extinctions and declines, due to numerous threatening processes, and extant mammal fauna are mostly restricted to remnant vegetation patches.

This chapter identified several threatening processes that affect terrestrial fauna in the GSS study area, most of which are discussed in detail in Chapters 6 to 10. Dependence on groundwater and sensitivity to declining groundwater levels is a significant threat to at least 88 fauna in the GSS study area (not including fish). Declining groundwater levels will have a direct impact on primarily aquatic species and those strongly associated with wetland or dampland habitats. Additional terrestrial fauna species are also likely to be affected, albeit in a more indirect manner, as a result of declining groundwater levels affecting the vegetation upon which these species rely.

At the start of the DEC–GSS project, spatial models were not available to evaluate biodiversity values and a range of land management scenarios at a landscape scale. We have addressed some of these issues by examining some species habitat preferences and requirements (e.g. see Valentine *et al.* 2009) and by identifying taxa within the GSS that may be susceptible to declining groundwater and other threats (see Chapters 6 to 10). Further work on specific taxa is still required.

Analyses of data and field assessments have advanced our understanding of the terrestrial fauna biodiversity values in the GSS study area. During a vertebrate fauna survey that was undertaken in 2007–08 to assess the occurrence and distribution of ground-dwelling vertebrate fauna in the GSS study area, skinks were the most commonly caught group of species, and the highest number were recorded at two Bassendean sites in *Banksia* woodland with old fuel ages (> 20 years since fire). Mammal capture rates were very low, with the exception of the introduced the native honey possum and the introduced house mouse. Targeted surveys for the quenda (or southern brown bandicoot) and rakali (or water rat) within the GSS study area recorded the highest numbers of quenda at the only site that is fenced and baited against introduced predators. Rakali were captured at all three of the lakes that were surveyed within the study area. A survey conducted to assess the status of frogs in the GSS study area recorded 9 of the known 13 frog species in the GSS study area. Frog species that are reliant on permanent or near-permanent wetlands may have already suffered some local contractions in range, while frog species that rely on seasonal wetlands will most likely be affected by slight or moderate hydrological changes.

## Recommendations

In order to compile information on fauna diversity that will assist with monitoring changes in biodiversity due to declining groundwater levels, it is recommended that:

- the fauna monitoring sites that were established in the GSS study area for this project should be used for future fauna monitoring
- further research be carried out to determine the taxonomic diversity and endemicity of invertebrate fauna in the GSS study area
- further research be carried out to establish wetland bird persistence in the GSS study area
- further research be carried out to establish the persistence of key mammal species, including quenda, rakali and honey possums, in the GSS study area
- strategic monitoring of taxa identified as being susceptible to declining groundwater levels commence.

In order to conserve and manage populations of target species (including threatened and priority-listed species), it is recommended that:

- potential refugia sites for target species be identified
- expert advice be sought on possible mitigation methods for threatened taxa.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter Four: Wetlands and Groundwater-Dependent Ecosystems

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Our environment, our future

September 2009

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Department of Water
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# 4. WETLANDS AND GROUNDWATER-DEPENDENT ECOSYSTEMS

## Key points

- There are many definitions of what constitutes a ‘wetland’, including non-tidal surface water systems, other groundwater-dependent ecosystems, communities and components such as caves, stygobiota, phreatophytic vegetation, and areas of groundwater discharge including coastal areas.
- Trends in ground and surface water levels experienced at wetlands of differing geomorphology, soil type and hydrology show short- and longer-term patterns. Hydrograph analyses of 12 wetlands on the Gnangara groundwater system representing different landform units, management regimes and urban influences demonstrate the hydrological variation found in average conditions in recent years (over 5 years).
- The main wetland sediment types on the Swan Coastal Plain are peat, diatomite and calcilutite (carbonate mud). These wetland sediments accumulate in depressions under conditions of permanent, seasonal or intermittent saturation, and are then further subjected to biological, chemical and physical processes in situ.
- Wetland vegetation of the Swan Coastal Plain has been commonly categorised at two levels: the uppermost level or ‘complex’ refers to vegetation units linked by dominant plant species and structural attributes, and the secondary level for classification or ‘community’ is based on common or typical species within the overall complex. Wetlands on the GSS study area feature a variety of flowering plants that are highly disjunct or at the ends of their (frequently mesic) ranges.
- Rare and endemic invertebrate taxa are generally associated with rare wetland types such as cave streams and mound springs which provide a unique wetland environment with characteristic stygofaunal assemblages distinguishable from the unconfined aquifer.
- Sampling of Yanchep caves and Ellen Brook Valley Springs identified a number of newly identified species of stygofauna, including three crustacean species apparently endemic to the GSS study area: one amphipod and two copepod species.

- There is a relative deficiency of invertebrate data from wetlands on the north-eastern portion of the Gnangara groundwater system and from wetlands immediately north of the Moore River–Gingin boundary.
- Wetlands on the Gnangara groundwater system are generally species rich but have low levels of local endemism. For example, the species richness of aquatic invertebrate taxa is surprisingly high.
- At least four mammal species (honey possum, rakali, quenda and chuditch), two reptile species (western swamp tortoise and oblong tortoise), two frog species (the quacking frog and Glauert's froglet), and two fish species (black striped minnow and mud minnow) are significant from a wetland perspective. Approximately 70 wetland birds make use of wetlands in the GSS study area.
- Seven threatened ecological communities are considered to be either wetland communities (occurring in seasonally or permanently inundated areas) or groundwater dependant whereby elements of the community are reliant on access to the groundwater table.

# Introduction

This chapter has a specific focus on the inland, freshwater ecosystems of the Gnangara groundwater system, principally those ecosystems with surface water, an interaction between surface water and groundwater, or a dependence upon groundwater. The focus is necessarily non-coastal and non-tidal since these processes are managed separately from surface and groundwater (notwithstanding the fact that groundwater and surface water flows can have ecological implications for nearshore, estuarine and tidal systems). The seasonal rainfall patterns, temperature regimes, landforms and soil types of the area examined by the Gnangara Sustainability Strategy create relationships of dependence between the biota and water that are bioregionally idiosyncratic,.

The aim of this chapter is to review wetlands and other groundwater-dependent ecosystems on the Gnangara groundwater system, as bounded by the GSS study area. In addressing this aim we provide:

- a description of relevant aspects of wetland hydrology

- a review of databases and literature on wetlands and their biotic and abiotic components, ecosystem services provided by wetlands, and landscape–waterscape associations and contributions
- an assessment of rare and priority flora, fauna and threatened ecological communities associated with wetland ecosystems, and their reservation and protection status.

In some cases general information is presented for the Swan Coastal Plain bioregion as a whole. Information for the review has come predominantly from published literature; unpublished sources have been used where appropriate and relevant. In particular, the chapter draws heavily on key sources where information has already been synthesised, for instance Cargill and van Etten (2008) for wetland vegetation, Froend and Loomes (2008) for wetland hydrology, CA and V Semeniuk (publications over the period 1987–2006), Horwitz *et al.* (in press) for invertebrates, and Bamford and Bamford (2003) and others for vertebrates.

This chapter should be read in conjunction with Chapter 5, which deals with the changes that have occurred to wetlands over the last 200 years, wetland loss, and processes that continue to change the ecological values of wetlands.

## Wetland definitions and values

We regard wetlands as including:

- non-tidal surface water systems
- other groundwater-dependent ecosystems, communities and components such as caves, stygobiota, and phreatophytic vegetation
- areas of groundwater discharge including coastal areas.

More specifically we recognise Semeniuk's (1987) categorisation of surface water wetlands based on water permanency and cross-sectional shape of the wetland (Table 4.1).

Table 4.1: Semeniuk's (1987) classification of Swan Coastal Plain wetlands (x = does not exist).

| Cross-sectional shape | Permanent | Seasonally inundated | Seasonally waterlogged |
|---|---|---|---|
| Basin | Lake | Sumpland | Dampland |
| Gully | River | Stream | x |
| Flat | x | Floodplain | Palusplain |

To fully incorporate groundwater-dependent systems we need to go beyond these surface water categorisations. Horwitz *et al*. (2008) categorise hydrology–biology linkages to demonstrate those biotic components (from individuals, populations, communities and ecosystems) that have a hydrological requirement for their continued survival. Their scheme is based on the principle that linkage between hydrology and biology is a question of degree (similar to Hatton and Evans' (1998) principal criterion that the degree of 'groundwater dependence' is proportional to the fraction of the annual water budget that an ecosystem derives from groundwater). Their eight types are separated primarily along an axis of water permanence, but secondarily according to the nature of the linkage, related to life history characteristics of organisms and biogeochemical requirements. All eight types are found on the Gnangara groundwater system (Table 4.2). See Chapter 5, Table 5.3 for a treatment of how these linkages might be affected by groundwater decline.

These eight types therefore build on the categories of groundwater dependence defined by Semeniuk (1987), Hatton and Evans (1998) and others (e.g. sumpland, floodplain, dampland, palusplain, aquifer and cave systems, terrestrial vegetation and river base flow systems).

By 'ecological values' we mean those associated with biodiversity (the variety of life) and geodiversity (the natural diversity of geological, landform and soil features, and processes), and what emerges from the interactions between them. Values can be attributed to: (i) biotic and abiotic components, including intrinsic values, biodiversity and/or geodiversity contribution, any instrumental, Indigenous and non-Indigenous cultural values); (ii) wetland functions (i.e. 'ecosystem services' perceived as valuable for human well-being, such as the provision of water); and (iii) those related to the role wetlands play in the landscape and waterscape (Table 4.3). Documentation of these wetland values is not dissimilar to the emerging schemes for gauging 'ecological character' of wetlands as recognised by the Ramsar Convention on Wetlands (see DEWHA 2008).

Table 4.2: Hydrology–biology linkages to demonstrate those biotic components (from individuals, populations, communities and ecosystems) that have a hydrological requirement for their continued survival (adapted from Horwitz *et al*. (2008).

| **Hydrology-biology linkage** | **Relevance to the Gnangara groundwater system** |
|---|---|
| Requirement for seasonal soil moisture | All vegetation, to different degrees |
| Requirement for seasonally moist habitat for aestivation/drought avoidance | Burrowing crayfish (*Cherax quinquecarinatus*), burrowing frogs (*Heleioporus eyrei*) or aestivating fish (*Galaxiella nigrostriata*) |
| Requirement for a seasonal or intermittent surface saturation | Microcrustacean faunas with drought resistant eggs. Waterbirds that feed on them (e.g. palusplain wetlands on eastern part of the GSS study area associated with Ellen Brook). |
| Terrestrial requirement (obligate and facultative) for access to groundwater table | Phreatophytic vegetation (e.g. *Banksia* woodlands or tuart (*Eucalyptus gomphocephalus*)) |
| Requirement for groundwater discharge to maintain a particular quality of surface water or habitat. | Tumulus mound springs: groundwater discharge enables build up of organic material, creating well vegetated, well shaded habitats for other organisms |
| Requirement for permanent surface or subsurface saturation | Any lake with permanent water (e.g. Loch McNess, Yonderup, Goolellal Lakes). Permanent subsurface saturation required to prevent acidification. |
| Requirement for an exchange between surface/subsurface flows and groundwater | Baseflows in Ellen Brook and Gingin Brook. Flow through lakes such as Lake Jandabup, Lake Nowergup. |
| Requirement for saturated hypogean (interstitial) spaces (aquifer) | Stygofauna in spaces between sand grains. Fauna dependent on saturation of tuart root mats in caves. |

In the above treatment of ecological values we have not sought to distinguish between human and non-human aspects of wetland and groundwater-dependent ecosystem values because ultimately values reside with humans, not in nature. Specialised and more detailed treatments of cultural appreciations of biodiversity and geodiversity are required in addition to this review. For instance, for Indigenous values:

> There are two schools of thought held by Nyungar people regarding the spiritual significance of wetlands. One holds that all sources of freshwater are sacred because the Waugal created them; the other holds that only specific places where the Waugal resides are sacred … While there is no readily available explanation for divergent views, it is recognised that both schools of thought may be legitimate in the Nyungar context (McDonald *et al.* 2005).

Similarly divergent views of wetlands are found within other cultures:

> Results … confirm that people's connection to places on the [Gnangara Groundwater System] is related to behavioural and emotional bonds, and that some groups display stronger bonds than others. This suggests that intrinsic values associated with places on the Gnangara are worthy of consideration. Various localities on the [Gnangara Groundwater System] are considered inspirational places, historically important and iconic to Perth, and aesthetically pleasing. Further, many people feel a strong emotional connection with these places. In terms of use value, respondents feel that 'places on the [Gnangara Groundwater System] provide recreational experiences that are second to none' and are good places for families to get together (Tapsuwan *et al.* 2009).

This chapter then works towards the documentation of examples of where values of wetlands and GDEs, are held within the Gnangara groundwater system. As values are determined by different forms of agreement, it is probably best to subject the features regarded here as 'significant' to a process of dialogue with local and regional stakeholders. The constituents in the final table (Table 4.3), therefore, are less absolute than they appear.

Table 4.3: Broad wetland and groundwater-dependent ecosystem value categories, with criteria used to elucidate them.

| Wetland value category | Wetland value criteria |
|---|---|
| Biotic and abiotic values – where components of aquatic ecosystems are regarded as special | Intrinsic value |
| | Biodiversity and geodiversity contributions |
| | Instrumental |
| | Indigenous and non-Indigenous cultural values |
| Wetland function values – ecosystem services that wetlands provide | Maintenance of key ecological processes |
| | Maintaining and influencing water quality |
| | Provision of water resource |
| Landscape and waterscape values – wetlands as part of larger scaled systems | Connectivity |
| | Representativeness |
| | Uniqueness |
| | Habitat values |
| | Waterscape ecology, ecological integrity and resilience |
| | Values recognised by Indigenous peoples, sense of place, and other cultural values |

# Water cycle and groundwater interface

Ultimately, biophysical characteristics of wetlands and groundwater-dependent depend on their water regime, the domain of the study of hydrology.

## Hydrological cycles of permanently or seasonally inundated wetlands of the Gnangara groundwater system

The shallow, unconfined aquifers of the south-west respond to seasonal rainfall, groundwater levels rising during winter to a spring peak and decreasing during summer to an autumn minimum. Groundwater recharge occurs when soil field capacity is exceeded and water moves below the root zone under gravity (Commander and Hauck 2005). This generally takes place only during periods of intense, prolonged rainfall.

Wetlands usually occur in depressions (see below). The majority of permanently inundated wetlands, or lakes, are shallow expressions of the groundwater table which fill following winter rainfall and dry during summer as groundwater levels fall, rainfall decreases and surface evaporation increases. Sumplands, holding water only during winter and spring, and damplands, seasonally waterlogged areas, can form as expressions of underlying groundwater. Floodplains hold water when streams such as Ellen Brook, Gingin Brook or Quin Brook overflow in winter or spring, and palusplains become saturated from seasonally elevated groundwater levels. Sumplands and palusplains can also form from rainfall perching over more impermeable soils (like diatomaceous earths which have increased water holding capacities).

Trends in ground and surface water levels experienced at wetlands of differing geomorphology, soil type and hydrology show short-term and longer term patterns. To demonstrate the hydrological variation found in average conditions in recent years (2001–02 to 2007–08) the hydrograph analyses of 12 Gnangara groundwater system wetlands are described here. Wetlands have been chosen to represent different landform units, management regimes and urban influences.

Hydrological parameters include maximum and minimum seasonal (August–July) surface water levels (m AHD), seasonal variation (m), month of minimum and maximum surface water levels, number of months taken each year for water levels to decline from maximum to minimum (or for a wetland to dry at the staff gauge) and number of months wetlands were inundated at the staff gauge each year (Table 4.4). These parameters are also presented for a 21-year period (1985–86 to 2007–08) for a subset of wetlands representing different geomorphologic units, soil types and hydrology, enabling description of historic water level trends and comparisons with recent trends (Figure 4.1 to Figure 4.6 and Table 4.5).

## *Current hydrological regimes*

The current (2001–02 to 2007–08) hydrological regimes of the 12 wetlands indicate that the greatest mean seasonal variation in surface water levels is experienced by Lake Gwelup (1.30 m) and Big Carine Swamp (0.89 m), despite the fact that both of these wetlands are only inundated at the staff gauge for an average of 8.4 and 8.8 months per year respectively (Table 4.4). It is probable that these large seasonal variations reflect their urban nature, a use of these relatively small wetlands as local storm water drainage basins, where even low inflow would have an impact on surface water levels as well as the catchment hydrology and hydrogeology of these systems.

Lake Yonderup and Loch McNess in comparison are permanently inundated Spearwood interdunal wetlands and experience only a very small seasonal variation (0.05 m and 0.17 m respectively). In addition, they differ from the other permanent wetlands with respect to the timing of minimum and maximum water levels, with maximums generally recorded 1 to 2 months earlier and minimums 1 to 5 months earlier. These differences reflect the unique hydrology of Lake Yonderup and Loch McNess, influenced at least in part by the high transmissivity aquifer in Tamala limestone (Davidson 1995), close to the coast where watertables are relatively flat. The permanently inundated Lake Goollelal experiences seasonal variations of 0.74 m.

The relatively small seasonal water level variations recorded at Lake Mariginiup of 0.31 m and a wetland Lexia 86 of 0.32 m reflect the fact that both dry at the staff gauge each year and are only inundated for 5.4 and 2.4 months respectively. Similarly, Lake Gnangara, inundated for 8.8 months per year, experiences a mean seasonal variation of 0.65 m, drying by March during most years.

The permanently inundated Lake Jandabup and Lake Nowergup experience average seasonal variations of 0.60 m and 0.57 m respectively, and are both artificially maintained. Lake Jandabup should respond like lakes Mariginiup and Gnangara (being part of the same wetland suite), but the influence of artificial maintenance can be seen by the gradual reduction in seasonal variation over the last 10 years, from around 1 m per annum to around 0.5 m per annum (Figure 4.3). Similar reductions in seasonal variation would be noted at Nowergup; in addition the ongoing pump operations at Lake Nowergup may offer a possible explanation for early minimums (January) in surface water levels (Table 4.4).

Table 4.4: Hydrological variables of 12 Gnangara groundwater system wetlands over a five year period 2001–02 to 2007–08. Wetlands are arranged according to their distance from the coast (as expressed by their elevation in metres AHD).

| **Wetland** | **Mean max seasonal surface water level m AHD**[1] | **Mean min seasonal surface water level m AHD** | **Mean seasonal water level change m** | **SE**[2] **m** | **Month (max)** | **Month (min)** | **Mean number of months max–min (or to dry)** | **Mean number of months inundated at staff gauge** | **Generalised geomorphology (Davidson 1995)** |
|---|---|---|---|---|---|---|---|---|---|
| Big Carine Swamp | 3.89 | 3.00 | 0.89 | 0.28 | Oct | April | 5.11 | 8.8 | Low level marsh in depression Spearwood |
| Lake Gwelup | 6.42 | 5.12 | 1.30 | 0.16 | Sept | Feb | 5.89 | 8.4 | Low level marsh in depression Spearwood |
| Loch McNess | 7.03 | 6.86 | 0.17 | 0.04 | Aug | Feb | 6.86 | 12 | Interbarrier depression Spearwood |
| Lake Yonderup | 5.94 | 5.88 | 0.05 | 0.01 | Aug | Feb | 6.71 | 12 | Interbarrier depression Spearwood |
| Lake Nowergup | 16.75 | 16.18 | 0.57 | 0.14 | Sept | Jan | 6.57 | 12 | Interbarrier depression Spearwood |
| Lake Joondalup | 16.93 | 16.09 | 0.85 | 0.04 | Oct | April | 5.86 | 12 | Interbarrier depression Spearwood |
| Lake Goollelal | 27.41 | 26.66 | 0.74 | 0.06 | Sept | Feb | 6.71 | 12 | Interbarrier depression Spearwood |
| Lake Mariginiup | 41.60 | 41.28 | 0.31 | 0.07 | Sept | Jan | 4.14 | 5.4 | High level marsh in depression Bassendean/ Spearwood |
| Lake Jandabup | 44.83 | 44.23 | 0.60 | 0.05 | Sept | Mar | 5.86 | 12 | High level marsh in depression Bassendean |
| Lake Gnangara | 42.12 | 41.47 | 0.65 | 0.03 | Sept | Feb | 5.57 | 8.8 | High level marsh in depression Bassendean |
| Lexia 86 | 48.36 | 48.04 | 0.32 | 0.15 | Sept | Nov | 3.29 | 2.4 | High level marsh in depression Bassendean |
| EPP 173 | 51.05 | 50.41 | 0.64 | 0.03 | Sept | Mar | 5.57 | 8 | High level marsh in depression Bassendean |

[1] m AHD = elevation with reference to Australian Height Datum
[2] SE = standard error

## *Extended hydrological regimes*

### Big Carine Swamp

Big Carine Swamp was permanently inundated up to 1995–96. Although it too has dried each season since that time, surface water is retained longer within the basin than at other seasonal wetlands. Maximum water levels have occurred between August and November (Table 4.5). However, the wetland has progressively dried earlier at the staff gauge and has generally dried quicker over time. Although there has been a coinciding decline in surface water levels, seasonal variation has increased (Table 4.1). This may be a result of increased storm water runoff.

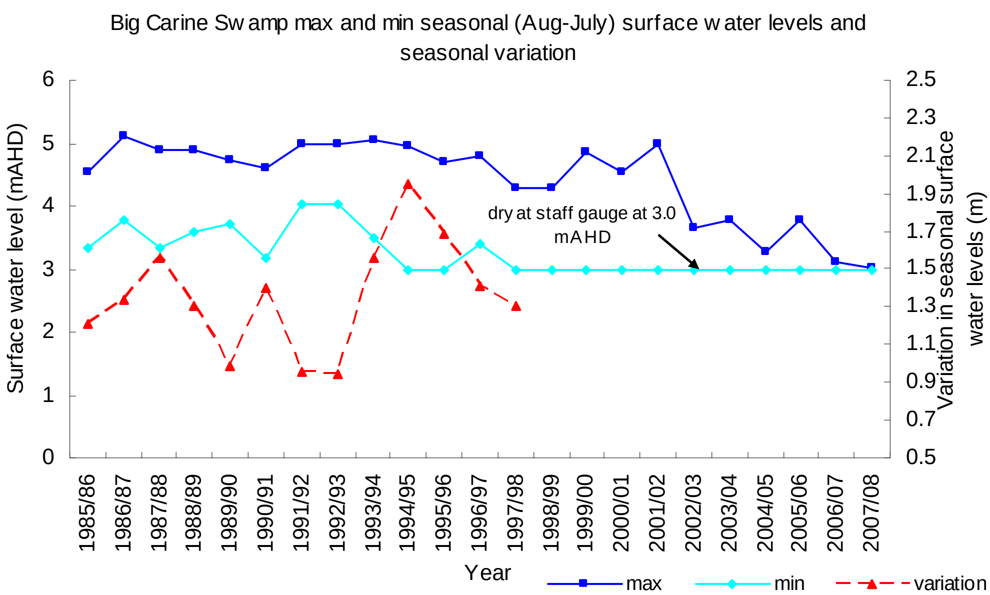


Figure 4.1: Maximum, minimum and seasonal (July-August) variations in surface water levels at the Big Carine Swamp staff gauge over a 19 year period 1985–06 to 2007–08.

### Loch McNess

Loch McNess has also been permanently inundated over the past 20 years and experienced a slow long-term decline in maximum and minimum surface water levels (Figure 4.2). Unlike other permanent wetlands, with the exception of Lake Yonderup, the decline in maxima has been less than 0.1 m and minima less than 0.2 m over that 21-year period. However, there has been a marked decline in 2007 and 2008 with the lowest ever recorded value at Loch McNess south in the 21 years of monitoring (6.631 m AHD). Spring peak water levels (6.937 m AHD) were also registering the lowest level since recording commenced in 1973. Given that seasonal fluctuations for this wetland have been minimal in the past, such a dramatic drop suggests the possibility of a threshold change in water levels, possibly like that experienced at other karstic systems like Coogee Springs or in the Yanchep Caves.

Loch McNess has reached maximum water levels between June and September and minimum water levels between January and March, earlier in the year than other wetlands. The wetland takes between four and nine months each year to reach minimum water levels.

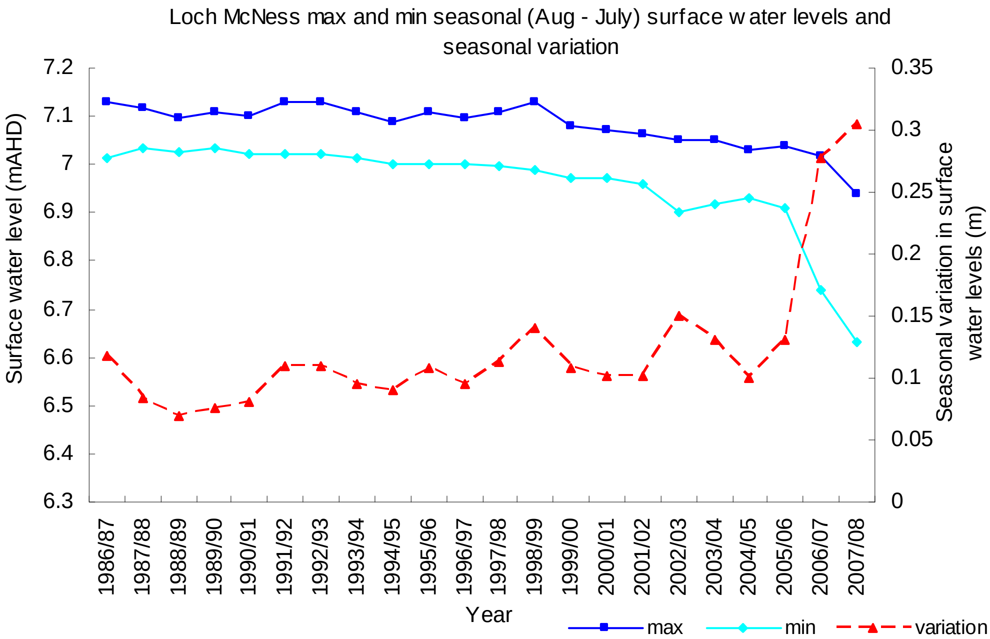


Figure 4.2: Maximum, minimum and seasonal (July–August) variations in surface water levels at the Loch McNess staff gauge over the periods 1986–87 to 2007–08.

## Lake Joondalup

Although Lake Joondalup has remained permanently inundated at the staff gauge since 1986–87, vast areas of the basin dry during most summers with maximum and minimum water levels and seasonal variations showing a declining trend (Figure 4.3). Over a 21-year period maximum water levels have consistently been achieved during September or October (Table 4.5). Although minimum levels have historically occurred in April or May (1999–00 to 2007–08) they appear to have occurred slightly earlier in recent years. Minimum levels have also been reached over a shorter time period in recent years.

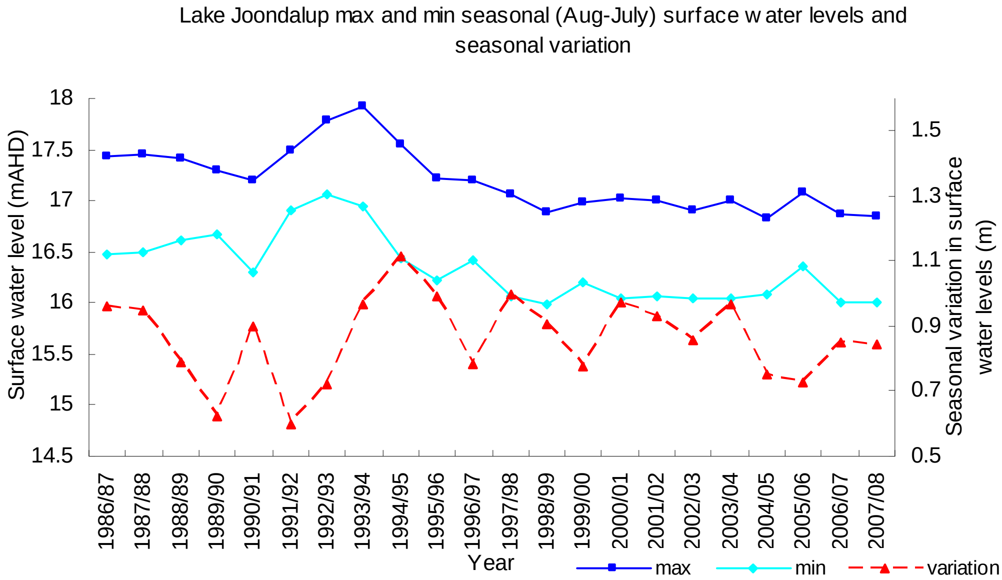


Figure 4.3: Maximum, minimum and seasonal (July-August) variations in surface water levels at the Lake Joondalup staff gauge over the periods 1986–87 to 2007–08.

## Lake Jandabup

Surface water at Lake Jandabup has been artificially maintained since the late 1990s, resulting in permanent inundation since 1999 and smaller declines in maximum and minimum water levels (Figure 4.4) than at other wetlands with similar characteristics (e.g. Lake Mariginiup and Lake Gnangara). The variation in water levels, however, has declined over time. Artificial maintenance may also have regulated the timing of maximum and minimum water levels, which have been become less variable in recent years (Table 4.5). Maximum and minimum water levels now occur in September and March respectively, and take months to decline.

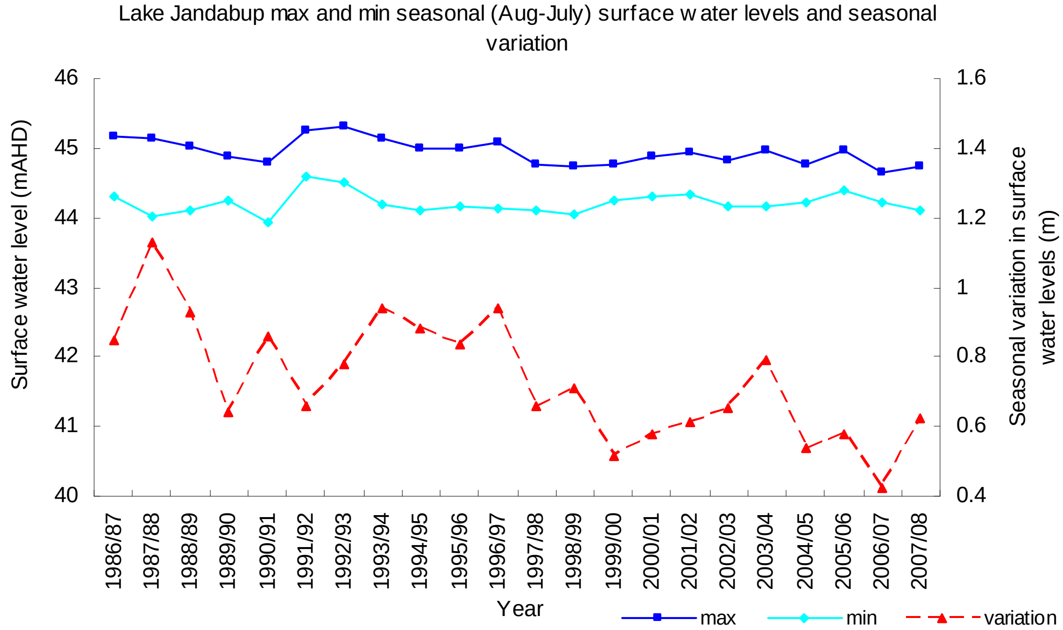


Figure 4.4: Maximum, minimum and seasonal (July-August) variations in surface water levels at the Lake Jandabup staff gauge over the periods 1986–87 to 2007–08.

### Lake Mariginiup

Lake Mariginiup was permanently inundated from 1987–88 to 1993–94 after which time it generally dried for longer periods each year until 2007–08. There has been a coinciding decline in maximum surface water levels (Figure 4.5). Despite the apparent levelling out of minimum surface water levels from 1999–2000, which in fact reflects drying of the wetland at the staff gauge, minimum surface water levels have also declined. Maximum water levels have occurred during September or October over the entire 21-year period (Table 4.5). However, the wetland has progressively dried earlier at the staff gauge and has dried quicker over time.

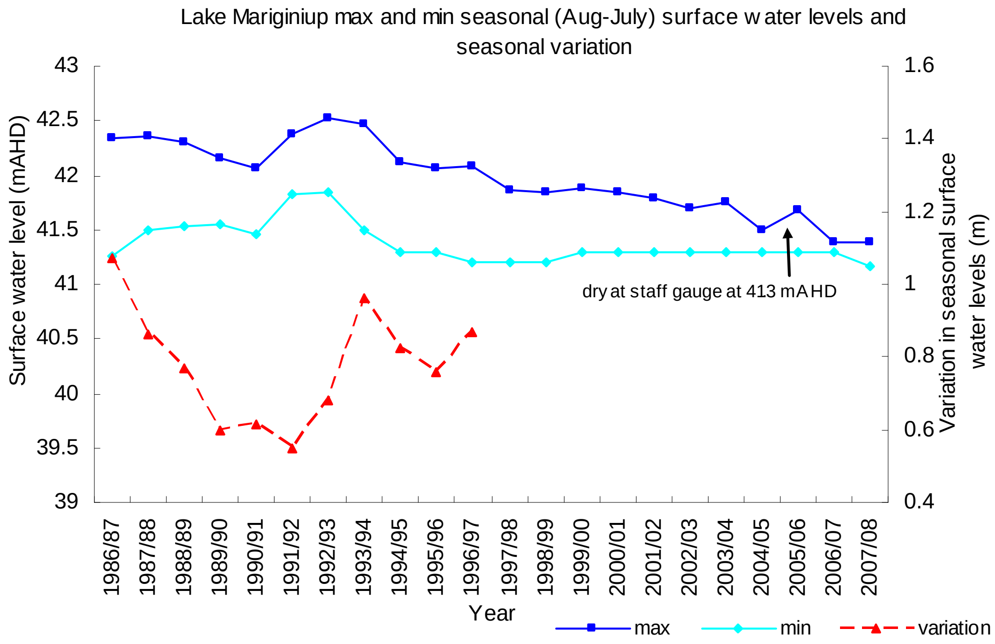


Figure 4.5: Maximum, minimum and seasonal (July–August) variation in surface water levels at the Lake Mariginiup staff gauge over the periods 1986–87 to 2007–08.

## EPP 173 wetland

This Bassendean dune wetland has dried at the staff gauge each year since surface water level monitoring commenced in 1996–97. Over this period, maximum water levels have been achieved from August to October (Table 4.5) and minimums from February to June. Prior to 2004–05 minimum levels were reached over shorter time periods with successive years. There has been a coinciding decline in surface water levels and seasonal variations (Figure 4.6).

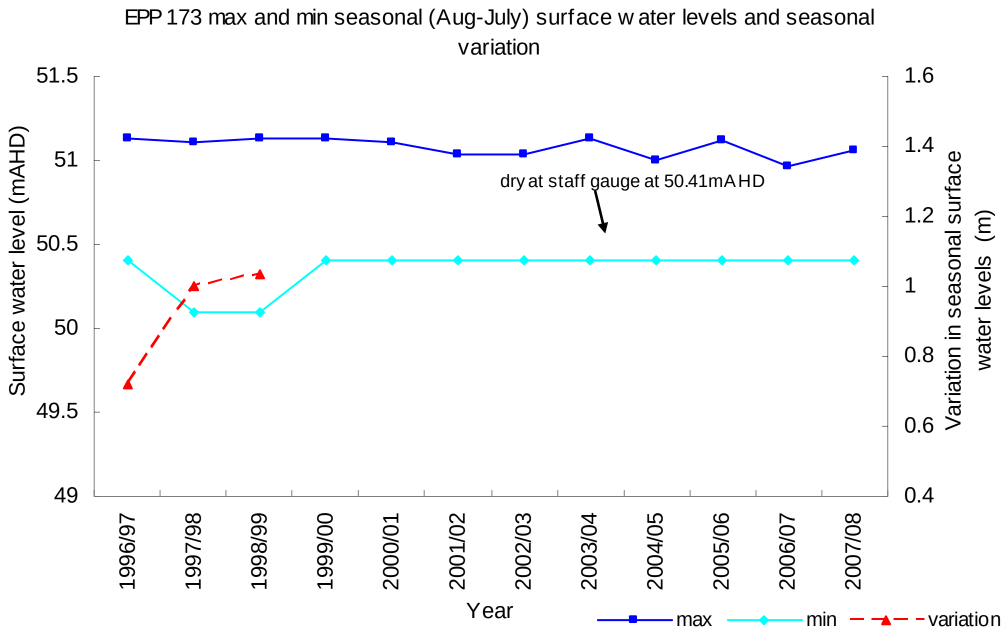


Figure 4.6: Maximum, minimum and seasonal (July-August) variations in surface water levels at the EPP 173 staff gauge over the periods 1996/97 – 2007/08.

Superimposed on the seasonal variations experienced by wetlands in a Mediterranean climate, the temporal and spatial changes in hydrology across the Gnangara groundwater system have serious implications for wetland sediments and biodiversity. Temporal changes in the form of altered hydro-periods have occurred in at least two ways. In urban areas where wetlands receive greater runoff, they can remain wetter for longer. Elsewhere, and much more widespread across the GSS study area, wetlands are drying earlier in the season and remain drier for longer periods. It is also noteworthy that for many of the wetlands, staff gauges no longer provide adequate measurements for seasonal minima. This is because the gauge was installed when water levels were higher and the gauge when dry can only register the sediment surface level, not the watertable which declines beneath it.

Table 4.5: Hydrological variables for 6 GSS study area wetlands over a 21-year period 1986–87 to 2007–08. The table includes months of minimum and maximum surface water levels and number of months taken each year for water levels to decline from maximum to minimum, or for the wetland to dry at the staff gauge.

| **Wetland** | **Joondalup** | | | **Loch McNess** | | | **Lake Jandabup** | | | **Lake Mariginiup** | | | **EPP 173** | | | **Big Carine Swamp** | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **Month Max** | **Month Min** | **No. months max–min** | **Month Max** | **Month Min** | **No. months max–min** | **Month Max** | **Month Min** | **No. months max–min** | **Month Max** | **Month Min** | **No. months max–min** | **Month Max** | **Month Min** | **No. months max–min** | **Month Max** | **Month Min** | **No. months max–min** |
| 1986–87 | Oct | Apr | 6 | Aug | Mar | 8 | Aug | Mar | 7 | Oct | May | 7 | - | - | - | Sept | Apr | 7 |
| 1987–88 | Sept | Mar | 5 | Aug | Feb | 7 | Aug | Mar | 7 | Sept | Apr | 7 | - | - | - | Aug | Apr | 8 |
| 1988–89 | Oct | May | 7 | Aug | Jan | 6 | Oct | Apr | 6 | Oct | May | 7 | - | - | - | Sept | Apr | 7 |
| 1989–90 | Oct | Apr | 6 | Aug | Jan | 6 | Oct | Mar | 5 | Oct | Apr | 6 | - | - | - | Oct | Apr | 6 |
| 1990–91 | Oct | May | 7 | Aug | Feb | 7 | Oct | Feb | 4 | Oct | Feb | 4 | - | - | - | Sept | Mar | 6 |
| 1991–92 | Oct | May | 7 | Aug | Feb | 7 | Oct | May | 7 | Oct | May | 7 | - | - | - | Sept | Apr | 7 |
| 1992–93 | Oct | Apr | 6 | Sept | Feb | 6 | Sept | Apr | 7 | Oct | Apr | 6 | - | - | - | Aug | May | 9 |
| 1993–94 | Oct | May | 7 | Sept | Feb | 6 | Nov | Apr | 5 | Oct | May | 7 | - | - | - | Oct | Apr | 7 |
| 1994–95 | Sept | May | 8 | July | Feb | 7 | Sept | Feb | 5 | Sept | May | 8 | - | - | - | Sept | May | 8 |
| 1995–96 | Sept | May | 8 | Aug | Mar | 8 | Sept | Apr | 7 | Sept | Mar | 6 | - | - | - | Sept | Mar | 6 |
| 1996–97 | Oct | Apr | 6 | Aug | Feb | 7 | Oct | Apr | 6 | Oct | Mar | 5 | Aug | Mar | 7 | Nov | Apr | 5 |
| 1997–98 | Oct | May | 7 | Aug | Feb | 7 | Oct | Feb | 4 | Oct | Jan | 3 | Sept | Feb | 5 | Sept | May | 8 |
| 1998–99 | Oct | Apr | 6 | Sept | Mar | 6 | Oct | Feb | 4 | Oct | Mar | 5 | Oct | Mar | 5 | Oct | Feb | 4 |
| 1999–00 | Oct | Mar | 5 | Sept | Jan | 4 | Oct | Mar | 5 | Oct | Jan | 3 | Sept | Apr | 6 | Oct | Mar | 5 |
| 2000–01 | Oct | Mar | 5 | Aug | Jan | 5 | Oct | Feb | 4 | Oct | Jan | 3 | Aug | Feb | 6 | Oct | June | 8 |
| 2001–02 | Oct | Apr | 6 | Aug | Jan | 5 | Sept | Mar | 6 | Oct | Jan | 3 | Oct | Feb | 5 | Oct | Apr | 6 |
| 2002–03 | Sept | Mar | 6 | Aug | Feb | 6 | Sept | Mar | 6 | Sept | Dec | 3 | Sept | Jan | 4 | Oct | Apr | 6 |
| 2003–04 | Oct | Apr | 6 | Aug | Feb | 6 | Sept | Mar | 6 | Sept | Dec | 3 | Oct | Mar | 5 | Oct | Dec | 2 |
| 2004–05 | Oct | Mar | 5 | June | Jan | 7 | Sept | Feb | 5 | Sept | Nov | 2 | Sept | June | 9 | Oct | Jan | 3 |
| 2005–06 | Oct | Apr | 6 | June | Mar | 9 | Sept | Mar | 6 | Sept | Jan | 4 | Aug | Mar | 7 | Oct | Jan | 3 |
| 2006–07 | Sept | Mar | 6 | July | Mar | 8 | Sept | Mar | 6 | Sept | Jan | 4 | Sept | Jan | 4 | Oct | Apr | 6 |
| 2007–08 | Oct | Apr | 6 | Aug | Mar | 7 | Sept | Mar | 6 | Sept | July | 10 | Sept | Feb | 5 | Oct | Apr | 6 |

# Wetland geomorphology and wetland sediments

Strongly related to the hydrological regimes are the soils and sediments of the Swan Coastal Plain. On the Gnangara groundwater system the large scale landforms run roughly parallel with the Darling Fault, the Gingin Scarp and the coast, corresponding with sedimentary formations. Landforms and groundwater-dependent ecosystems occur in the following sequence east to west (information adapted from Davidson 1995; McArthur and Bettenay 1960; Semeniuk and Semeniuk 2004 and others where specified).

**Pinjarra plain**

Landforms that occur as flat gently undulating alluvial fans fronting the Darling and Gingin Scarps, as well as the fluvial systems which form channels and associated floodplains underlain by Guildford clays. In the context of this chapter, this landform unit includes predominantly surface water features: almost the entirety of the Ellen Brook plus associated wetland systems, aeolian basins, and middle reaches of Gingin and Lennard Brook and associated systems. Because the clays are less permeable, the soils of the Pinjarra Plain form the eastern boundary of the Gnangara groundwater system.

**Bassendean dunes**

This geomorphological unit comprises undulating terrain of low (up to 20 m relative relief) old degraded dunes underlain by deep, heavily leached Bassendean sand (quartz) of Pleistocene age. The Bassendean sands contain the Gnangara groundwater system (large superficial aquifer). Most lakes, sumplands and damplands are (or were, see above and Chapter 5) hydraulically connected to the groundwater. Many other wetlands are now, or were originally, perched above the watertable where the downward and down-gradient leakage of water is inhibited by wetland sediments. In some cases the less permeable layer is a ferruginous hardpan termed 'coffee rock'. On the eastern boundary of the aquifer the groundwater is forced to the surface at a series of discharge points where waters encounter the relatively impervious Guildford clays. These discharge points include the tumulus mound springs which were locally relatively common (English and Blyth 2000). Wetlands occur predominantly in interdunal swales, or interbarrier depressions at the western edge of the Bassendean dunes where they interface with the Spearwood dunes landform. Their aeolian influence and age mean that these wetlands vary on a theme of circular basin shapes.

**Spearwood dunes**

These dunes are higher (to 60 m relative relief) and younger than eastern dune systems. They are near-continuous and linear, more conspicuously nearly parallel to one another, with intervening narrow and steep-sided depressions or narrow plains, underlain by Pleistocene limestone blanketed by quartz sand. Sands are less leached than the Bassendean dunes and formed from the weathering of underlying limestone. In the interdunal swales lie linear chains of elongated wetlands (some of them permanent and deeper than elsewhere on the Gnangara groundwater system), often with more conspicuous limestone ridges on their western edge. Along the coastal belt of limestone some wetlands lie in dolines where the land surface is karstic. These surfaces also contain larger subterranean spaces, such as caves, where groundwater-dependent biota occur.

**Quindalup dunes**

The most recent (Holocene) and most coastal landform consisting of quartzocalcareous sand dunes and beach ridge plains. Estuaries, or formative wetlands, in this landform will not be discussed in this chapter.

These landforms, combined with the hydrology and location , contribute to the identification of different groups of wetlands based on similarity of particular characteristics (consanguineous suites sensu Semeniuk 1988). The Swan Coastal Plain has been mapped for its geomorphic wetland categories (Table 4.1) and consanguineous suites at a scale of 1:100 000. This mapping database is held by the Western Australian Department of Environment and Conservation and shown in Figure 4.7 for the GSS study area.

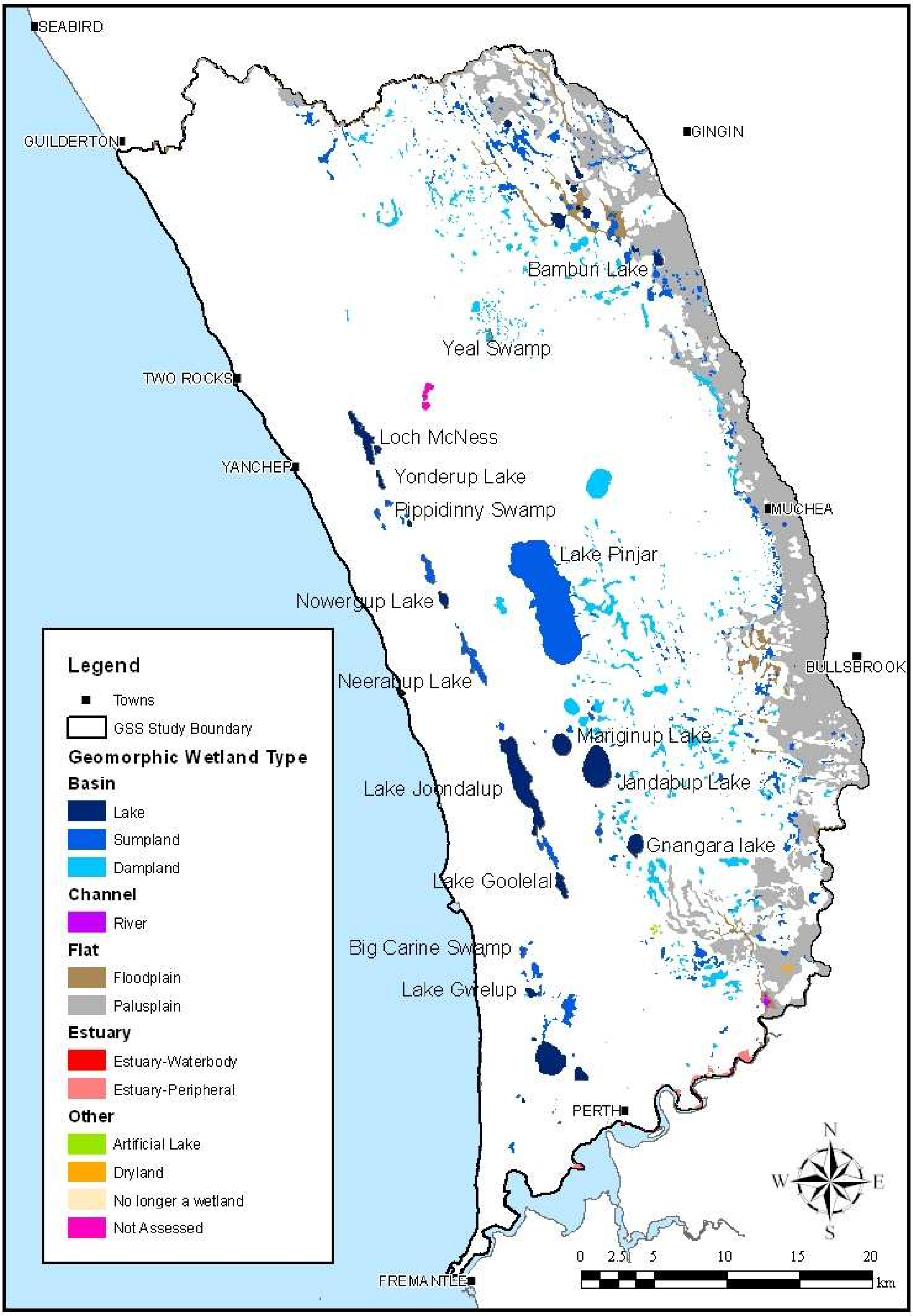


Figure 4.7: Wetland mapping for the GSS study area. Mapped wetlands are assigned geomorphic attributes in the database held by the Department of Environment and Conservation.

## Wetland sediments of the Gnangara groundwater system

The work of Semeniuk and Semeniuk (2004; 2005; 2006) has advanced our understanding of the types and distribution of sediments on the Swan Coastal Plain. It describes the main wetland sediment types as peat, diatomite and calcilutite (carbonate mud); wetlands can also have intermediaries with proportional mixtures of these (see Table 4.6). These wetland sediments accumulate in depressions under conditions of permanent, seasonal or intermittent saturation, mainly from internal biological processes (biogenic – from sponges, diatoms, shells, charophytes and so on) or from basin setting (internal) or extrabasinal (washed in) (terrigenic) processes. These accumulated sediments are then further subjected to biological, chemical and physical (diagenic) processes in situ. Consanguineous suites of wetlands, and those wetlands on major landforms, and in interbarrier depressions, tend to have characteristic types of sedimentary fill. Table 4.6 shows stratigraphic types and water characteristics for wetlands across dune systems of the Swan Coastal Plain. The information on basement setting, types of sedimentary fill and consanguineous suites is from Semeniuk and Semeniuk (2006).

Sediment characterisation should be the first step in the understanding of wetland management requirements (Sommer 2006). Sommer presented details of a physical and bulk sediment characterisation of the sediments of Lake Goollelal, a permanent, deeper, non-coloured, alkaline wetland in an interdunal Spearwood dune depression. Two distinct sediment types were identified: peat, and a suspended detrital floc (settled particles that coalesce to form an unconsolidated sediment). Both were organic-rich, sulfidic and iron-rich. Organic carbon to nitrogen ratios indicated that the peat organic matter was primarily of vascular plant origin, while the floc was of aquatic (algal) origin. There were higher concentrations of most elements in the floc, probably because of more reactive organic matter and consequently higher iron sulfide concentration which is very efficient at scavenging trace elements from the water column. From the literature it was deduced that unique microbial ecology and physical structure of the floc must be responsible for very high nitrogen and phosphorus concentrations. Both sediments became severely water repellent when dried. Allowing these permanently inundated sediments to dry partially may result in symptoms of either acidification or eutrophication. Allowing them to severely dry may irreversibly change the sediment–water relationships.

Understandings of wetland sediments like these therefore make significant contributions to conservation of flora and fauna and management of wetland systems overall.

Table 4.6: Stratigraphic types and water characteristics for wetlands across dune systems of the Swan Coastal Plain.

| **Dune system** | **Basement setting** | **Types of sedimentary fill** | **Consanguineous wetland suite** | **Wetland examples** | **Water characteristics** |
|---|---|---|---|---|---|
| Pinjarra Plain | Quartz sand, muddy sand | Kaolinite-dominated mud, muddy sand, extrabasinal sand | Mungala suite | Perth Airport swamps<br>(also Ellen Brook floodplains, Lake Chandala) | Tannin-rich, acidic (to alkaline), and cation poor |
| Pinjarra Plain–Bassendean dune contact | Quartz sand, muddy sand and mud | Kaolinite-dominated mud, muddy sand, extrabasinal sand, peat | Bennett Brook suite | Bennett Brook wetlands<br>(also middle reaches Gingin Brook) | |
| Bassendean | Quartz sand | Intrabasinal peat, diatomaceous peat, diatomite | Gnangara suite | Lakes Gnangara, Jandabup | Tannin-rich, acidic (to alkaline), and cation poor |
| Bassendean dune–Spearwood dune contact | Quartz sand | Diatomaceous peat, diatomite | Pinjar | (Lake Mariginiup) | Tannin-rich, acidic (to alkaline), moderate cation levels |
| Spearwood | Quartz sand and/or limestone | Intrabasinal peat, diatomaceous peat, or calcilutite, and extrabasinal quartz sand | Yanchep | Loch McNess, Lake Yonderup, Big Carine Lake, Lake Gwelup | Ranging from tannin-rich to tannin-poor, alkaline (to acidic), and cation enriched |

# Biodiversity: flora and fauna

## Common wetland vegetation types of the Swan Coastal Plain

Wetland vegetation of the Swan Coastal Plain has been commonly categorised at two levels: the uppermost level or ‘complex’ refers to vegetation units linked by dominant plant species and structural attributes, and the secondary level for classification, the ‘community’, based on common or typical species within the overall complex (Cresswell and Bridgewater 1985; Gibson *et al.* 1995; Pen 1981; Semeniuk *et al.* 1990). Table 4.7 (adapted from Cresswell and Bridgewater 1985; Gibson *et al.* 1994; Pen 1981) lists the most widespread species assemblages across wetlands of the Swan Coastal Plain and draws on common elements of vegetation classifications. Note however that specific species within communities may vary.

Table 4.7: Wetland vegetation complexes and their common communities (based on dominant species) of the Swan Coastal Plain.

| **Vegetation complex** | **Typical vegetation communities** | | | |
|---|---|---|---|---|
| | **a** | **b** | **c** | **d** |
| *1. Typha-Baumea* | *Typha orientalis* | *T. orientalis – B. articulata* | *Baumea articulata* | |
| *2. Melaleuca rhaphiophylla* | *Schoenoplectus validus* | *Eucalyptus rudis* | *Melaleuca teretifolia* | *Gahnia trifida* |
| | *Carex fascicularis* | *Persicaria salicifolia* | *Lepidosperma longitudinale* | *Lepidosperma longitudinale* |
| | | *Centella asiatica* | *Astartea fascicularis* | *Baumea juncea* |
| *3. Melaleuca preissiana* | *Pericalymma ellipticum* | *Pericalymma ellipticum* | *Hypocalymma angustifolium* | *Banksia littoralis* |
| | *Hypocalymma angustifolium* | *Kunzea ericifolia* | *Ehrharta spp.* | *Baumea vaginalis* |
| | *Astartea fascicularis* | *Banksia ilicifolia* | | *Kunzea ericifolia* |
| *4. Melaleuca viminea* | *Melaleuca uncinata* | *Viminaria juncea* | *Eutaxia virgata* | |
| | *Melaleuca cuticularis* | *Verticordia densiflora* | *Astartea fascicularis* | |
| | *Schoenus spp.* | *Pericalymma ellipticum* | *Hakea varia* | |
| *5. Eucalyptus rudis* | *Melaleuca rhaphiophylla* | *Astartea fascicularis* | *Melaleuca teretifolia* | |
| | *Kunzea ericifolia* | *Lepidosperma longitudinale* | *Lepidosperma longitudinale* | |
| | *Hakea varia* | | *Baumea articulata* | |
| | *Melaleuca preissiana* | | *Astartea fascicularis* | |
| | *Baumea vaginalis* | | | |
| *6. Juncus kraussii* | *Melaleuca rhaphiophylla* | *Melaleuca rhaphiophylla* | | |
| | *Centella asiatica* | *Baumea juncea* | | |
| | | *Melaleuca teretifolia* | | |
| *7. Lepidosperma longitudinale* | *Astartea fascicularis* | *Viminaria juncea* | *Melaleuca teretifolia* | |
| | *Schoenus subfascicularis* | | | |
| | *Banksia littoralis* | | | |
| | *Melaleuca preissiana* | | | |
| | *Eutaxia virgata* | | | |
| *8. Casuarina obesa* | *Melaleuca rhaphiophylla* | | | |
| | *Samolus repens* | | | |
| | *Sarcocornia quinqueflora* | | | |
| | *Melaleuca viminea* | | | |

A complex mosaic of vegetation structures or communities may be present within wetland systems. In addition, gradients in environmental variables, typically from the margin to the centre of the wetland, may result in gradual transitions between communities. Lake Joondalup, for example, consists of fringing *Melaleuca rhaphiophylla–Eucalyptus rudis* low-forest and *Typha orientalis–Baumea articulata* sedgeland (Cresswell and Bridgewater 1985; Semeniuk *et al.* 1990). Typically *T. orientalis* and *B. articulata* are the most common fringing emergent macrophyte species across wetlands on the Swan Coastal Plain, with the exotic *Typha* species becoming more dominant in wetlands subject to altered water regimes and increased nutrient content (Froend and McComb 1994).

Wetlands have been classified using vegetation pattern and form, by determining the scale, extent of vegetation cover, internal organisation of vegetation, structure of vegetation zones and the floristics of the vegetation zones. The spatial extent of wetland systems and names used to describe this category are (Semeniuk 1987; Semeniuk *et al.* 1990):

- mega – over 10 000 ha
- macro – 10 000 to 100 ha
- meso – 100 to 25 ha
- micro – 25 to 1 ha
- lepto – less than 1 ha.

The pattern of vegetation units and the extent of vegetation cover within the wetlands can be described as:

- homogeneous – consisting of a single dominant vegetation assemblage
- zoned or heterogeneous – containing a number of different vegetation assemblages.

The areal extent of vegetation can be described as (Semeniuk *et al.* 1990):

- complete – covering over 90% of the wetland complex
- mosaic – occurring in no defined pattern
- peripheral – fringing the wetland complex.

By combining these elements with the dominant wetland communities of the Swan Coastal Plain as described by Cresswell and Bridgewater (1985) and Gibson *et al.* (1995), a detailed description of specific wetland complexes across the plain can be developed. (Table 4.8).

Table 4.8: Examples of 'Semeniuk' vegetation classification and likely corresponding species assemblages for major Gnangara groundwater system wetlands.

| **Wetland** | **Classification** | **Species assemblage** |
|---|---|---|
| Lake Pinjar | Mega maculiform closed low forest / heath/sedgeland | *M. rhaphiophylla/E. rudis/M. preissiana/ M. teretifolia/Lepidosperma sp./B. articulata* |
| Lake Mungala | Micro zoniform woodland/low forest/shrubland | *M. preissiana/M. rhaphiophylla/M. teretifolia* |
| Lake Joondalup | Macro zoniform closed low forest/sedgeland | *M. rhaphiophylla-E. rudis/B. articulata-T. orientalis* |
| Lake Jandabup | Macro bacataform open woodland sedgeland | *M. preissiana.B. articulata/Baumea-Juncus-Lepidosperma-Leptocarpus-Typha* |
| Lake Gnangara | Macro zoniform open woodland/open forest | *M. preissiana/M. rhaphiophylla-Baumea-Juncus-Lepidosperma-Leptocarpus-Typha* |
| Lake Carabooda | Macro maculiform open low forest/tall scrub/sedgeland | *M. rhaphiophylla/E. rudis-B. littoralis/T. orientalis* |

Hopper and Burbidge (1989) suggest that *Banksia* woodlands in general appear to contain few rare localised endemic plant species. This accords with the conclusions reached by Horwitz *et al*. (in press, for invertebrates, below) for invertebrates (see Aquatic invertebrates section below). Some less common or less widespread species are found in ecological communities in groundwater-dependent ecosystems on the Gnangara groundwater system, plant species that are either highly disjunct or south-west or Darling Range species that are otherwise uncommon on the Swan Coastal Plain (Pinder 2005). Wetlands can feature a variety of flowering plants that are highly disjunct or at the ends of their (frequently mesic) ranges. Lakes or swamps with often long periods of inundation have disjunct populations of ferns, orchids, sedges, herbs and shrubs. The three well surveyed occurrences of mound springs also contain a number of plant species that are highly disjunct or at the ends of their southern ranges: Examples include several liverworts, ferns (*Cyclosorus gongyloides* and *Lycopodium serpentinum*), orchids, sedges (*Cyathochaeta teretifolia* and *Empodissima gracillima*), shrubs (*Hibbertia perfoliata* and *Boronia molloyae*) and trees (*Homalospermum firmum*). *Hibbertia perfoliata* was presumed to have become extinct on the Swan Coastal Plain until found on these mound springs recently (Pinder 2005).

Characteristic aquatic plants are those that are submerged in wetland water for some or all of the year, or floating on water, and many species are known from wetlands on the Gnangara groundwater system (Table 4.9).

Table 4.9: Selected genera of aquatic plants and their species assessed as likely to be found in Gnangara groundwater system wetlands based on distribution records (Western Australian Herbarium 1998-2009). Species are native unless otherwise indicated.

| **Genus** | **Species** |
|---|---|
| *Hydrocotyle* | *alata, blepharocarpa, diantha, hispidula, lemnoides, pillifera, striata* and *callicarpa* |
| *Myriophyllum* | *aquaticum* (alien), *crispatum, drummondii, echinatum, limnophilum* and *tillaeoides* |
| *Villarsia* | *albiflora, capitata, submersa* and *violifolia* |
| *Azolla* | *filiculoides* |
| *Lemna* | *disperma* |

## Non-vascular plants

Conspicuous forms of non-vascular plants are the larger stoneworts or Charophytes (e.g. *Chara* and *Nitella*) that are known to occur in wetlands on the Swan Coastal Plain. Wrigley *et al*. (1991) described the species of blue-green bacteria and genera of green algae present in 16 Gnangara groundwater system wetlands, and showed that the forms present were clearly related to the degree of nutrient enrichment present in the waterbodies. The blue-green species *Anabaena spiroides*, *Microcystis aeruginosa* and *Oscillatoria* were most commonly (but not only) present in enriched wetlands. Fourteen genera of green algae were recorded, with the filamentous forms *Oedogonium*, *Mougeotia* and *Spirogira* the most common forms. They found that diatoms were dominant in relatively nutrient poor and coloured wetlands.

McHugh (2004) used fossil diatoms sampled from sediment sequences, and a statistically derived relationship between modern diatom distribution and low pH conditions, to reconstruct the history for three lakes on the Gnangara groundwater system (Jandabup, Gnangara, Mariginiup)(see also Chapter 6). Her work described 106 taxa (at the species or subspecies level), from 30 genera. It remains the most comprehensive list of diatoms likely to occur in wetlands on the Gnangara groundwater system.

## Aquatic invertebrates

A compilation of aquatic invertebrate taxa recorded from wetlands (on both the Gnangara groundwater system and Jandakot Mound) between 1977 and 2003, from 18 studies of 66 wetlands, was prepared by Horwitz *et al*. (in press) and has revealed a surprisingly high richness considering the comparatively small survey area and the degree of anthropogenic alteration of the plain. A total of over 500 taxa from 176 families or higher order taxonomic levels were identified.

In general the aquatic invertebrate fauna consisted of some highly diverse groups:

- a dytiscid water beetle assemblage probably in excess of 50 species
- a microcrustacean fauna with over 60 cladoceran species or sub-species
- over 30 copepod species
- over 40 ostracod species.

Moderately diverse groups included:

- chironomid midges (31 species)
- at least 10 culicid mosquito species (although several mosquito taxa are unlikely to be collected using standard invertebrate collection techniques)
- at least 9 corixid and 10 notonectid species of hemipterans (backswimmers and water boatmen)
- at least 11 damselfly (Zygoptera) and 12 dragonfly (Anisoptera) species
- at least 28 aquatic mite taxa
- about 14 inland aquatic molluscan snail and bivalve species (some of which are introduced)
- at least 17 oligochaete worm taxa.

Some present, but relatively depauperate groups were notable:

- the ephemeropteran (mayfly)
- trichopteran (caddisfly)
- amphipod (scud shrimp) fauna, and fauna which were found more commonly in clear, cool, flowing freshwater habitats (e.g. Simuliidae and Plecoptera).

When compared to other regional surveys of wetland invertebrates, proportionate local and regional endemism appear relatively low for the survey area; only three species are thought to be endemic to the GSS study area (one amphipod, *Hurleya* sp. and two copepod species; see below). Overall Horwitz *et al.* (in press) proposed that levels of richness and endemism are attributable to geologically recent colonising forces associated with sea level changes, dune formation and periods of aridity on the plain. Invertebrate colonisation from multiple directions is possible: from the cooler southern, the warmer northern, and from the flowing and non-flowing wetlands on the adjacent Darling Scarp.

Rare invertebrate taxa appear to be associated with rare wetland types harbouring very specific (and perhaps unusual) microhabitats such as caves and tumulus springs. Horwitz *et al.* (in press) have re-evaluated important wetlands from an invertebrate perspective on the Gnangara groundwater system (Table 4.10).

High priority wetlands with ‘significant’ invertebrate fauna in terms of aquatic invertebrate richness, endemism and/or rarity include:

- aquatic habitats in cave systems in karstic areas around Yanchep
- permanent deeper surface waters in northern linear chain wetlands of the Spearwood interdunal system
- tumulus springs (organic mound springs) in the Ellen Brook region of the eastern Gnangara groundwater system
- surface waters in the Ellen Brook region of the eastern Gnangara groundwater system
- habitat complexes in large shallow wetland systems on the interface between Bassendean dune and Pinjarra Plain systems.

Table 4.10: High priority wetlands on the Gnangara groundwater system in terms of richness, endemism or rarity criteria for aquatic invertebrate records. Wetlands are ordered from north to south. (adapted from Horwitz *et al.* in press).

| High priority wetland | Richness[1] | Endemism[2] | Rarity[3] | Wetland habitat descriptors |
|---|---|---|---|---|
| Yanchep Caves | | LE | X | Underground karstic stream, root mat fauna |
| Loch McNess | X | RE | | Permanent lake, spring, karstic system, diverse littoral vegetation communities, low conductivity, low colour and low turbidity |
| Lake Yonderup | X | | | Permanent lake, karstic system, unconsolidated and consolidated organic soils, low conductivity, low colour and low turbidity |
| Lake Nowergup | X | | | Deep, permanent lake, Spearwood sands, diverse littoral vegetation communities, unconsolidated and consolidated organic sediment, low conductivity, low colour and low turbidity |
| Lake Jandabup | X | RE | | Semi permanent, weakly-coloured water, a mix of diatomaceous-organic sediment and leached Bassendean dune sands, relatively shallow, with a variable drying regime but mostly with complex littoral vegetation communities that are seasonally inundated |
| Twin Swamps | X | RE | X | Discrete bodies of shallow seasonal surface water influenced by both the clays of the Pinjarra Plains (Guildford) and sands of the Bassendean dunes, with associated complex littoral vegetation communities and darkly stained water |
| Muchea/Peter's Spring, Kings Spring, Bullsbrook Channel, Edgerton Spring, Edgecombe Lake, Nursery Dam | | LE, RE | X | Mound spring, or small (created) depression fed by spring |
| Ellenbrook Nature Reserve | | RE | X | Shallow seasonal clay-based wetland fed by surface run-off on Pinjarra Plain, with littoral vegetation |

[1] Richness – more than 100 species and/or 65 Families shown with 'X'

[2] Endemism – wetlands with known local endemic species (LE; restricted to the Swan Coastal Plain bioregion) or with greater than 20% regional endemics (RE; restricted to the South-west Australian Floristic Region)

[3] Rarity – wetlands with more than 25% rare taxa shown with 'X'

To summarise habitat specificity, invertebrate data for 16 regularly monitored wetlands on the Gnangara groundwater system, analysed by Sommer *et al.* (2008) for their frequency of occurrence in 16 different vegetation communities (as a surrogate for 'habitat'), reveals the following generalities:

- most invertebrates are found in most habitats, and most vegetation types have similar invertebrate assemblages, suggesting relatively little habitat specificity
- recorded richness of invertebrates is related to sampling effort, not necessarily to habitat type
- very few taxa show some degree of habitat specificity
- pattern analyses show that the eastern Bassendean wetlands form outliers in terms of both the vegetation communities sampled and their invertebrate assemblages.

Analyses undertaken by Sommer *et al.* (2008) and Horwitz *et al.* (in press), by compiling existing data from a range of independent studies, have highlighted deficiencies in knowledge of the GSS study area and beyond. It is clear that there is a lack of invertebrate data from wetlands on the north-eastern portion of the Gnangara groundwater system and from wetlands immediately north of the Moore River–Gingin boundary, is. This lack of data means that some wetlands that might actually conform to the 'high priority' definition given in Table 4.10 above have not been classified as such.

Limited but strategic sampling was carried out in September 2008 as part of the Gnangara Sustainability Strategy. Three wetlands were examined in the north-eastern part of the Gnangara groundwater system. These were Lake Bambun, Yeal Lake and Quin Brook. The wetlands appear to have an 'average' aquatic macroinvertebrate assemblage (including some species with restricted distribution). From these initial investigations, evidence for degradation similar to that seen in other parts of the GSS study area was noted, including nutrient enrichment and weed infestation at all of the wetlands. Of particular interest was the intact nature of the surface hydrology, particularly for Yeal Lake and Quin Brook, despite some drain construction in their upper reaches. These three wetlands have now been added to the Gnangara groundwater system Macroinvertebrate Monitoring Program (EPA Section 46) and may be monitored twice yearly.

Furthermore, there is a suggestion that the Moore River–Gingin Brook 'line' across the Swan Coastal Plain might be regarded as a significant biogeographical boundary for inland aquatic fauna and this boundary may require specific management attention or more detailed surveying to search for floral or faunal elements thought now to be either locally endemic or regionally extinct or both (see below).

## Wetland vertebrate fauna

### *Mammals*

Of the 23 native mammal species likely to be found in the Gnangara groundwater system area, at least four are likely to be significant from a wetland perspective. The honey possum, or noolbenger, (*Tarsipes rostratus*) has been cited by Froend *et al.* (2004) as one of few surviving native mammals not currently monitored but highly dependent on the phreatophytic *Banksia* spp. for its year-round nectar supply. The water rat, or rakali, (*Hydromys chrysogaster*), requires permanent water for at least part of the year, dispersing to seasonal wetlands when conditions are suitable (Froend *et al.* 2004). Recent work as part of the Gnangara Sustainability Strategy will provide important detail on the ecology and water dependence for this species. Bamford and Bamford (2003) regarded wetlands and their margins to be a significantly productive habitat for the chuditch (*Dasyuris geoffroii*). They also referred to the preference of the southern brown bandicoot, or quenda, (*Isoodon obesulus*) for denser vegetation and association with wetland habitats. The quenda and rakali are both listed as Priority 4 species by the Department of Environment and Conservation.

### *Birds, particularly waterbirds*

A list of 172 bird species that have been observed or that are expected to make regular use of the area encompassed by the Gnangara groundwater system has been compiled by Bamford and Bamford (2003) and most of the following comes from their work (unless otherwise cited). About 10% of the species in the area are land bird species that make some use of vegetation around wetlands, so having some degree of dependency on phreatophytic vegetation. Another 5% of the species might be regarded as vagrants or introduced. Of the remainder, around half of the species from the area are waterbirds.

Storey *et al*. (1993) recorded 79 species of waterbirds on the Swan Coastal Plain. A list of 58 common waterbirds recorded at wetlands in the Perth metropolitan area by Bekle (1981) and Cole (2003) is provided in Table 4.11.

Of the 67 species for which numerical data are available, Bamford and Bamford (2003) note that highest counts were made for 26 species from Lake Joondalup and 17 species from Lake Jandabup. Highest counts for one or more species were also recorded for Lake Chandala (five species), Lakes Goollelal, Mungala and Bambun (three species), and Lake Nowergup, Emu-Ballajura Ponds, Big Carine Swamp, Loch McNess (one or two species).

To emphasise the importance of abundance data for waterbird conservation, recent unpublished bird counts reported in the *Yellagonga Integrated Catchment Management Plan 2009–2014* (City of Wanneroo 2009), show Lake Joondalup to have a very high bird count for some species, including the Eurasian coot (*Fulica atra*), Pacific black duck (*Anas superciliosa*), hardhead (*Aythya australis*) and the black-winged stilt (*Himantopus himantopus*).

Storey *et al*. (1993) concluded that wetlands with high species richness and abundance of water birds tended to be the deeper, larger wetlands with high productivity, low colour and plenty of fringing emergent vegetation. A total of 39 species were recorded breeding in wetlands of the Swan Coastal Plain.

Significant waterbird species include those listed on schedules of state or federal biodiversity conservation legislation, or listed as migratory species and thereby protected by the EPBC Act for migratory bird agreements and/or Bonn Convention (Table 4.12). Bamford and Bamford (2003) also mention species listed in Bush Forever as ‘Significant species’, including nine more species (including ducks and waterbirds). They regard all these species as being very sensitive to the effects of groundwater decline.

Table 4.11: Waterbirds of the Perth Region reported by Bekle (1981) during a census undertaken between March 1980 and June 1981 (Source 1) and by Cole's (2003) annual report for the period June 2002 – May 2003 (Source 2).

| Scientific name | Common name | Source | Scientific name | Common name | Source |
|---|---|---|---|---|---|
| *Actitis hypoleucos* | common sandpiper | 1 , 2 | *Gallirallus philippensis* | buff-banded rail | 1 , 2 |
| *Anas Castanea* | chestnut teal | 1 , 2 | *Gallinula ventralis* | black-tailed native hen | 1 |
| *Anas gracilis* | grey teal | 1 , 2 | *Himantopus himantopus* | black-winged stilt | 1 , 2 |
| *Anas rhynchotis* | Australian shoveler | 1 , 2 | *Limosa limosa* | bar-tailed godwit | 1 |
| *Anas platyrhynchos* | mallard | 2 | *Malacorhynchus membranaceus* | pink-eared duck | 1 , 2 |
| *Anas superciliosa* | Pacific black duck | 1 , 2 | *Nycticorax caledonicus* | nankeen night heron | 1 , 2 |
| *Anhinga melanogaster* | darter | 1 , 2 | *Oxyura australis* | blue-billed duck | 1 , 2 |
| *Ardea alba* | great egret | 1 , 2 | *Pandion haliaetus* | osprey | 1 , 2 |
| *Ardea pacifica* | white-necked heron | 1 , 2 | *Pelecanus conspicillatus* | Australian pelican | 1 , 2 |
| *Aythya australis* | hardhead | 1 , 2 | *Phalacrocorax carbo* | great cormorant | 1 , 2 |
| *Biziura lobata* | musk duck | 1 , 2 | *Phalacrocorax melanoleucos* | little pied cormorant | 1 , 2 |
| *Calidris acuminata* | sharp-tailed sandpiper | 2 | *Phalacrocorax sulcirostris* | little black cormorant | 1 , 2 |
| *Calidris ruficollis* | red-necked stint | 2 | *Phalacrocorax varius* | pied cormorant | 1 , 2 |
| *Calidris subminuta* | long-toed stint | 2 | *Platalea flavipes* | yellow-billed spoonbill | 1 , 2 |
| *Calidris ferruginea* | curlew sandpiper | 2 | *Plegadis falcinellus* | glossy ibis | 1 |
| *Cairina moschata* | muscovy duck* | 1 , 2 | *Podiceps cristatus australis* | great crested grebe | 1 |
| *Charadrius ruficapillus* | red-capped plover | 2 | *Poliocephalus poliocephalus* | hoary-headed grebe | 1 , 2 |
| *Charadrius melanops* | black-fronted plover | 2 | *Porphyrio porphyrio* | purple swamphen | 1 , 2 |
| *Chenonetta jubata* | Australian wood duck | 1 , 2 | *Porzana fluminea* | Australian spotted crake | 1 , 2 |
| *Circus approximans* | swamp harrier | 1 , 2 | *Porzana pusilla* | baillon's crake | 1 , 2 |
| *Cladorhynchus leucocephalus* | banded stilt | 2 | *Porzana tabuensis* | spotless crake | 1 , 2 |
| *Cygnus atratus* | black swan | 1 , 2 | *Recurvirostra novaehollandiae* | red-necked avocet | 1 |
| *Dendrocygna eytoni* | plumed whistling duck | 1 | *Tachybaptus novaehollandiae* | black-throated grebe | 1 |
| *Egretta novaehollandiae* | white-faced heron | 1 , 2 | *Tadorna tadornoides* | Australian shelduck | 1 , 2 |
| *Elseyornis melanops* | black-fronted dotterel | 1 , 2 | *Threskiornis aethiopica* | sacred ibis | 1 |
| *Erythrogonys cinctus* | red-kneed dotterel | 1 , 2 | *Threskiornis molucca* | Australian white ibis | 1 , 2 |
| *Falco cenchroides* | nankeen kestrel | 1 , 2 | *Threskiornis spinicollis* | straw-necked ibis | 1 , 2 |
| *Fulica atra* | Eurasian coot | 1 , 2 | *Tringa glareola* | wood sandpiper | 1 |
| *Gallinula tenebrosa* | dusky moorhen | 2 | *Tringa nebularia* | common greenshank | 1 , 2 |

Table 4.12: Waterbirds species listed on schedules of state or federal biodiversity conservation legislation, or listed as migratory species and thereby protected by the EPBC Act for migratory bird agreements and/or Bonn Convention.

| **Common name** | ***Scientific name*** | **Protection mechanism** |
|---|---|---|
| Australasian bittern | *Botaurus poiciloptilus* | Listed as Vulnerable under EPBC Act, WA Wildlife Conservation Act |
| little bittern | *Ixobrychus minutes* | Listed as Priority 4 under WA Wildlife Conservation Act |
| glossy ibis | *Plegadis falcinellus* | JAMBA/CAMBA listed species (EPBC Act) |
| great egret | *Egretta alba* | JAMBA/CAMBA listed species (EPBC Act) |
| black-tailed godwit | *Limosa limosa* | JAMBA/CAMBA listed species (EPBC Act) |
| long-toed stint | *Calidris subminuta* | JAMBA/CAMBA listed species (EPBC Act) |
| curlew sandpiper | *Calidris ferruginea* | JAMBA/CAMBA listed species (EPBC Act) |
| pectoral sandpiper | *Calidris melanotus* | JAMBA/CAMBA listed species (EPBC Act) |
| red-necked stint | *Calidris ruficollis* | JAMBA/CAMBA listed species (EPBC Act) |
| sharp-tailed sandpiper | *Calidris acuminata* | JAMBA/CAMBA listed species (EPBC Act) |
| wood sandpiper | *Tringa glareola* | JAMBA/CAMBA listed species (EPBC Act) |
| marsh sandpiper | *Tringa stagnatalis* | JAMBA/CAMBA listed species (EPBC Act) |
| common sandpiper | *Tringa hypoleucos* | JAMBA/CAMBA listed species (EPBC Act) |
| common greenshank | *Tringa nebularia* | JAMBA/CAMBA listed species (EPBC Act) |
| rainbow bee-eater | *Merops ornatus* | JAMBA/CAMBA listed species (EPBC Act) |

## *Reptiles and frogs*

The Gnangara groundwater system area has approximately 54 species of reptiles, and of these only five species are associated with dense vegetation and seasonally damp soils around wetlands (Bamford and Bamford 2003). However, it is the two species of freshwater tortoises that attract the most herpetological attention on the Gnangara groundwater system, and the western swamp tortoise (*Pseudemydura umbrina*) in particular may well be the most high profile wetland species on the Gnangara groundwater system. It is listed nationally under the EPBC Act 1999 as critically endangered and in Western Australia is listed under the *Wildlife Conservation Act 1950* as rare or likely to become extinct. Internationally, the western swamp tortoise is listed as critically endangered on the 2008 *IUCN Red list of threatened species* as well as being listed under Family Chelidae (Appendix I) of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) (Burbidge and Kuchling 2007). Only one viable population survives at Ellen Brook Nature Reserve with two other locations, Twin Swamps and Mogumber Reserve populations being maintained with translocated individuals. Processes likely to influence the long-term survival of the species include habitat losses from agriculture and clay mining, swamp drainage, a geographically highly restricted range, inappropriate fire regimes and predation of eggs and hatchlings. The decline of the species is attributed mainly to fox predation that occurs year round but particularly during summer (Burbidge and Kuchling 2007).

The oblong turtle (*Chelodina oblonga*) is close to the northern limit of its distribution in the Gnangara groundwater system area where it occurs in permanent and seasonal wetlands. Like other freshwater turtles, it requires a number of different habitat types to satisfy its life history requirements: sufficient water depths for swimming and mating, an adequate terrestrial buffer surrounding a wetland for nesting, shelter for hatchlings and juveniles within the waterbody, access to adequate food resources with safe foraging areas, and basking and aestivation sites (Giles *et al.* 2008). Three concerns have been expressed for the ongoing survival of this species in wetlands (and this applies particularly to the populations on the Gnangara groundwater system). The first is that as wetlands change from permanent or seasonal to intermittent inundation, adult turtles will have to either relocate or perish (Bamford and Bamford 2003). The second concern is the widespread occurrence of foxes which means that recruitment of hatchlings into the populations should decline, and because of the longevity of turtles this decline has not yet been noticed in turtle populations (J. Giles, pers. comm.). Finally, hatchling turtles are top level predators and therefore important to wetland ecology; they are also important predators of invertebrates including nuisance species such as midges and mosquitoes. Any decline may have detrimental effects on the ecology of permanent and seasonal wetlands.

The frog fauna found in wetland systems of the Gnangara groundwater system has been recently summarised by Bamford and Bamford (2003), and ten species were recognised. With the exception of the turtle frog, all species were highly or very highly sensitive to groundwater drawdown (Bamford and Bamford 2003). Falling groundwater levels affect frog species in wetlands on the Gnangara groundwater system primarily by reducing breeding and recruitment of metamorphs (recently developed frogs from tadpole stage). The metamorphosis stage is a critical time when falling groundwater levels could most affect survival rates (Froend *et al.* 2004). The relatively long life span of many adult frogs means surveys which count adult males may fail to detect cessation of breeding until the population is irretrievably depleted. None of the species is listed as threatened on schedules of federal or state biodiversity legislation, but two (the quacking frog *Crinia georgiana* and Glauert's froglet *Crinia glauerti*) are regarded as being of conservation significance because their occurrences in the Gnangara groundwater system wetlands are at the northern most extent of their distributions and the degree to which they are dependent on wetland habitats throughout the year. More detailed treatment of frogs is provided in Chapters 3 and 11.

## *Fish*

Eight native species of fish are known from the inland waters within the area encompassed by the Gnangara Sustainability Strategy (Table 4.13). Two of them, *Nannatherina balstoni* (Balston's pygmy perch) and *Geotria australis* (lamprey), may now be regionally extinct. Two more, *Galaxiella nigrostriata* (black-striped minnow) and *Galaxiella munda* (mud minnow) are restricted in their distribution and listed as vulnerable in the EPBC Act 1999. Threats to fish populations from introduced species such as *Gambusia holbrooki* and *Geophagus brasilensis* and several others may be as severe as, and compound the effects of, other changes to habitat loss such as changes in water regime.

One wetland with high conservation values on the Gnangara groundwater system is EPP 173 at Melaleuca Park, an interdunal wetland on the Bassendean dune landform. The very darkly stained water of EPP 173 and interactions between light attenuation and solar heating provide ideal conditions for the development and maintenance of strong daytime thermal stratification over summer, resulting in the maintenance of cooler benthic waters and the continued survival of an outlier population of the endemic *Galaxiella nigrostriata* (black-striped minnow) (Knott *et al.* 2002). Here the potential for future extinction of the resident population of *G. nigrostriata* will increase with the encroachment of further development (Knott *et al.* 2002). In this instance, reduction of the volume of groundwater entering EPP 173 occurs to the point where summer thermal stratification is not maintained. A range of factors can have dramatic effects on the resident relict *G. nigrostriata* population including: mixing of the water column by wind, total loss of surface water, dilution of water with nutrient enriched surface runoff and associated algal blooms

Given the widespread nature of hydrological change and of habitat loss and degradation on the Gnangara groundwater system, any wetland harbouring a population of native freshwater fish should be regarded as having a high management priority (Sommer *et al.* 2008). A corollary of this is important too: introduced fish are very capable of changing the structure of invertebrate populations and wetland ecology in general, so wetlands where there have been reliable historical records of an absence of introduced fish have considerable value and should be monitored regularly to limit the establishment of invasive species (and arguably introduction of native species too).

Table 4.13: Fish species currently found in wetlands of the Gnangara groundwater system (Adapted from Sommer *et al.* 2008).

| **Fish species** | **Common name** | **Moore River Gingin Brook and assoc. northern wetlands** | **Spear-wood linear wetlands** | **South-west GSS study area wetlands** | **East Wanneroo wetlands** | **Bennet Brook (incl. Mussel l Pool)** | **Ellen Brook (incl. EPP173)** |
|---|---|---|---|---|---|---|---|
| *Galaxiella nigrostriata* | black-striped minnow | x | | | | | x |
| *Galaxiella munda* | mud minnow | x | | | | | x |
| *Galaxias occidentalis* | eestern minnow | x | | | | x | x |
| *Tandanus bostocki* | vatfish | x | | | | | x |
| *Bostockia porosa* | nightfish | x | x | | | x | x |
| *Edelia vittata* | pygmy perch | x | x | | | x | x |
| **Estuarine** | | | | | | | |
| *Leptatherina wallacei* | | x | | | | x | |
| *Afurcagobius suppositus* | | x | | | | | x |
| *Pseudogobius olorum* | Swan River goby | x | x | x | x | x | x |
| **Introduced** | | | | | | | |
| *Carassius auratus* | | x | x | | x | x | x |
| *Cyprinus carpio* | | | | x | | x | |
| *Gambusia holbrooki* | gambusia | x | x | x | x | x | x |
| *Geophagus* sp. (*Geophagus brasiliensis*?) | introduced cichlid | | | | | x | |

## Wetland communities

One important level of biological organisation is that of the community, including populations of species occurring together in conspicuous or recognisable spatial or temporal patterns. For the Gnangara groundwater system, many of these assemblages are idiosyncratic of the area, or otherwise important for biodiversity conservation purposes.

## *Threatened ecological communities*

Seven threatened ecological communities in the GSS study area are considered to be wetland communities, including:

- *Banskia attenuata* woodland over species-rich dense shrubland (community type 20a as described by Gibson *et al.* 1994)
- Aquatic root mat community of caves of Swan Coastal Plain (Yanchep Caves)
- Communities of tumulus springs (organic mound springs, Swan Coastal Plain)
- Woodlands over sedgelands in Holocene dune swales of the southern Swan Coastal Plain (community type 19b as described by Gibson *et al.* 1994)
- Herb rich saline shrublands in clay pans (community type 7 as described by Gibson *et al.* 1994)
- Forests and woodlands of deep seasonal wetlands of Swan Coastal Plain (community type 15 as described by Gibson *et al.* 1994)
- Perth to Gingin Ironstone Association (Northern Ironstones).

More information on communities of a wetland nature is given below (adapted from Froend *et al.* 2004) and in Chapter 2. *Banksia attenuata* woodland over species rich dense shrublands (community type 20a), Aquatic root mat community of caves (Yanchep Caves) and Communities of tumulus springs are discussed further in the sections on phreatophytic communities and cave and aquifer ecosystems below.

### Sedgelands in Holocene dune swales of the southern Swan Coastal Plain (community type 19b described by Gibson *et al*. 1994)

This critically endangered wetland community type occurs within the Wanneroo Linear Wetlands west of Lake Wilgarup and north of Pipidinny Swamp. A species poor community, this threatened ecological community is dominated by *Lepidosperma longitudinale*, *Isolepis nodosa* and *Muehlenbeckia teretifolia*.

### Herb rich saline shrublands in claypans (community type 7 described by Gibson *et al* 1994)

Community type 7 is described as a mosaic of structural types ranging from open herbs, through dense heath to low woodland (Gibson *et al.* 1994). It is a species rich community occurring on heavy clay soils that are generally inundated from winter to mid summer. Typical species include *Melaleuca viminea* and *Centrolepis aristata*. This community occurs in a number of locations near the north-eastern boundary of the Gnangara study area, including sites around Lake Bambun and Bullsbrook and to the north of the study area at Lake Muckenburra. It is listed as Vulnerable.

### Forest and woodlands of deep seasonal wetlands of the Swan Coastal Plain (community type 15 described by Gibson *et al.* 1994)

Community type 15 occurs on alluvial sediments that are inundated for long periods (Gibson *et al.* 1994). *Melaleuca rhaphiophylla* and *Casuarina obesa* dominate this species poor community that has been listed as Vulnerable. Within the GSS study area one occurrence of this community type is located at Lake Bambun north of the Yeal Nature Reserve, near the south-east boundary north-east of Lexia.

### Perth to Gingin Ironstone Community (Northern Ironstone)

This critically endangered community occurs on seasonally inundated ironstone and heavy clay soils in a low area adjacent to a peak in the groundwater mound (English and Blyth 2000). The only known occurrences of this community are in the east of the Gnangara study area on land adjacent to the Gingin airfield. Typical and common species include *Melaleuca viminea*, *Grevillea curviloba* subsp. *incurva* and *Kunzea* aff. *recurva* (English and Blyth 2000).

## *Phreatophytic (groundwater dependent)* **Banksia** *woodland communities*

Woody phreatophytes are trees and shrubs that have been shown to be dependent on groundwater for their water requirements, and these types of plant species tend to dominate in Mediterranean ecosystems overlying shallow transient aquifers (Zencich *et al.* 2002). *Banksia* woodland communities on the Gnangara groundwater system are phreatophytic and they have been subject to considerable research to determine their water requirements

– in particular, the degree to which species show groundwater dependency. The majority of this work has used the proximity of a shallow, unconfined aquifer to plant rhizospheres as an inferential measure of potential groundwater dependency (Groom *et al.* 2001). Zencich *et al*. (2002) showed that both *Banksia attenuata* and *B. ilicifolia* were phreatophytic because they derived some of their water from groundwater throughout the dry-wet cycle. The relative importance of groundwater for the survival of any species in a *Banksia* woodland community is influenced by the species' depth of roots, where the individuals of species occur topographically, when they derive water from groundwater in seasonal terms, and in terms of their life history and age, and whether the species has the genetic capacity to adapt to different water levels (either in their own life, or in subsequent generations). For instance Groom et al. (2000) examined the impact of groundwater drawdown (predominantly from pumping) on *Banksia* woodland community species, describing a loss of adults of between 20% and 80% for overstorey and up to 64% for understorey species.

*Banksia* woodland communities are diverse and widespread on the Gnangara groundwater system. Dodd and Griffin (1989, p. 93) concluded, 'Despite their simple structure and uniform appearance, *Banksia* woodlands are floristically rich and taxonomically diverse.' They considered that floristically they have close affinities to the kwongan regions further north. The authors listed the dominant canopy species as *Banksia attenuata* and *B. menziesii* (less frequently *Eucalyptus todtiana* and *Nuytsia floribunda*) and *B. ilicifolia* in wetter stands. However, it is on the basis of their understorey composition that most draw distinctions between different types of *Banksia* woodlands. The '*Banksia attenuata* woodland over species rich dense shrublands' (community type 20a as described in Gibson *et al*. (1994) is an endangered community distinguished by its species-rich understorey.

Groom *et al*. (2000) examined the abundance of individuals for understorey species in *Banksia* woodland along a transect where groundwater extraction was occurring, comparing it to an unaffected site. The species were classed into three categories according to the depth of their roots (and therefore the reach to the groundwater and reliance upon it). They found that deep-rooted tree and shrub species were more susceptible to water and temperature stress than shallow-rooted shrub species (Table 4.14).

Table 4.14: Rooting depths of native perennial shrub species adapted from Groom *et al.* (2000).

| **Shallow-rooted species** < 1 m | **Medium-rooted species** 1–2 m | **Deep-rooted species** > 2 m |
|---|---|---|
| *Acacia huegelii* | *Acacia barbinervis* | *Adenanthos cygnorum* |
| *Acacia pulchella* | *Beaufortia elegans* | *Calytrix flavescens* |
| *Astartea fascicularis* | *Gompholobium tomentosum* | *Eremaea pauciflora* |
| *Bossiaea eriocarpa* | *Hibbertia huegelii* | *Jacksonia floribunda* |
| *Conostephium pendulum* | *Scholtzia involucrata* | *Jacksonia sternbergiana* |
| *Daviesia physodes* | *Verticordia drummondii* | *Melaleuca seriata* |
| *Euchilopsis linearis* | *Verticordia nitens* | *Petrophile linearis* |
| *Gastrolobium capitatum* | | |
| *Hibbertia helianthemoides* | | |
| *Hibbertia spicata* | | |
| *Hibbertia subvaginata* | | |
| *Hypocalymma angustifolium* | | |
| *Leucopogon conostephioides* | | |
| *Leucopogon sprengelioides* | | |
| *Leucopogon racemulosus* | | |
| *Philotheca spicatus* | | |
| *Regelia ciliata* | | |

## *Communities in cave and aquifer ecosystems*

Extensive karst (limestone) systems occur within Yanchep National Park in the Gnangara study area. Over 400 karst features have been documented with approximately 50 known to have (or have had) permanent streams and pools (Froend *et al.* 2004). A number of these are known to contain submerged root mats from overlying, living tuart trees (*Eucalyptus gomphocephala*) and thus six of the Yanchep Caves form the aquatic root mat community of caves (English *et al.* 2003). Jasinska and Knott (2000) describe the requisite factors for the formation of aquatic root mats in caves:

> i) permanent water bodies at, ii) shallow depth, in iii) fissured cavernous rock which supports, iv) the growth of trees, and v) where, at least for part of the year, the local climate and soil structure create arid conditions forcing the trees to grow roots in cave waters in order to meet their water requirements.

These root mats are constructed of rootlets in an ecto-endomycorrhizal association with a number of fungi species. Together this constitutes a primary food source for invertebrate faunal assemblages (Jasinska and Knott 2000). The assemblage, in Yanchep caves, includes the nightfish (*Bostockia porosa*), crayfish (*Cherax quinquecarinatus*), leeches, naidid oligochaete worms, microcrustaceans (copedods and ostracods), microturbellarians (small flatworms), amphipods, isopods and water mites (Jasinska and Knott 2000). Some of these are known to be Gondwanan relicts (Blyth *et al.* 2002). The root mat assemblage of Cabaret Cave includes some 22 species of invertebrates (Jasinska and Knott 2000). The age of the Tamala limestone in which these communities have developed, mid Pleistocene, suggests that the fauna in them will be generally younger than this in origin.

At least two conspicuous features distinguish these communities from the stygofaunal assemblage in the unconfined aquifer of the Gnangara groundwater system. The first and most obvious one is the type of habitat (flowing water in a cavernous environment where root mats have developed). The second is the species richness. Halse (Bennelongia Environmental Consultants 2008) reported on the Western Australian Museum's ad-hoc stygofauna surveys which have revealed that stygofauna occur within the unconfined aquifer of the Gnangara groundwater system but species richness is surprisingly low. There were only 24 records of 11 species from a moderately extensive sampling program. This is consistent with other surveys in the south west showing within-site richness to be low (Bennelongia Environmental Consultants 2008). However, undescribed, possibly restricted, species may occur among the amphipods and isopods, that may have significant conservation value. Further survey to species level identification is required.

Tang and Knott (2008a) identified eight cyclopoid cyclopod species from recent and historical samples obtained from 12 Yanchep Caves and five Ellenbrook Valley Springs (including 6 caves and four tumulus mound springs listed as threatened ecological communities), four species of which are new to science. An undescribed species of *Australoeucyclops* was the most common species amongst 12 Yanchep cave sites although the abundance of copepod species in nearly all samples was relatively low (<15 individuals). Only an undescribed species of *Eucyclops*, found in four Ellen Brook Valley Springs and one cave location, was thought to be endemic to the Gnangara groundwater system region (GSS study area). As several of the habitats in which this species is found are currently under threat of destruction, this taxon may qualify for listing as Vulnerable under the EPBC Act 1999 (Tang and Knott 2008a). The remaining copepod taxa are relatively widespread, stygophilic forms. Six of the eight copepod species were found in caves containing tuart root mats as well as in habitats lacking roots mats, which indicate that the occurrence of most cyclopoid copepod taxa in the Yanchep Caves is not dependent on the tuart root mat system (Tang and Knott 2008a). Tang and Knott (2008b) also identified six harpacticoid copepod species from recent and historical samples obtained from eight Yanchep caves (five of which were aquatic root mat communities) and three Ellen Brook Valley Springs (two occurrences of the tumulus mound spring community). An undescribed species in the family Ameiridae was thought to be endemic to the Gnangara groundwater system region (GSS study area), occurring only in caves that contain submerged tuart root mats in the caves at Yanchep National Park, and it may qualify for listing as Endangered under the EPBC Act 1999 as the root mats have mostly dried up (Tang and Knott 2008a). The remaining harpacticoid copepod taxa were considered widespread.

Management actions to alleviate this threat include the installation of small and large scale watering systems to maintain the root mat communities. The mitigation of watering the cave systems was developed over a number of years, at a cost of $1.7M for the initial capital outlay (bore, piping system and water filtering system) and ongoing costs of $110 000 per annum (Perriam *et al.* 2008). Despite these efforts, the water systems have not prevented the targeted cave root mats and cave streams from drying up intermittently over the last few years.

In addition to having biodiversity value as described above, stygofauna may provide an ecosystem service function by assisting to maintain water quality and aquifer transmissivity. They may also be useful indicators for monitoring water quality, by responding to changes in aquifer condition ((Bennelongia Environmental Consultants 2008).

Threats to the persistence of stygofauna in the Gnangara groundwater system include:

- the lowering of water tables as a result of drying climate or sustained abstraction for drinking supplies or horticulture, leading to population decline and reduced long-term viability
- the widespread increase in nutrients and/or pollutants causing tolerant species to replace sensitive species through competitive interactions (Bennelongia Environmental Consultants 2008).

## *Directory of important wetlands in Australia*

In addition to the recognition of discrete communities, some wetlands or groundwater-dependent ecosystems themselves have been identified as being worthy of protection for their biodiversity values. The *Directory of important wetlands in Australia* has been compiled:

- to identify sites and the wetland values present in their local area, particularly in relation to regional natural resource management planning and investment
- to identify sites of importance for particular taxa, including threatened and migratory species
- as the primary data source for identifying potential Ramsar sites, and potential sites for the East Asian–Australasian shorebird site network.

To be considered nationally important, a wetland must meet at least one of the six nationally agreed criteria. The criteria cover the following areas:

- biogeographic representativeness
- important ecological or hydrological functions
- provision of animal habitat during times of vulnerability or adverse conditions
- support for more than 1% of the national population of any taxa
- support for threatened taxa or communities
- historical or cultural significance.

Seven wetland systems on the Gnangara groundwater system are recognised by this scheme. Three of them are classified as including permanent freshwater lakes (> 8 ha) (B5), and one includes permanent freshwater ponds (< 8 ha), marshes and swamps on inorganic soils with emergent vegetation waterlogged for at least most of the growing season (B9). Three are included for seasonal/intermittent freshwater ponds and marshes on inorganic soils (sloughs, potholes, seasonally flooded meadows, sedge marshes) (B10), and one includes shrub swamps (shrub-dominated freshwater marsh on inorganic soils) (B13). Five wetland systems are recognised for their freshwater swamp forest (seasonally flooded forest, wooded swamps on inorganic soils) (B14). Two of them contained peatlands (forest, shrub or open bogs) (B15) and one of them included inland, subterranean karst wetlands (B19) (See Table 4.15).

Table 4.15: The seven Gnangara groundwater system wetland systems recognised in the *Directory of important wetlands in Australia*.

| Important wetland | Identification code | Size ha | Inland water features | Inclusion criteria |
|---|---|---|---|---|
| Chandala Swamp | SWA006WA WA075 | 100 | B14 | 1, 2, 3, 4, 6 |
| Ellen Brook Swamps system | SWA007WA WA076 | 20 | B13 | 1, 3, 4, 5, 6 |
| Herdsman Lake | SWA011WA WA080 | 250 | B5, B10, B14, B15 | 2, 3, 4, 6 |
| Joondalup Lake | SWA012WA WA081 | 530 | B5 | 1, 2, 4, 6 |
| Loch McNess system | SWA016WA WA085 | 255 | B5, B9, B14, B15, B19 | 1, 3, 6 |
| Palmer Barracks, Guildford | C WA118 | 5 | B10, B14 | 1, 2 |
| RAAF Caversham | C WA120 | 30 | B10, B14 | 2, 3 |

## Protection of ecological values

In terms of priorities for conservation and reservation, different wetlands emerge as special depending on the biotic group of interest (i.e. vertebrates vs. invertebrates vs. plant communities etc.). The directory of important wetlands identifies five wetland systems that might be commonly regarded as high priority for conserving all biotic groups. There are, however, wetlands of importance that stand outside this list (most noticeably mound springs and other threatened ecological communities, wetland systems in the north-eastern portion of the GSS, and Lakes Jandabup and Nowergup).

While recognition in the directory is important, and acknowledges values reported in Table 4.15, there are no legal mechanisms to protect wetlands identified under this scheme. Protection rests with:

- the federal *Environmental Protection and Biodiversity Conservation Act 1999* where listed threatened species and communities are concerned, and where East Asian flyway migratory water birds are concerned
- the state's *Wildlife Conservation Act 1950* for declared rare flora and threatened species of fauna
- Environmental Protection Policy: Swan Coastal Plains Lakes Policy that allows for the creation of a register of wetlands to be protected (but only those wetlands as lakes greater than 1000 $m^2$ on 1 December 1991)
- a form of protection that involves Ministerial conditions associated with the extraction of water, whereby an agency regulating the extraction of water becomes obliged to monitor wetland values to ensure that the process of water extraction does not degrade

those values. This process depends on the ability of the monitoring to detect 'sole causation', something that is rather difficult to demonstrate where multiple degrading processes operate (see Chapter 5).

Under the current legal framework, if any wetland systems are going to be protected, it will tend to be the larger, more permanent and iconic ones. Others may gain protection if their wetland values are picked up under provisions related to threatened ecological communities. This suggests that most of the poorly known, more ephemeral, and more localised wetlands and groundwater-dependent ecosystems remain poorly protected on the Gnangara groundwater system.

## Discussion

Physiographic features (principally soils and landforms) allow for differentiation between wetland systems revealing discrete and unique consanguineous suites on the Gnangara groundwater system. While this is true, it is also clear that wetlands and groundwater-dependent ecosystems on the Gnangara groundwater system are characteristic of those found across the Swan Coastal Plain bioregion in general.

There is some corroborated evidence presented in this chapter to suggest that biological diversity of plants and invertebrates in wetlands on the Gnangara groundwater system is generally species rich but with low levels of local endemism. Evidence from inland fish, mammals and plants also indicates that the northern boundary of the GSS area and the Gnangara groundwater system (marked by the Moore River and Gingin Brook on the Swan Coastal Plain) may represent a biogeographic boundary, beyond which some taxa are unlikely to be found further north.

Wetlands and groundwater-dependent ecosystems on the Gnangara groundwater system are recognised for special components like rare, threatened or otherwise important species, communities and other components Table 4.15). There is also a growing awareness of functional values attributable to wetlands, as well as their role in functional landscapes and waterscapes (Table 4.15).

Our knowledge and understanding of wetland hydrology, wetland sediments and physiographic settings, vegetation communities, wetland vertebrates and invertebrates are sufficient to draw some conclusions about management needs and priorities. These understandings of wetland ecological values and biodiversity on the Gnangara groundwater system have been influenced by a combination of proximity to researchers' workplace, preferential treatments of components of wetlands according to scientific and recreational interests (i.e. sampling bias), and government obligations with respect to natural resource management. While there is no doubt that knowledge of wetlands has expanded as a result, the knowledge base is incomplete and patchy, and lacks the comprehensive features of a knowledge base which has grown from a broadly-based biogeographical survey.

There is considerable value in compiling, analysing and reviewing existing information, and at the very least this should provide a gap analysis and such a study is definitely warranted for wetland floristics where considerable information exists but in disparate sources. A similar approach undertaken by Horwitz *et al.* (in press) to review more than 20 studies of wetland invertebrates revealed priority taxa, priority wetlands and habitats as described above. They concluded by recognising the limitations of this approach.

> Interpretation of the database is limited by its nature. Presence/absence data were used to derive the measure of richness; neither abundances nor dominance of taxa were considered. Neither sampling location (i.e. habitat) within a wetland nor sampling time (i.e. seasonality) were analysed here, let alone the inter-annual variations for taxon occurrences, or changes in fauna over time due to effects of human activities. Taxa have been collected evenly and identified consistently across few wetlands only, making comparisons between wetlands more difficult. In addition, some significant taxa (most notably the Rotifera) have been poorly treated by all or most studies. Indeed, these limitations are precisely the reasons why systematic approaches using standard methodologies are preferred for making regional biodiversity assessments … and why species richness alone can be challenged as a basis for setting conservation and management priorities … These limitations strengthen the case for a systematic and thorough survey even in this wetland region, containing perhaps the most well-known wetlands in Western Australia.

For other biota compilation of existing information will be of limited value, and broadly-based surveys will be more necessary. Microbial communities in wetlands, in both water and sediments, have been neglected (indeed this is a global situation). Bacterial assemblages play an important role in wetland ecology, including the Cyanobacteria, and bacteria associated with methane, nitrogenous and sulphurous processes. The diatom assemblages of Gnangara groundwater system wetlands are poorly described, apart from one or two detailed studies noted above. Indeed the non-vascular assemblages in wetlands in general warrant a broad and comprehensive survey and should include green algae (Chlorophyta), euglenoids (Euglenophyta), diatoms (Bacillariophyta), dinoflagellates (Dinophyta (formerly Pyrrophyta)). Particular macroinvertebrate and microinvertebrate groups also need systematic survey.

Another area of need for further work is in the preparation of metadata statements for databases on hydrology, flora, fauna and sediments for wetlands and groundwater-dependent ecosystems on the Gnangara groundwater system. Spatial data will also be important: groundwater contours and to a certain extent the relationships between groundwater and surface water are represented spatially. However, significant work will be required to build upon geomorphic wetland mapping to incorporate sediments more explicitly, as well as vegetation communities and modelling for important faunal groups. Comments on the need for further studies on thresholds of change are given at the end of Chapter 5.

Table 4.16: Summary of ecological values, showing wetland value criteria from Table 4.2, descriptions of those values, and examples of those values from the Gnangara groundwater system.

| **Wetland value criteria** | **Wetland value description** | **Examples from the Gnangara groundwater system** |
|---|---|---|
| Intrinsic value | A species or population’s right to exist irrespective of humanity | All life |
| Biodiversity contribution | A species or population is identified and described as being ‘Rare’ | *Galaxiella nigrostriata* is rare (uncommon) on the Gnangara groundwater system |
| | Key species: meeting a critical need for other organisms within the ecosystem | *Baumea articulata* provides wetland habitat, and litter can dominate the accumulation of organics in sediment |
| | Threatened, vulnerable or endangered species or population | Western swamp tortoise is critically endangered |
| | An assemblage of taxa that together perform a special ecological function, is unique due to assemblage or diversity or representativeness | Root mat faunal assemblage in caves is unique. Perth–Gingin Ironstone communities are unique |
| | Threatened ecological community: an assemblage of taxa that is threatened in some way. | Any one of 10 threatened ecological communities described in this chapter (i.e. tumulus springs) |

| Wetland value criteria | Wetland value description | Examples from the Gnangara groundwater system |
|---|---|---|
| Instrumental | A significant population or species that can have value resulting from its contribution to education or scientific research or raises awareness for environmental management | All waterbirds draw public attention (bird watchers) and government commitment (treaties) |
| | A significant population species or assemblage that can have value for the contribution to scientific management | Macroinvertebrates, diatoms or plants: bio-indicators currently used for monitoring flowing or non-flowing waters |
| | A species or wetland component may have value as a food source or resource | Peat and diatomaceous earth for horticulture |
| | A species or component is valued for recreational use | Clear open water for canoeing, sailing, skiing (Lake Gnangara, Lake Monger, in previous decades for both) |
| Indigenous/ cultural | Species or population is valued for instrumental or non-instrumental (iconic, aesthetic, spiritual) reasons to a particular group of people | Nyoongar use of turtle as food item |
| Maintenance of key ecological processes | Maintenance of nutrient cycling – mainly nitrogen and phosphorus – associated with maintaining productivity | Sedimentation processes (floc formation) lock up nitrogen and phosphorus that builds up over longer time frames, avoiding short-term effects of eutrophication |
| | Maintenance of soil biogeochemistry; biological, chemical and geological processes that have relational interactions to give soils a unique suite of properties. If this balance is disturbed the buffering capacity of the soil is decreased | Anaerobic soils perpetuated by decomposition of organic matter in sediment maintains iron and sulphur in the inert form of pyrite, preventing the mobilisation of heavy metals under oxidising conditions. |
| | Maintenance of decomposition processes and organisms allowing for the mobilisation of organic carbon through a detrital food web, or storing carbon in the soil | Dense canopy cover maintains cool moist sediments. Accumulation of litter fall, organic material as peat, maintaining moisture during infrequent periods of drought |
| Maintaining and influencing water quality | Water quality is maintained via biofiltration and filtration through sediment, and preventing mobilisation of sediment. Water quality is also influenced by leaching from sediment. | Littoral vegetation communities in and around all wetlands are a key factor in this wetland function. Biogenic and diagenic processes in all wetland sediments, particularly where groundwater flow through exists. |
| Provision of water resource | Provision of flood mitigation and recharge or discharge as a value that wetlands provide | Wetlands across the GSS study area function as compensation basins in times of water abundance, and focal points for groundwater recharge at other times. Evidence is emerging for the role that wetlands in the north-eastern part of the Gnangara groundwater system play in recharge of not just unconfined aquifer but also the deeper more confined aquifers of the Swan Coastal Plain. |
| Connectivity | Where wetlands are linked to important ecotypes – such as national forests, or to other wetlands forming a waterscape or watershed | North-eastern Gnangara groundwater system wetlands remain comparatively well connected in hydrological terms |
| Representativ eness | The wetland is or possesses features (geological, pedological, geomorphological or biological) typical of the natural system to which it belongs, or belonged (in the case of wetlands that previously existed) | The presence and functioning of the 'spring' into Loch McNess is representative of such discharges which were more widespread in the past. Tumulus springs where intact and undisturbed represent those that were localised but more numerous in the past. |

| Wetland value criteria | Wetland value description | Examples from the Gnangara groundwater system |
|---|---|---|
| Uniqueness | Rarity: Where the system demonstrates low numbers, restricted distribution or small areas of specific feature. The features may be rare, unusual, unique, outstanding or symbolic of natural or ethnographic environment (can also be cataclysmic forces). | Yanchep caves and their faunal assemblages are unique to the Gnangara groundwater system. All consanguineous suites by definition are statements of unique systems to the Gnangara groundwater system. |
| | Diversity: Refers to biological or physiological features such as vegetation structure, landforms and complexity of arrangements | Loch McNess and Yonderup Lake, Lake Jandabup, Nowergup Lake and Twin Swamps as significant for the invertebrate richness. |
| | Unique use: Where the wetland is relied upon by fauna (refuge, breeding, nesting, migration) | Extreme importance of winter wet swamps on the Swan Coastal Plain for breeding Pacific black duck and grey teal in south-western Australia. Great egret and ibis breeding colonies on Swan Coastal Plain important in south-west Western Australian regional context. *Galaxiella nigrostriata* is totally reliant on wetland EPP 173 for its continued survival on the Gnangara groundwater system. |
| Habitat values | Naturalness: Refers to the wetland state, or degree of disturbance and degradation of the system. Measure of habitat resilience. | Mid-lower reaches of Quin Brook appear to have darkly stained waters, elevated pH, organic sediments that are comparative undamaged by fire, low levels of weed infestation and no introduced fish, all signs of naturalness and resilience. |
| | Sanctuary: The importance of the habitat to act as a sanctuary to native flora and fauna, supporting flora and fauna (i.e. even if the fauna or flora is not rare, endangered or threatened) | Deeper more permanent wetlands during drought, like Lake Nowergup, Loch McNess. Organic rich sediments maintain moisture during seasonal drought allowing biota to seek refuge during seasonal extremes. |
| Waterscape ecology and ecological integrity | The ability of a wetland to recover from a disturbance or suite of disturbances. This can be the system's ability to absorb seasonal changes (drought or flooding), outside disturbances (fire, anthropological) or internal changes compounded by external disturbance (algal bloom after flooding). | Some evidence exists to show that wetland ecologies recover from extremes of seasonal variation (drought), eutrophication (Lake Monger), acidification (Lake Jandabup) and that with sufficient opportunities these abilities exist (see Chapter 6). Elsewhere wetland ecology recovery has not occurred because hydrological regimes have not been or cannot be reinstated. More study is required on thresholds of change. |
| Indigenous and other cultural values | Aboriginal significance: Wetland value associated with current and historical Aboriginal use and value | Many wetlands are recognised for their significance to Indigenous peoples (see McDonald *et al*. 2005 for details). |
| | Non-consumptive water use: cultural wetland values | Immense scientific, educational (especially secondary and tertiary education institutions), historical (cultural artefacts in and around freshwater sources), and recreational (e.g. bird watching), values of Gnangara groundwater system wetlands |
| | Wetlands as part of a sense of place | The contributions that wetlands make to the inhabitants of the Perth metropolitan area and surrounds |

In summary there are many definitions of what constitutes a ‘wetland’, including non-tidal surface water systems, other groundwater-dependent ecosystems, communities and components such as caves, stygobiota, phreatophytic vegetation, and areas of groundwater discharge including coastal areas.

Values of wetlands and groundwater-dependent ecosystems have been described as belonging to either of three broad categories:

- biotic and abiotic values
- wetland function values
- landscape and waterscape values (values of wetlands as part of larger systems).

Monitoring of ground and surface water levels across the Gnangara groundwater system has shown that wetlands are drying earlier in the season and are remaining drier for longer periods. Wetlands on the Gnangara groundwater system have a relatively low level of endemism and rare invertebrate taxa. Rare or locally endemic invertebrate taxa appear to be associated with the rare wetland types such as caves and tumulus springs. For example, four cyclopoid cyclopod and one harpacticoid copepod species, which are new to science and two of which are endemic to the GSS study area, were obtained from Yanchep Caves and Ellen Brook Valley Springs.

Three well-surveyed occurrences of Mound springs in the GSS study area have found that they contain a plant species that are disjunct from, or at the northern ends of, their southern ranges.

Strategic but limited macroinvertebrate sampling of three wetlands was carried out in 2008 for the Gnangara Sustainability Strategy. These three wetlands have now been added to the Gnangara groundwater system Macroinvertebrate Monitoring Program (EPA Section 46) and may be monitored twice yearly.

Drying of permanently inundated wetland sediments can have irrevocable consequences such as acidification, eutrophication or irreversible changes to sediment-water relationships.

# Recommendations

In order to improve understanding of wetlands, conservation planning and priorities it is recommended that:

- specific management attention or more detailed surveying be carried out in the north of the GSS study area towards Moore River–Gingin Brook to search for floral, faunal and inland aquatic faunal elements that may be locally endemic and/or regionally extinct.
- the newly discovered, regionally endemic harpacticoid and cyclopoid copepod species of Yanchep Caves and Ellen Brook Valley Springs should be considered for listing as threatened species under the EPBC Act 1999.
- trend data showing the patterns of wetland use by waterbirds during wetter years and drier years, and long-term trends, be obtained for the GSS study area.
- a systematic survey be conducted throughout the GSS study area for particular macroinvertebrate, microinvertebrate, and microbial groups.
- spatial data and metadata statements for databases covering wetland and groundwater-dependent ecosystems be prepared.
- high management priority be given to any wetland harbouring populations of native freshwater fish, and that these be monitored regularly to limit the establishment of invasive species.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter Five: Wetlands – Changes, Losses and Gains


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Our environment, our future

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Forest Products Commission Western Australia

# 5. WETLANDS – CHANGES, LOSSES AND GAINS

## Key points

- Wetlands of the Swan Coastal Plain have experienced periods of higher water levels and lower water levels over the last 10 000 years, but also in more recent times. Wetlands have undergone many changes over the last 200 years, including drainage of low-lying areas on the Swan Coastal Plain, infilling of wetlands, groundwater abstraction, and a more recent decline in rainfall. These factors have produced current groundwater declines across the GSS study area and associated declines in water levels within the wetlands.
- While there has been a history of wetland loss in the GSS study area since European settlement, more recently there has been an increase in the understanding and recognition of wetlands and wetland functioning as ecologically and socially important.
- Wetlands are dynamic systems that are influenced by geography and climate. Superimposed on, or overriding, these influences are anthropogenic effects.
- Processes that are involved in the degradation and loss of wetlands and groundwater-dependent ecosystems include:
  - altered hydrological regimes
  - land use change
  - nutrient enrichment
  - acidification
  - the impact of fire
- Hydrological changes on the Swan Coastal Plain is therefore *the* underlying cause of wetland degradation and loss, but this is driven by several processes, mostly anthropogenic: climate variability patterns of land-use change, patterns of water regulation, patterns of groundwater extraction, water infrastructure and distal societal drivers (e.g. lack of adequate water pricing).
- Seven threatened ecological communities in the GSS study area are either wetland in nature or are groundwater-dependent ecosystems. Threats to these communities include altered hydrological regimes, clearing, fire, disease, climate change and invasive weeds.

- Management methods for restoring wetland systems, including groundwater-dependent wetland systems, or management methods for preserving them, include shutting off nearby extraction bores to reduce local drawdown effects, artificially supplementing water (using either recycled water or pumped groundwater), and the rehabilitation (of mostly inner urban) wetlands from the effects of eutrophication and habitat alteration.

## Introduction

Wetlands have unquestionable value as remnants of a once more widespread, connected coastal wetland system. Included amongst them are nature reserves, wetlands of national importance, and wetlands in national parks, regional parks, recreational reserves, or state forest. While representatives of most wetland types remain in the GSS study area, most of our knowledge of these wetlands comes from monitoring work (or limited survey work) conducted in the larger, more iconic and usually more permanent wetlands. In addition, our understandings of wetland functioning have improved, and wetlands are regarded by modern Perth residents as an important part of the landscape.

Our present understanding of wetlands of the Swan Coastal Plain are that they have undergone changes over the last two hundred years, starting with the alienation of the land and water from Indigenous Nyungar stewardship (Marchant 1973; O'Connor *et al.* 1989), followed by the damming of some of the rivers flowing onto the plains, and the drainage of low-lying areas to allow for building, for agriculture and routes of movement during wet periods (Bradby 1997) characterised the 1800s. Concomitant with these changes were altered fire regimes and the onset of clearing of native vegetation (both of which continue to this day). These changes were supplemented in the 1900s by further drainage, wetland infilling, successive periods of rapid urban expansion, groundwater abstraction, extensive plantings of non-native pine trees, and more latterly a decline in rainfall. Together these factors have resulted in groundwater declines across the Mound, and associated declines in water levels within wetlands.

These wetland management issues raise an interesting series of questions. Is there evidence to suggest that the wetland water levels experienced now have occurred in the geological or historical past? How has our behaviour in and around wetlands influenced their current condition? This chapter outlines the history and processes that have led to these wetland losses and gains, and wetland change in the GSS study area.

## Palaeohistory, palaeoclimate and wetlands

Wetlands on the Swan Coastal Plain are said to have developed as aeolian depressions in dune systems, and alluvial systems closer to the Darling Scarp, formed east to west from the Pleistocene to early Holocene (where they continue to form) (Semeniuk 2007; Semeniuk 1995).

Periods of sea levels similar to those at present in wetter phases have allowed for the formation of dunes and interdunal depressions and subsequent expansion of these habitats in a seaward direction. However, the development of the Swan Coastal Plain has been punctuated by aridity usually associated with glacial maxima (and lowest sea levels and therefore most expansive plains; see Tapsell *et al.* 2003), and periods of even higher sea levels (i.e. in the late Pleistocene 150 000 to 130 000 years BP which would have inundated most of what is now the Swan Coastal Plain (Kendrick *et al.* 1991)). These phases have diminished the likelihood of long-term persistence of surface dwelling freshwater flora and fauna.

The wetland sediments and biotas that we see today have developed from when post-glacial rising sea levels caused regionally rising of groundwater tables, which eventually stabilised, and from lowlands and depressions in the landscape that were close in elevation to the present watertable that became inundated or waterlogged (Semeniuk *et al.* 2006). Therefore, the wetlands of the Swan Coastal Plain, including those in the GSS study area, are the product of relatively recent, geologically speaking, formation and expansion of wetland habitat allowing for colonisation of aquatic biota.

Attempts to reconstruct the palaeoclimate of, and water level fluctuations in, wetlands of the Swan Coastal Plain during the Holocene have yielded divergent conclusions. Across a variety of techniques, researchers have relied on the stratigraphic (sedimentary layers) record of wetlands since these depositional layers are inferred to be sequential over time, where dating can be performed. Authors have used pollen counts (Churchill 1968; Pickett

1997), mollusc, ostracod and foraminifer fossils (i.e. Yassini and Kendrick 1988), diatom fossils (McHugh 2004), calcrete deposition (Semeniuk 1986) and lithology to infer climatic variability. Few generalisations can be made, perhaps because of the varied locations of studies across the Swan Coastal Plain (e.g. different wetland suites), perhaps a function of the different techniques used, or perhaps both. There does seem to be agreement that a marine transgressive phase occurred mid Holocene where sea levels were 0.5–0.9 m higher than present (Yassini and Kendrick 1988).

For the climate of the mid Holocene, Semeniuk (1986) describes a relatively stable semi-arid to sub-humid climate prevailing in the south-west from ca. 7000 years BP to 3500 years BP. On the other hand, Pickett (1997) reported that groundwater and lake levels were generally higher than present between 8000 years BP and 4000 years BP. Churchill (1968) came to the conclusion that the climate was generally wetter between 7000 years BP and 4500 years BP, and Backhouse (1993) found a wetter period between 7500 years BP and 6600 years BP (preceding the marine transgression). According to McHugh (2004), at around 8000 years BP lake rejuvenation commenced due to a warming climate and elevated moisture regime about the time of the marine transgression, with evidence for shallow and fluctuating water levels (0–2 m range). At around 5900 years BP, lake water levels, although still fluctuating, had risen (to ~ 4m average depth). McHugh (2004) estimates the average rainfall of the period to be 10% to 15% above current levels and that these levels persisted until 4000 years BP.

Regarding climate, one school of thought says that drier conditions after about 4000 years BP (Backhouse 1993; Pickett 1997). Other schools of thought suggest that establishment of the modern hydrological regime has occurred in the period following about 4100 years BP (Yassini and Kendrick 1988), that the present humid climate experienced in the region today has persisted for the last 2800 years (Semeniuk 1986), or that wetter periods have occurred 2500 years BP – 1300 years BP and from 500 years BP till the present (Churchill 1968).

# A recent history of wetlands: colonisation north of the Swan River

When the site for Perth was chosen those areas of the Swan Coastal Plain now known as Northbridge, North Perth, East Perth, Highgate and Leederville were characterised by a chain of mostly interconnecting swamps and wetlands (Giblett 1996, p.127). Many of these wetlands linked together so that during winter floods, natural drainage lines carried waters from as far west as Herdsman Lake to discharge through Clause's Brook (now Claise Brook) into the Swan River (Bekle 1981). Among the reasons Captain James Stirling chose the site for the new Swan River Colony was the strategic placement of the site between the new port town of Fremantle and the fertile alluvial soils immediately to the north-east (Swan Valley). Plentiful supplies of timber, stone, lime, clay and rushes were available for building, as well as abundant freshwater (Markey 1979, p.349). However, it was soon very apparent that the local wetlands were not reliable sources for water due to annual decreases of water levels, and many settlers were forced to extract groundwater from shallow wells and/or collect water in water tanks (Lund and Martin 1997).

Early on, drainage ditches were used to decrease the area occupied by wetlands, to reduce the negative effects the wetlands had on expansion of the new township and to provide fertile soil for food production (Lund and Martin 1997). One indication of the colonial government's disinterest in Perth's wetlands was the granting to Samuel Kingsford in 1833 of perpetual rights to the waters between what is now Lake Monger and Claisebrook, including the option to use the waters of Lake Monger and Herdsman's Lake for five years, plus the use of as much land as required to connect the lakes (Hasluck and Bray 1930, p 351). At that time, small parcels of land surrounding Herdsman Lake and Lake Monger were traded off to workmen who were able to build and maintain drainage ditches (Cooper and McDonald 1999).

Many European settlers found the local wetlands alien, barren, lifeless and dangerous to human health (in comparison to those left behind). One newspaper article of the time described Lake Monger as flat, half burnt, dried up and devoid of even birdlife (Webb 1847). This negative view of the swamps persisted so that in 1869 the Inspector of Nuisances reported that Perth had 'witnessed a great deal of sickness with many cases of local fever occurring, some of them fatal' (Stannage 1979, p.162), and in 1873 the

Reverend Meadowcroft warned that if effectual drainage was not adopted quickly then 'the city would probably be visited by some dire epidemic' (Stannage 1979, p.169). Indeed the placement of sewage pits proximal to wetlands and sources of water undoubtedly contributed to the spread of typhoid and paratyphoid, with over three thousand cases of typhoid in Perth 1895-1899.

Early engineers and town planners struggled to understand the hydrology of the region, including the many streams that drained to either side of the ridge dissecting Perth, associated wetlands and their connection to each other and groundwater (Balla 1994). As a consequence flooding remained a significant problem in the early years (Markey 1979, p.351). It is hardly surprising that the swamps north of Perth town site would become loathed by many settlers, seen only as obstacles in the path of the town's expansion and the breeding grounds of mosquitoes and other vermin (Markey 1979, p.351).

Following a rapid (five-fold) increase in Perth's population between 1850 and 1884 (largely due to the introduction of convict labour) a correspondingly large increase in area taken up for market gardens occurred (Giblett 1996, p.132). These reclaimed wetland areas supported a thriving food supply industry due to the proximity of the new market gardens to the groundwater. Many wetland areas were reclaimed or reduced in size as the area taken up by the colony stretched out towards the foothills in the east and the coast in the west (Giblett 1996, p.132). By 1870 horticultural and agricultural landholdings stretched further to the north and west and included Little and Big Carine Swamps, Lake Karrinyup and Careniup Swamp.

The establishment of the Perth Roads Board in 1871 helped urban expansion to extend north to Lake Pinjar and east to Lake Gnangara (Cooper and McDonald 1999). While funds were short at the time, minor roads fanning out from Wanneroo Road enabled the dairy and market garden industries to set up around Lakes Joondalup, Jandabup, Pinjar and Nowergup by the late 1880s (Cooper and McDonald 1999). Around this time the first speculation driven urban boom began with small allotments taking over many of the larger properties as far north as Little Carine Swamp, west to the coast and encompassing the wetland areas. However, those farmers who were able to continue profiting from their lands continued to farm the lakes (Cooper and McDonald 1999) with their more fertile sediments.

Urbanisation of the region continued unabated with the state's population reaching 400 000 by 1929 with almost half of the population living close to Perth (Jarvis 1986, p.50). The introduction of the motor car in the early 1900s (Inglis 1988, p.480), expanding public transport systems and rapid industrial growth led to the introduction of the Metropolitan Region Scheme in 1963, as a means of controlling development of the city and suburbs (Jarvis 1986, p.51). Over the same period rainfall increased, resulting in expansion of lakes and forcing many horticulture enterprises to drain, or abandon their allotments.

In the eleven years prior to 1966, Riggert (1966) found that 49% of the estimated wetland area or least 150 500 acres (61 000 ha) of wetland habitat on the Swan Coastal Plain between Yanchep and Rockingham had been drained for grazing, agriculture, urban, industrial and recreational uses. Dredging and contouring of wetlands and the planting of introduced grasses and trees to conform to European ideals (e.g. Hyde Park) has resulted in the so-called 'saving' of many wetlands, although ecological values have been destroyed in the process (Jennings 1996, p. 149).

Halse (1989) estimated that 70% of pre-settlement wetlands of the Swan Coast Plains had been reclaimed. When Hill *et al*. (1996) conducted a comprehensive mapping exercise between Mandurah and Wedge Island they estimated the presence of 10 000 basin and flat wetlands covering a total area of 362 253 ha or more than 25% of the total land surface. Although these figures at first appear to be high one must consider them representative of the approximately 30% of original wetlands remaining, and note that many of these wetlands are seriously degraded (Balla 1994; Davis and Froend 1999). Another reason for questioning such proportions of wetlands 'remaining' is that water levels and wetland extent have been fluctuating over the last 200 years, meaning that 'original' and 'pre-settlement' baselines are not at all clear.

Starting in the 1980s, understandings of wetland functioning, and recognition of wetlands either in their own right, or as objects of nature conservation, have increased. In 1986 the Environmental Protection Authority released its *Draft guidelines for wetland conservation in the Perth metropolitan area*, designed to assist in establishing management priorities for wetlands in the vicinity of Perth, and to protect wetland conservation values, both for wildlife and for humans. The guidelines were revised and updated in 1990 and again in

1993, culminating in a position statement by the authority in 2004 outlining significant environmental values and functions of wetlands, and principles of environmental protection for wetlands.

Public groundwater abstraction schemes established at Mirrabooka, Mirrabooka East, Wanneroo and Pinjar wellfields were established in the 1980s and 1990s, and Ministerial conditions were imposed on these to ensure wetland and other values were maintained. By 1998 public water bores near wetlands were shut off due to significant effects on adjacent wetlands. Alteration of wetland habitat, eutrophication, acidification and other anthropogenic pressures continue to require management attention by state and local government agencies.

Over recent years the minerals boom has led to the continued expansion of Perth, and reclamation of existing wetland areas has occurred, particularly for palusplains and damplands. As GSS study area wetlands become terrestrialised due to groundwater decline, arguments for retaining wetlands and groundwater-dependent ecosystems for their biodiversity values are often insufficient to withstand the pressure from land speculation and land developers. Meanwhile, residential estates recognise the aesthetic value of promoting urban life adjacent to water, and constructed wetlands have become central features of most new developments. These constructed wetlands are filled with groundwater, and they double as treatment facilities for water distribution in parks and gardens, where aeration systems facilitate the precipitation of iron (and ideally phosphorus as well). They may also play a role in the conservation of biodiversity in the GSS study area in the future.

In keeping with these modern aesthetic appreciations, wetlands in the GSS study area are popular as tourist destinations (e.g. in Whiteman Park, Yanchep National Park) and as urban areas for recreation (Lake Monger and Herdsman Lake (Tapsuwan *et al.* 2009).

These patterns of change can be characterised as continually evolving phases of our understanding of, and attitude to, wetlands (Table 5.1). The last column summarises a ‘phase’ of wetland understanding – in general, it might be argued that each phase has continued so that what we have now is an accumulation of many of them.

Table 5.1: A history of wetland loss in the GSS study area.

| Time | Description of wetland-related events and attitudes | Phase of wetland understanding |
|---|---|---|
| Pre–colonial times | Nyungar ownership. Wetlands as expressions of the Waugal. Sources of food and freshwater. | Indigenous embedded |
| Colonial establishment | Wetlands perceived as source of freshwater, instrumental in establishment of the site for Perth. | Establishment opportunity |
| First years of colony | Wetlands perceived as source of power (drains), ownership transferred, drainage important for land development. Perceived as dangerous to human health, sources of disease, obstruction to development, places of seasonal flooding. | Instrumentalist (including dangerous) |
| Mid to late 1800s | Wetlands included in subdivision and alienation. Drained wetlands become principal locations for market gardens and mixed farming: horticulture and agriculture dominate, and food supply industry expands northwards from the city. | Wetland alienation |
| Late 1800s | Roads allow northward expansion of dairy and horticulture, drains feature prominently, isolated urban areas developing. Smaller urban allotments close to Perth. Typhoid outbreaks. Lower water levels allow land use of wetland basins. | Urban and horticulture expansion |
| Late 1800s–early 1900s | Urban development obliterates some wetlands close to city; those remaining are park-like and receive urban runoff. Horticulture and agriculture continue to expand in and around wetlands of the southern and eastern Gnangara groundwater system | Early Urbanism |
| 1920s–1960s | Urban development continues to expand around peri-urban wetlands, and gradual increase in rainfall and increase in water levels in wetlands, together bring watertables up, significantly inundating wetlands, prompting calls for more drainage and urban groundwater bore installations and pumping. Infilling of urban areas consumes some of the remaining wetlands in these areas. | Urban infilling |
| 1960s–1970s | Wetlands starting to be seen as valuable, but mainly as isolated icons. Some better understanding of hydrology emerging, at least partly driven by early implementation of public groundwater abstraction schemes. | Water resource focused |
| 1980s and 1990s | Pines mature, public groundwater abstraction, urban development and horticulture continue in mixed proportions. Increased concern over groundwater levels; significant drying of iconic wetlands; wetland evaluation and assessment, and long-term monitoring implemented. Draining and infilling consumes more wetlands. Construction of urban wetlands in residential estates. | Wetland aestheticism: integration and conservation |
| Early 2000s | Climate change emerging as the principal 'problem'. Justice Wilcox reaffirms that Nyungar are a single people, recognising their belief systems (including the primacy of the Waugal). Wetland conservation entrenched. Wetland aestheticism entrenched. Some trends towards better technologies and practices in urban hydrology. | Globalism |

## Processes involved in degradation and loss of wetlands and groundwater-dependent ecosystems

As dynamic systems, changes that occur to wetlands are influenced by geography and climate, with anthropogenic effects superimposed, or overriding other influences (Arnold 1990). While direct effects of agriculture, urbanisation and construction may be obvious, more subtle indirect effects such as fluctuations in the watertable or changes in amplitude of those fluctuations, increased sedimentation, eutrophication and changes in fire regimes may alter the type or rate of change in plant succession and habitat available for fauna over the longer term.

### Altered hydrological regimes

Significant changes in climate have been noted in recent decades in many parts of the world, and in south-western Australia the most significant change has been a decrease in rainfall resulting in a stepdown decrease in surface runoff and groundwater recharge rates. For Perth and the Swan Coastal Plain, cumulative deviations from the mean rainfall analyses (Yesertener 2005) show a wet period from 1914 to 1968, and a dry period after 1969. Since the mid 1970s annual rainfall has declined 10%, with the greatest decreases noted in autumn and early winter. Although there has been little change in late winter and spring rainfall and a slight increase in summer, the region relies on good winter rainfall (Hope and Foster 2005).

It is generally accepted that rainfall and catchment runoff have the greatest impact on groundwater and wetland depth under natural conditions. A decrease in long-term rainfall is therefore reflected in groundwater level decline in the GSS study area. There are other factors also influencing long-term groundwater levels, including urbanisation and land use changes. Unprecedented urban expansion is placing groundwater resources under growing pressure as private and public extraction increase to meet domestic, industrial and commercial demand. In addition, groundwater extraction for public water supply has increased in recent years due to the decreased rainfall and surface water runoff to dams. Land use changes can also influence groundwater recharge. Clearing of vegetation, roof runoff, road runoff and importing water for lawns and gardens can increase recharge locally. Conversely, pine plantations have had the opposite effect. The plantations were established north of Perth to meet the state's demands for sawlogs, and cover an area of about 22 000 ha, or 10% of the GSS study area (Bourke 2004). The replacement of native vegetation communities with pines has been shown to have contributed to decline (Yesertener 2007) of groundwater levels in the superficial aquifer by increasing interception of rainfall, and increasing transpiration and evaporation rates. These factors interact spatially so that changes to groundwater levels over time vary over the whole of the GSS study area. In fact modelling by the Perth regional aquifer model shows this well (Figure 5.1), where predictions made for the base case scenario show significant ongoing declines over the next 23 years for the north-eastern part, and to a lesser extent the south-western part, of the mound.

Froend *et al.* (2004) gave the following description of the key changes to vegetation in response to a drying hydrological regime. Because water requirements are not being met, the vigour of individuals within a population will decline (water stress, branch dieback, reduced growth, leaf shed, chlorosis), leading to loss of individuals in the drier areas of the water availability gradient across the littoral zone of wetlands (altered distribution), or total loss of the local (within ecosystem) population (altered composition). Due to the gradients in water availability, there is the potential for extreme variability in vegetation vigour. Vegetation at the driest extreme of its distribution will always reflect the poorest (relative) vigour. As water levels gradually decline, a greater proportion of the vegetation will have poor vigour and die, with progressively less habitat for colonisation.

A gradual reduction in the water available to plants usually sees a change in distribution along the water availability gradient across the littoral zone of wetlands. This reduction is also associated with changes in plant distribution in surrounding upland areas typical of shallow depths to groundwater as well as areas of permanent water that may occur. The characteristics of change in distribution of vegetation are influenced by many factors including the water requirements of resident plant species, sources of propagules for colonisation, the magnitude and rate of water level change and the geomorphology of the wetland and surrounding areas.

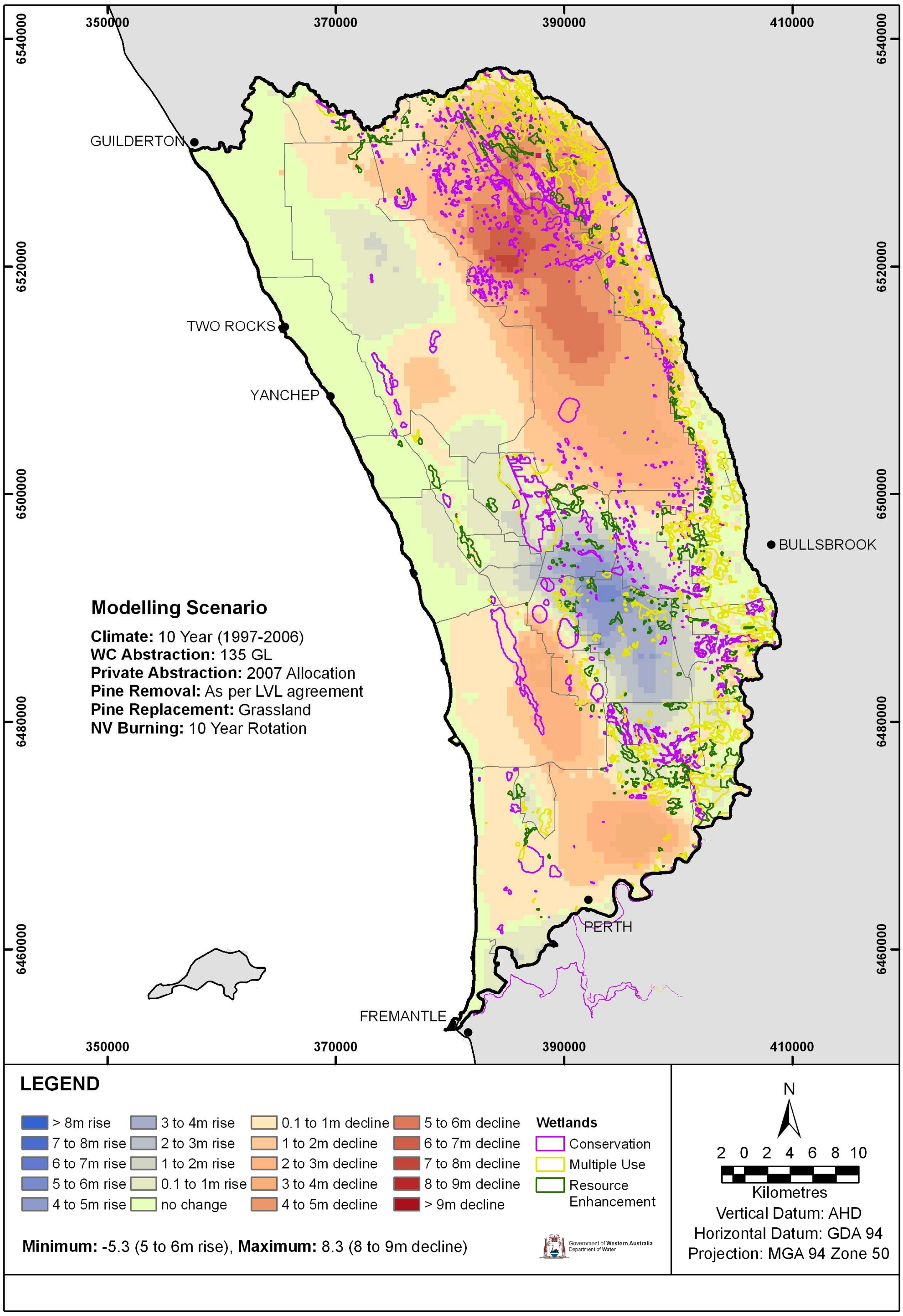


Figure 5.1: Perth regional aquifer modelling (PRAMS) base case scenario showing predicted changes in water levels should public and private extractions continue at 2007 rates, with recent rainfall declines.

Changes in vegetation composition are an important factor in the loss of native species and gain of exotic species. The factors that influence the character and rate of composition changes are the same as for distribution. Added factors include the potential for terrestrialisation as more xeric species colonise and the increase in habitat for exotics that may result from canopy decline.

For phreatophytic vegetation, research in the vicinity of criteria bores in the GSS study area give a sense of likely changes in condition. In areas of high groundwater (< 3 m), much of the remnant vegetation is likely to be dominated by species tolerant of high soil moisture (*M. rhaphiophylla, M. preissiana, E. rudis, B. littoralis* and *B. ilicifolia*), while at greater groundwater depths (3–10 m) less tolerant species may become more dominant (*B. attenuata* and *B. menziesii*). Generally, species tolerant of high soil moisture should be expected to respond most severely to groundwater declines while enduring increases. Species less tolerant of high soil moisture may respond less severely to groundwater declines yet be affected by increased levels (waterlogged). The degree of response depends on the magnitude, rate and duration of water level increase or decrease, the historic changes in water levels, the specific site conditions (stratigraphy, etc.), habitat type and species in question, the influence of disturbance impacts (fire, etc.), and finally the condition of the vegetation.

For fauna, a general principle is that the life cycles of aquatic animals tend to be associated with aquatic regimes (where they live in relation to where and when the water occurs). Accordingly, regime changes will challenge the persistence of some species locally, or improve the survival and reproduction of other species, depending on physiological and behavioural characteristics. The most obvious examples relate to species that require permanent water, like pygmy perch, or the root mat faunal assemblage in caves. If watertables in coastal plains drop then there will be an increased risk of stranding such fauna out of water (during a drought, for instance). Other more specific examples are given below, and in Table 5.2.

Wetland sediments will be affected by a drying regime. An increase in the seasonal exposure of sediments to drying will occur, changing growth rates of biota that contribute to sediment build up, and changing decomposition rates of their biomass. It may also lead to the build up of dry organic matter on the wetland basin as mesic species perish.

Historically, seasonally inundated wetlands such as Lake Mariginiup, EPP 173 and Big Carine Swamp in the GSS study area have been at greatest risk due to the longer period of sediment exposure. However, wetlands regarded previously as being permanent, such as lakes Joondalup and Jandabup (but see below), are now also drying at least partially each summer–autumn. Longer term spatial and temporal changes in sediment exposure leads to the ‘terrestrialisation’ of sediments, where repeated fire disturbance reduces the organic fraction of the sediment, and terrestrial flora (including weeds) encroach into the wetland basin (see below).

## Land use change – clearing, filling and fragmentation of wetland habitats

Clearing for urbanisation can cause watertable and wetland water levels to rise, with associated increases in ephemeral wetland areas (Arnold 1990) while excessive groundwater abstraction can reduce water levels in other areas. Clearing of the land for urbanisation and agriculture results in a rise in the watertable, increased surface runoff, higher water levels in wetlands for longer periods and associated deaths of wetland vegetation (Halse 1989). Drainage has always been, therefore, an important part of land use change on the Swan Coastal Plain, particularly during periods of increased rainfall.

Initially the land was cleared to make room for market gardens, crops or pasture. Crops were traditionally planted in areas that dried in summer but remained waterlogged during winter, reducing or eliminating the need for watering (Bekle 1981). Permanent wetlands (2.5cm - 90cm deep throughout the year) (Riggert 1966) were particularly targeted for drainage. The organic, rich soils of these wetlands were generally suitable for cropping (for a few years at least) and were often located in catchment areas so that more water was available for market gardens, again without the need for hand watering (Davis and Froend 1999). Many wetlands were affected by the removal of fringing vegetation to provide more area for vegetables and crops or to make way for landscaped areas as urban parks (Davis and Froend 1999). Other wetland types, including non-drainable areas such as seasonally flooded agricultural land, rivers and reservoirs, were less affected in the early years, because they at least provided valuable summer pasture, or had not yet come to be in the way of development or could not be economically or efficiently drained at that time (Riggert 1966).

The alteration of wetland areas through land use changes can effect subtle or dramatic changes over time. Prior to these land-use changes water levels were more reflective of rainfall patterns. Many Wanneroo wetland areas, including Lake Joondalup (Craigie Lake) and Nowergup Lake (Narago Lake) were used as watering holes on the original stock route that passed through Wanneroo (Kennealy 1994). Since colonisation and prior to recent unprecedented urban expansion, most of the Wanneroo wetlands including Lake Joondalup, Lake Goollelal, Lake Nowergup, Lake Neerabup (Pappas Swamp), Mariginiup Lake and Coogee Springs, were intensively managed, with market gardens, dairies, piggeries and vineyards being the main traditional land uses.

Oral histories cited in Kennealy (1994) suggest that in the early days of settlement many Wanneroo lakes would dry out over the summer (Lake Joondalup, Lake Goollelal, Lake Neerabup and to a lesser extent Pipidinny Swamp and Beonaddy Swamp). It is likely that wetter periods and wetland expansion from the period 1914 – 1969 resulted in more permanent wetlands, and hence the inundation of fence lines placed in these wetlands prior to this time.

Jandabup Lake (Big Dunbar Lake) has experienced a slightly different history of land-use with much of the bush on the western side of the lake remaining uncleared until the 1960s. Much of the eastern shore of the lake, however, like other wetland areas in the district, has a history of cattle grazing, dairy and pig farming, market gardens and woodcutting during the early years following settlement (Kennealy 1994). Prior to the planting of the pines, long-time residents of the area recall Lake Jandabup's water levels being reflective of rainfall patterns, often being deep enough to swim in (Kennealy 1994). More recently (and prior to artificial supplementation) lake levels have been harder to predict, 'filling up' only twice between 1940 and 1994 with the latter occurrence flooding over Gnangara Road (Kennealy 1994). A nearby wetland, Dunbar Lake displayed a similar drying regime until it dried up completely during the 1950s (Kennealy 1994).

Symptomatic of these drying trends, many of the wetlands in the East Wanneroo area and relatively high in the GSS study area are no longer inundated seasonally, and remain dry on an annual basis, such as Lake Pinjar to Lake Adams, Dunbar Lake, and the Badgerup suite. Many other wetlands have been completely subsumed, originally by agriculture and drainage, but latterly by urbanisation. Other wetlands which were once significantly inundated but are now inundated infrequently or not at all include Coogee Springs, Wilgarup, Pipidinny Swamp, and all parts of the Spearwood Linear chain of wetlands closer to the coast near Yanchep. Sumplands in the Lexia wetlands on Bassendean dunes in the south-east of the GSS study area are further examples.

## Nutrient enrichment

Wetlands are the part of the landscape where nutrients (principally carbon, nitrogen and phosphorus) generated in the catchment eventually accumulate. Here they are exchanged between sediments, water and biota, in a cyclical flux between in-solution, biological metabolic processes, biomass growth and decomposition and deposition, all moderated by the arrival of more nutrients (input) and the loss of nutrients (through outflow and fire). In general terms, the more nutrients in a wetland, the greater the rates of all of these processes, to the extent that an oversupply results in an altered ecology and diminished ecosystem services. Urban and agricultural activities are well known for providing elevated levels nutrients in short periods of time. All of these general behaviour patterns apply to the wetlands of the Swan Coastal Plain. However, these wetlands occur in landscapes and catchments with low levels of nutrients, so in the absence of urban and agricultural activity their productivity state is low.

The leaching sands of the coastal region of the Swan Coastal Plain are particularly conducive to the often rapid transfer of nutrients, that have been applied to catchments and to wetlands (McComb and Davis 1994). Upstream agricultural and horticultural practices, stormwater drains, urban gardens and municipal parklands all cause increases in nutrients and particulate material in groundwater and surface flow. This results in sedimentation and elevated nutrient levels in receiving wetlands, which in turn raises productivity.

An instructive case study for the GSS study area is Lake Monger (Lund and Davis 2000). As early as 1936 Lake Monger was identified as the breeding ground for nuisance midges. In 1961 eutrophication of the lake was so extreme that mass fish deaths occurred (albeit introduced carp) due to anoxia. This led to bird deaths from botulism. The following consequences resulted from the following factors, among other things (Bekle 1981; Town of Cambridge 2008):

- the removal of the indigenous vegetation from the catchment and the replacement of fringing vegetation with lawn and exotic trees
- widespread and frequent application of fertiliser around the lake
- the dumping of municipal waste as landfill into wetland areas
- the direct inflow of stormwater from urban areas
- spraying of hormone based herbicides to eradicate the introduced water hyacinth (*Eichhornia crassipes*).

The result was deterioration in water quality and the comprehensive destruction of the wetland habitat. Steps have since been taken to rehabilitate the lake (see below). Despite this it is a significant breeding wetland for the black swan (Brearley 2005).

A wide range of interacting and synergistic effects can affect wetlands as a result of nutrient enrichment. The development of nuisance blooms of cyanobacteria (blue-green algae) with their characteristic odours, diminishes water quality. Algal blooms are also associated with shading and/or smothering of benthic vegetation and sessile fauna, and the development and subsequent decay of large algal blooms results in deoxygenation of the water body. Waterfowl, fish and large invertebrates may suffer direct lethal effects due to both algal toxicity and periods of reduced oxygen availability (Davis and Froend 1999). Together these effects are enough for them to be a regular concern over the summer months at urban wetlands.

The release of heavy metals associated with anoxic waters, and the presence of other toxins can result in not only a threat to human health (e.g. salmonella) and birdlife (e.g. botulism caused by the bacterium *Clostridium botulinum*), but also in changes to wetland vegetation (including the loss of sensitive fringing vegetation, proliferation of invasive species, and terrestrialisation). Eutrophication can also lead to changes in faunal assemblages and the proliferation of nuisance midges and mosquitoes (Arnold 1990). Outbreaks of nuisance midges (*Polypedilum nubifer*) at Perth wetlands occupy a disproportionate amount of resources in local municipalities.

Often measures used to control problems associated with eutrophication can themselves further degrade wetlands. For instance, pesticide treatment of urban wetlands can result in the reduction of natural predators and further midge problems over the longer term (Arnold

1990). Indeed, changes to macroinvertebrate communities in Lake Joondalup may be traced to such applications at least in part. The conclusion of Davis and Froend (1999) still rings true today: 'The sources of nutrient enrichment and reasons for the occurrence of nuisance algal blooms are fairly well understood but strategies to control excessive nutrient inputs and prevent the development of nuisance algal blooms still need to be devised and implemented.'

## Acidification

Acidification in freshwater systems results from exposure and oxidation of the otherwise saturated sediments which contain reduced sulfur compounds (Baldwin *et al.* 2007). Downstream of wetlands (and their sediments), changes in the acidity of soils and shallow groundwater can occur if soils and aquifer sediments are sandy and have little or no carbonate content to provide an acid buffering capacity (Appleyard and Cook 2008). Where these processes are occurring, Appleyard and Cook (2008) suggest the following consequences:

- the leaching of aluminium, iron and organic matter and nutrients from soils can be accelerated
- eventually the base cations, calcium and magnesium, can be leached from the soil
- exchange sites in the soil profile replaced by soluble and phytotoxic aluminium and hydrogen ions
- loss of calcium, from soils in particular, can lead to a progressive decline in the health of both woodland and wetland ecosystems and the loss of plant and animal species.

In addition, oxidation of sulfidic sediments can cause mobilisation of metals from the sediments, and anoxia in the overlying water column (Sullivan *et al.* 2002).

Two exemplars of acidification on the Swan Coastal Plain are known from the wetland sediments of the GSS study area. These are the suburb of Stirling and the East Wanneroo wetlands. In the Stirling Shire, acidification and contamination of groundwater by arsenic and heavy metals has become an increasingly acute problem (Appleyard *et al.* 2004). This has been due to a reduction in rainfall, with associated decline in the watertable, over recent decades, and to disturbance of acidic peat soils, Saturated peaty sediments and acid sulfate soils are considered unsuitable as foundation material for housing, so these have been excavated and removed, dewatering has been carried out, and excavations to construct artificial lakes have been undertaken by the local government (Appleyard *et al.* 2004). A contaminated groundwater plume was detected by sampling water from domestic groundwater bores downstream of the areas where these activities occurred. Very low pH levels and elevated levels of aluminium, iron and arsenic were recorded. These were of concern for public health, management of nearby conservation category wetlands, and because of possible damage to subsurface infrastructure (Appleyard *et al.* 2004).

Three wetlands in the East Wanneroo area were chosen by Sommer and Horwitz (2001; 2009) when they sought to describe the ecological consequences of drought induced acidification. These wetlands had a long-term macroinvertebrate and water quality data set, available. The following ‘ecological state’ scenarios were considered:

- episodic/sudden acidification followed by intervention by re-inundation and reinstating anaerobia of sediments, and subsequent recovery (Lake Jandabup)
- gradual erosion of buffer capacity in the absence of intervention, leading to either an alternative permanently acidified state (as in the scenario below), or to recovery (as in the scenario above), depending on the position of the watertable; the wetland is in a transition state (Lake Mariginiup)
- buffer capacity exhausted and in a permanently acidified steady state (Lake Gnangara).

The findings are shown in Table 5.2. Sommer and Horwitz (2009) were able to characterise the effects of acidification, as well as a recovery from pH stress. They also described the characteristics of changes during the recovery process which resulted from artificial augmentation of water levels.

Table 5.2: Summary of the key aquatic macroinvertebrate responses to episodic/permanent acidification, water level decline, recovery of pH and artificial water level augmentation (Source: Sommer and Horwitz 2009).

| **Biotic response** | **Factor**<br>**Acidification** | **Water level decline** | **Recovery of pH** | **Artificial augmentation** |
|---|---|---|---|---|
| Taxa richness | - decreased summer richness; no change in spring richness<br><br>- tendency towards rarity | - decreased richness summer and spring<br><br>- tendency towards rarity | - recovery of summer richness; no change in spring richness<br><br>- decrease in rare taxa and increase in more abundant taxa | - increase summer richness; no change in spring richness<br><br>- decrease in rare taxa and increase in more abundant taxa |
| Taxa community structure | - local extinction/decrease in acid sensitive taxa (Ceinidae, Planorbidae, Cyprididae, Amphisopidae, Sphaeriidae, Caenidae, Daphnidae, Oligochaeta)<br><br>- increase in acid-tolerant taxa (mainly Ceratopogonidae, Macrothricidae, Corixidae, plus other Gnangara taxa: Dytiscidae, Chironomidaee, Hydrophilidae, Notonectidae, Aeshnidae, Lestidae, Limnichidae) | - not determinable from this study due to lack of un-acidified drought affected control | - reappearance and increased abundance of acid sensitive taxa<br><br>- decrease in acid tolerant taxa | - 'augmentation beneficiaries' increase in abundance or appear for the first time (Jandabup: Cnidaria, Ancylidae, Caenidae. Betidae, Turbellaria, Arrenuridae,, Hydrachnidae Mesoveliidae, Stratiomydae, Calanoida) |
| Functional feeding groups | - decrease in filter feeder abundance and gatherer richness | - decrease in richness and abundance of predators<br>- decrease in gather and mixed feeder abundance | - increase in filter feeder abundance and richness increase in gatherer richness | - increase in predators and parasites (mites) |

Whether induced by drought, a product of dewatering or a product of direct disturbance by excavation, acidification can have serious effects on the biological productivity of aquatic ecosystems (see review in Sommer and Horwitz 2009). These effects are:

- iron and aluminium flocs smothering vegetation and substrates, or smothering gills, restricting respiration

- the direct toxic effects of acidity (the harmful effects of $H^+$ ions)
- metal toxicity
- calcium limitation.

These result in changes in aquatic plant and algal communities, deaths of particularly sensitive species (e.g. fish, crustaceans and molluscs), lower richness and abundance and disrupted trophic structure of macroinvertebrates, and the danger of ‘seed bank’ depletion. Some invertebrate taxa are able to persist in low pH waters, including those with the ability to breathe oxygen from the surface rather than obtaining it through gills, or that have bodies that are highly chitinous and impermeable.

## Fire impacts

Although the general fire literature is extensive, comparatively little has been published on the effects of fires on wetlands, aquatic biota and water quality per se. Many plants and animals found in wetlands either require fire as part of their life history strategy, or can avoid, behaviourally or physiologically, the effects of fire. Many examples exist (including weedy species like *Typha orientalis*). Other wetland biota cannot survive the direct effects of fire, and Horwitz *et al.* (2003) suggest that such species are likely to be found in permanently wetter parts of the landscape where they have been able to evade fires over long periods of time (in ‘refuges’). Under particular circumstances, soils and sediments (such as peat) can be seriously affected by fire. Wetlands, therefore, can be something of a paradox in terms of fire management. In general terms, fire is a regular occurrence in catchments, and hence wetlands, and for some components it is needed as an important ecological disturbance. Other components need to be protected from fire. Of the various characteristics of fire, the intensity is considered to have the most significant effects on native vegetation. Intensity is usually an inverse function of fire frequency. For example, long periods between fires tends to allow accumulation of fuel loads resulting in higher intensity fires, whereas more frequent burning results in lower fuel loads and less intense fires (Underwood and Christensen 1981). However, an alternative view is that long-term accumulation of organic material in thickly vegetated shady situations keeps patches moister and therefore more likely to avoid the passage of a fire.

The potential consequences of fire on wetlands and water quality have been categorised in terms of five interrelated effects (Horwitz and Sommer 2005).

- *Catchment effects* operating through the processes of runoff and deposition, can result in mostly short-term changes to wetlands, such as elevated base cations, increased alkalinity, elevated nutrient concentration, pulses of sediment input, and greater groundwater recharge in wetlands.
- *Atmospheric effects* are also mostly short term, and include the return to the ground via rain of dissolved volatilised reactive and particulate compounds. They have a mildly acidifying and/or fertilising effect in wetlands.
- *Rehydration of burnt or overheated (organic) soils* can have either alkaline consequences, in which case the ash can act to fertilise and increase productivity in the short term, or it can have acidic consequences where acid sulfate soils are oxidised, in which case the effects are the same as described above for acidification.
- *Fire suppression* activities in a wetland, or in catchment in which a wetland is found, have different effects depending on the particular method.
- *Water movement* (taking water from a wetland for use elsewhere or dumping it into a wetland) might inappropriately translocate species, the construction of drains and runnels might result in disturbance to acid sulfate soils, and the chemicals used to suppress fires might be toxic to aquatic organisms.

All of the above have trophic consequences, particularly when considered together with the direct effects of the fire on wetland biota mentioned above.

Recent conditions of declining groundwater levels and reduced saturation of sediments such as peats that are now exposed to fires at particular times of the year, have created a particular set of issues for wetland management.

Semeniuk and Semeniuk (2005) described the sediments likely to be burnt as comprising peat, diatomaceous peat and spongolitic peat. These authors (and others) have stressed the importance of understanding wetland sediments, stratigraphy and hydrology in the management of fire, for several reasons:

- the calorific value of the sediment can be determined from the type of sediment and degree of moisture
- the annual and longer term hydrologic patterns determine when sediments are susceptible to burning

- the effects of regional groundwater decline on the exposure of different stratigraphic layers to fire
- the chemical composition of the sediment, particularly the heavy metal and metalloid, and biogenic origin, for the consequences to human health in terms of exposure to water or smoke or dust
- the altered wetland biogeochemistry post-fire, where sediments have combusted
- burning of wetland sediments results in destruction of wetland stratigraphic geoheritage.

Horwitz *et al.* (2007) reported on a study conducted during 2005–06 to investigate the likelihood that burnt, dried and oxidised pyritic soils might lead to acidification in either groundwater or surface water following an intense fire in the Yanchep National Park. Three wetlands (Yonderup, Pipidinny and Wilgarup) were examined seasonally, to test the following hypothesis: 'where organic rich pyritic soils are exposed to fire, the oxidation of iron sulfides will not be severe enough to override the buffering effects of both the soil itself, the resultant ash or their rehydration.'

The results (and/or design, sampling strategy) were insufficient to reject this hypothesis for surface waters – no before-fire and after-fire effects were noted. For groundwater plume sampling and in situ groundwater sampling, results were highly variable between replicates of the same wetland and overall the hypothesis could not be rejected. However, some sites where wetland sediments were severely burnt showed an in situ, and a downstream acidic metal-rich plume characterised by elevated ammonia (up to 3.9 mg N/L), Al (up to 190 mg/L), Fe (up to 290 mg/L), phosphorus (up to 0.5 mg/L), and higher levels of Co, Cu, Pb, Mn and Ni, than found elsewhere in their study. The authors concluded that these results are sufficient to suggest that the hypothesis might be rejected with an appropriate design and sampling strategy. Such a study is now underway (D. Blake, Edith Cowan University, pers. comm.).

There has been recent discussion among water resource management agencies regarding the management of groundwater recharge through fire. The concept is that increasing the frequency of controlled burns within *Banksia* woodland will reduce evapotranspiration and allow increased recharge of shallow aquifers. Increased frequency of controlled burns must also be evaluated in terms of the likely consequences for wetland ecology. For instance, there may also be a loss of heterogeneity in vegetation as species that are intolerant of frequent fire are gradually lost and age class structures change to reflect predominately post-fire regeneration stages. Burning around wetlands more often may increase the likelihood of fire entering wetland sediments, particularly if they are in a relatively dry state. Therefore, such a ‘water resource management by fire’ regime may conflict with current conservation-based concepts of desired future states for the wetland ecosystems.

## Threatened wetland and groundwater dependent ecological communities

Seven threatened ecological communities are either wetland in nature or are considered to be phreatophytic and therefore representative of groundwater-dependent ecosystems (see Chapter 4). Table 5.3 lists the main threats to these. Despite the high number of groundwater dependent threatened ecological communities within the GSS study area are groundwater dependent, little is known about how these communities might respond to altered hydrological regimes from lowering local groundwater levels and climate variability.

Table 5.3: Summary of threats to threatened ecological communities in the GSS study area that are wetland in nature or are groundwater dependent.

| **Threatened ecological community** | **Description of threats** |
|---|---|
| Perth to Gingin ironstone association (Northern ironstones) | Altered hydrological regimes, clearing, fire and invasive weeds (see Chapter 2). |
| Forests and woodlands of deep seasonal wetlands of Swan Coastal Plain (community type 15; Gibson *et al.*1994) | These occur on low lying seasonally inundated flats with poor drainage that results from an impeding clay layer. The effects of altered hydrology and climate change in particular ,on the community, are not known (DEC 2008; Gibson *et al.* 1994). |
| Communities of tumulus springs (Organic mound springs, of Swan Coastal Plain) | Altered hydrological regimes, fire and clearing (see Chapters 2 and 4). |

| Threatened ecological community | Description of threats |
| --- | --- |
| *Banksia attenuata* woodland over species rich dense shrublands (community type 20a; Gibson *et al.* 1994) | Altered hydrological regimes, climate change, *P. cinnamomi*, fire, invasive weeds and clearing threaten this community. *Banksia* woodlands are considered to be 'phreatophytic' or groundwater dependent (see Chapters 2 and 4). |
| Aquatic root mat community of caves of Swan Coastal Plain (Yanchep caves) | Altered hydrological regimes and destruction of tuart trees by fire (see Chapters 2 and 4). |
| Woodlands over sedgelands in Holocene dune swales of the southern Swan Coastal Plain (community type 19b; Gibson *et al.* 1994) | This critically endangered community, which has one occurrence, is on the Quindalup Dune system in a flat seasonal wetland (English *et al.* 2002). It is not known what effects changes in rainfall or watertable levels may have on this community, although as the community is a wetland severe hydrological changes would be expected to be highly detrimental. Important known threats also include too frequent fire and invasive weeds. (DEC 2008). |
| Herb rich saline shrublands in clay pans (community type 7; Gibson *et al.* 1994) | This community is on low lying, seasonally inundated flats over an impeding clay layer. Reduced rainfall and altered surface flow from climate change and/or alteration of the height of the local watertable may change the duration or depth of ponding. The effects of altered hydrology on the community are not known. |

## Hydrology – biology – society linkages

Our desire to separate causes of environmental degradation ('threats') is problematic when complex systemic factors are at play (Horwitz *et al.* 2008), and as the earlier descriptions of the historical processes associated with wetland loss demonstrate. Hydrological change on the Swan Coastal Plain is another example where underlying or distal causation must include all of the following, to various degrees:

1. climate variability: in particular inter-annual variability in the pattern of rainfall, temperature regimes (air and water), cloud cover, wind direction and strength, and potential evaporation rates
2. patterns of land-use change: shifts from woodlands or heath to agriculture and horticulture, in some places followed by urban development (Table 5.1). For urban areas, land-use patterns change with age as suburbs become more vegetated. Each

of these shifts in land use is accompanied by different patterns of surface runoff and groundwater recharge from rainfall, and different types of microclimatic feedbacks to the macroclimate.

3. patterns of water regulation: damming of catchments has substantially reduced flow from the Darling Scarp onto the Swan Coastal Plain and nearly 120 years of draining has virtually eliminated previously extensive areas of inundation of the Swan Coastal Plain
4. patterns of groundwater extraction: intensive localised extractions for horticulture, agriculture and domestic water supplies plus the extraction of water from private residential bores
5. water infrastructure: patterns of groundwater recharge and behaviour in terms of overconsumptive water use are influenced by reticulated distribution of potable water through the Integrated Water Supply Scheme, and collection in ‘wastewater’ facilities before most of it is discharged offshore. Stormwater runoff is also ‘collected’ resulting in similar discharge or concentrated recharge in sumps or wetlands (creating a local mounding effect in places).
6. distal societal drivers: the macroeconomy, lack of adequate water pricing, political ideologies and population growth scenarios influence all of the above. Similarly, all phases since colonisation have involved different forms of wetland alienation as changing attitudes to wetlands, from Nyungar times, through early colonial times, expansion times and now more urban times (see Table 5.1).

If hydrological change is considered as *the* underlying feature of wetland change, this allows us to pinpoint our influences on the biodiversity of wetlands and groundwater-dependent ecosystems. Requirements for water, as categorised in Chapter 4, will respond differently to scenarios of declining groundwater levels, decreasing extent and duration of inundation of surface waters, and decreasing rainfall.

Table 5.4 shows broad classes of linkages between hydrology and biology for inland aquatic and terrestrial species, communities and/or ecosystems in the GSS study area. For each linkage it gives:

- a prognosis for the species, communities and/or ecosystems under drying scenarios
- examples from the literature where such effects have been detected or predicted
- possible irreversible changes

- thresholds of change.

Table 5.4: Hydrology–biology linkages, prognoses under drying scenarios, examples from the literature, possible irreversible changes and thresholds of change (see text; see also Table 4.2 Chapter 4 for examples of each linkage on the Gnangara groundwater system; adapted from Horwitz *et al.* 2008).

| **Hydrology–biology linkages** | **Prognosis under scenarios of declining groundwater levels and decreasing rainfall** | **Possible irreversible changes and thresholds of change** |
|---|---|---|
| Requirement for seasonal soil moisture | Reduced vigour as a result of lower water availability in summer (Zencich *et al.* 2002), altered rates of surface soil carbon and nutrient cycling. Potentially reduced seed set and shift in population distribution of plants (Groom *et al.* 2001) and persistence and community composition (Pettit *et al.* 2001). | Transitional states with drying – more research required. |
| Requirement for seasonal availability of moist habitat for aestivation/ drought avoidance | Survival provided moisture levels are sustained. However, if otherwise moist habitats dry, less frequent re-emergence and probably reduced reproduction, may result (examples include burrowing crayfish and frogs, aestivating fish). | Once-off severe drying (of one or more years) may eliminate local and regional populations. |
| Requirement for a seasonal or intermittent surface saturation | Inundation less frequent (e.g. inundation once every 5 years to once every ten years) or seasonality of inundation changes (decreasing winter–spring inundation and possible incidence of summer inundation).<br>Decreased areal extent and duration of inundation can result in reduced frequency of plant recruitment events (Pettit and Froend 2001), and reduced richness of wetland invertebrates (J Davis, unpublished data). | Transitional states with drying for vegetation – more research required. |
| Terrestrial requirement for access to watertable | Acute drawdown and low recharge can result in loss of adult individuals of overstorey and understorey species (Groom *et al.* 2000) or local extinction of susceptible species (Froend and Drake 2006). Less severe circumstances can result in reduced vigour of adults and a shift in the distributions of established juveniles (Groom *et al.* 2001). | Transitional states with drying for vegetation – more research required. |
| Requirement for groundwater discharge to maintain a particular quality of surface water | Extinction of discharge means loss of suitable habitat. Declined volumetric discharge can mean local contraction of habitat, altered water temperature regimes and chemical characteristics. | Local and regional extinction of endemic and other forms wedded to discharge habitats such as mound springs. |
| Requirement for permanent surface or subsurface saturation | Loss of flora and fauna dependent on permanent water when change lakes and streams change from permanent to temporary.<br>Sediments exposed to more frequent drying, potentially displacing biota. Most severe prognosis will be drying, heating and cracking of sediments that have never experienced these conditions, changing sediment structure (Horwitz *et al.* 1999; Semeniuk and Semeniuk 2005) and biogeochemistry; acidification under certain conditions (Sommer and Horwitz 2001). | Local and regional extinction of endemic forms wedded to permanent surface water.<br>Permanent change in sediment structure from severe drying; perpetual acidification from continued drying. Permanent loss of calcium from system.<br>Permanent loss of sediment from burning. |

| Hydrology–biology linkages | Prognosis under scenarios of declining groundwater levels and decreasing rainfall | Possible irreversible changes and thresholds of change |
|---|---|---|
| Requirement for an exchange between surface/subsurface flows and groundwater | Altered patterns of carbon and nutrient cycling. Reduced ability to retreat or emerge according to life history requirement. Potential loss of habitat. Seasonal switches between surface water recharging to groundwater discharging, shifts to surface water recharging only. | If shift to recharge only occurs, then subsurface ecologies change permanently. |
| Requirement for saturated hypogean (interstitial) spaces in aquifer | Where habitat is fixed at a certain stratigraphic level then declines in the saturated zone will strand dependent biota resulting in local extinctions (Boulton *et al.* 2003). Otherwise distributions of short-range endemics may change according to extent of groundwater level reduction (Humphreys 2006). | Local extinctions of biota wedded to particular stratigraphic zones. |

# Wetland recovery and monitoring

Ecological restoration is usually undertaken by addressing the causes of ecosystem degradation and, as discussed above, these causes are many. However, they invariably involve changes to hydrological regimes in some way. Accordingly, management interventions that seek to address hydrological regimes will need to be essential components of wetland restoration on the Swan Coastal Plain.

There are a number of management interventions that aim to prevent permanent loss of, or permanent change in, groundwater-dependent wetland systems. One method involves shutting off nearby extraction bores to reduce local drawdown effects.

Another significant intervention is artificial water supplementation to address wetland degradation associated with groundwater decline. Four iconic wetland systems are currently being managed in this way. These are Lake Nowergup, Twin Swamps, Lake Jandabup, and the root mat communities in Yanchep Caves. This measure requires a lot of management effort and is costly and energy intensive. It aims to prevent the severe effects of drying by establishing a hydrological ‘holding pattern’ until hydrological conditions improve.

The restoration program at Lake Jandabup has used ongoing supplementation to reinstate anaerobia in wetland sediments to reverse the effects of drying-induced acidification. Groundwater has been pumped from the Superficial aquifer in areas adjacent to, and directly into, Lake Jandabup. The recovery has been sound, with the return of aquatic macroinvertebrate communities in less acidic waters, despite the resulting water chemistry and community structure being somewhat different to the original (Sommer and Horwitz 2009, see Table 5.2). Infrastructure installation and maintenance costs, plus a nominal cost of water, have been estimated at a little less than $9m over a ten-year period. Improved efficiencies with respect to feedbacks and timing of groundwater pumping are possible (Horwitz *et al.* in prep.).

The issues of fire and acidification have significant implications in circumstances of declining groundwater levels. If current groundwater extraction rates continue, and rainfall continues to decline, *terrestrialisation* of these sediments will make them vulnerable to acidic metal-rich plumes and fire (where iron pyrite, and organic rich sediments occur, respectively), and neither of these states is desirable. Artificial maintenance of water levels to keep sediments saturated is an obvious solution, using groundwater, stormwater or recycled water. However, it will be difficult for such attention to be given to all the iconic (or not so iconic) wetlands in the GSS study area that need restorative action and to all the sediments requiring saturation.

Another significant type of management intervention for wetland restoration has been that associated with the rehabilitation of inner urban wetlands, addressing the effects of eutrophication and habitat alteration. For at least the last 25 years Lake Monger (described above) has undergone intensive restoration works to achieve '… a more diverse and self sustaining ecosystem that provides a variety of fauna habitats, improves water quality, enhances and protects the natural and historical value and allows for passive recreation, education and community involvement.' (Town of Cambridge 2008).

Studies measured the nutrient loads and levels of other contaminants entering the lake through stormwater drains, and located the point sources of the contaminants (Town of Cambridge 2008). Various methods of treating stormwater to improve the quality of drainage waters were reviewed and preference was given to creating nutrient stripping channels, constructing sediment ponds in front of the drains and planting fringing and emergent native vegetation. Midge reductions have been noticed as a result of rehabilitation and the improved lake water quality (Town of Cambridge 2008). Current investigations seek to determine the contribution of nutrients coming from groundwater seepage, particularly from the areas east of the lake, with a view to its interception. Good progress has been made in increasing the fauna habitat around the lake. Throughout the

1980s and 1990s, native vegetation was returned to the reserve, providing an important habitat and drought refuge for waterbirds and tortoises. Refuge islands have been developed and extensive habitats created. Efforts to minimise feeding of waterbirds by visitors, and removal of rocks used in the lake wall, have resulted in an improvement of bird health (Town of Cambridge 2008). Other management strategies, such as public education have also played a significant role.

## Wetland monitoring

Wetlands within the GSS study area on the Swan Coastal Plain come under the jurisdiction of local, state and national authorities. Sectors include natural resource management, health, planning and defence authorities. Wetland management and monitoring is consequently a broad agenda in governmental terms. This official activity is complemented by community interest and the involvement of groups such as Friends of Yellagonga, Friends of Star Swamp Bushland, and Friends of Western Swamp Tortoise who play an important role in management and monitoring.

The monitoring of hydrology is regarded as being of the utmost importance. It is conducted principally by the Western Australian Department of Water, using monitoring bores adjacent to wetlands and staff gauges in wetlands themselves. However, this type of monitoring cannot on its own reveal deleterious changes to ecosystems, or attribute such changes to particular water resource management activities. More appropriate for this purpose are the monitoring programs established for ‘criteria’ wetlands in the GSS study area following the review of environmental conditions carried out under Section 46 of the *Environmental Protection Act 1986*, and also following the release of the *East Gnangara water provisions plan*. Surface and/or groundwater levels are monitored at least monthly at all of these wetlands in the GSS study area. Linked with the water level measurements are measurements of vegetation, macroinvertebrate communities, water chemistry and in some instances frogs (as below, reported by Froend *et al.* 2004).

Macroinvertebrates and water quality at up to 20 GSS study area wetlands are monitored twice per year – in spring when water levels and biological activity are at their highest and in summer when water levels at permanent wetlands are at their lowest, or shortly before ephemeral wetlands dry out, when the major stress to fauna occurs. The main objective of this monitoring is to describe the status of aquatic macroinvertebrates in terms of family richness and community structure and their seasonal response to changes in water quality and water levels. Macroinvertebrate parameters rely on an historical data set, working on the assumption that variation within the historical range of fluctuations is acceptable. Similar monitoring occurs for significant root mat communities and tumulus spring sites.

Vegetation monitoring is undertaken annually in spring across permanent transects at a similar number of wetlands in the GSS study area. The main objective is to monitor changes in fringing and emergent vegetation and to determine if any of these are related to changes in groundwater levels or other factors (WAWA 1995). This monitoring is an important source of information on changes in ecological condition.

As part of the environmental commitments made in the public environmental review for the *East Gnangara environmental water provisions plan*, monitoring of frog populations in six wetlands in the GSS study area has been carried out for the Department of Water since 2000 (Davis and Bamford 2003). During the 2003 survey period, changes were implemented so that the project focused more intensely on a smaller number of wetlands than was the case in the original study. Waterbirds are not formally monitored at wetlands in the GSS study area so ad hoc or community based activities are important.

The breadth of chemical and biological parameters should be sufficient to detect trends in all the types of hydrology–biology linkages mentioned in Table 5.4. Ideally, therefore, the parameters will provide early warning indicators, sufficiently instructive to allow for adaptive management of the water resource. To achieve this, the links between indicator trigger levels, wetland management objectives, and water resource management must be well understood and articulated. So far in the GSS study area, such imperatives have probably not been achieved.

## Discussion

While the geological history of the wetlands on the Swan Coastal Plain is complex and equivocal, there is sufficient evidence to argue that since the mid-Holocene the wetlands in the GSS study area have experienced wetter and drier periods than today. Similarly it is clear that in recent historical times since colonisation, water levels in wetlands in the GSS study area have fluctuated significantly, with periods of much higher water levels, and periods with equivalent low water levels to those experienced currently. Further groundwater declines in the Gnangara groundwater system in line with current projections will push wetlands towards, and probably beyond, these drier historical limits.

The ability of wetland biota to respond to lower water levels depends on their particular requirements for water. Some plants and animals are undoubtedly more vulnerable than others in this respect and will become rarer in the GSS study area. Others with elements of drought tolerance will become more common or widespread. Of critical importance here will be the types of disturbances that produce irreversible impacts, such as local or regional extinctions of populations or species, removal of sediment (for example by excavation or severe fire), or fundamental changes in sedimentary processes, where hydrological regimes have been altered.

Changes to hydrological regimes are implicated in all issues of wetland loss, degradation and change, so interventions that seek to address problems with hydrological regimes will need to be essential components of wetland management and restoration on the Swan Coastal Plain.

Current wetland management, biological monitoring (this chapter) and wetland reservation (Chapter 4) both emphasise iconic wetlands where surface waters are prominent or permanent. Wetland management in the GSS study area does not at present cover the full spectrum of hydrology–biology linkages across all different wetland types. Highly localised damplands and extensive palusplains for example do not receive the same attention as permanent wetlands, and these types of wetlands are just as vulnerable to groundwater decline. The role that constructed wetlands, with their permanent waters, can play in the conservation of biodiversity in the GSS study area remains an important area for investigation.

Because of cost, and the volume of water required, it will be difficult to use artificial augmentation of water on all the iconic, let alone the not so iconic, wetlands on the Mound that need protection from, and restoration as a result of, groundwater decline. Where supplementation is warranted, water can be obtained from groundwater, stormwater or recycled water, and the relative advantages and disadvantages of these options needs closer investigation. Closer attention to longer term restoration strategies for the Mound as a whole will be required to ensure wetland values are maintained in the region.

Wetlands of the Swan Coastal Plain have experienced periods of higher water levels and lower water levels over the last 10000 years, but also in more recent times. Wetlands have undergone many changes over the last 200 years, including drainage of low-lying areas on the Swan Coastal Plain, infilling of wetlands, groundwater abstraction, and a more recent decline in rainfall. Together these factors have produced current groundwater declines across the GSS study area and associated declines in water levels within the wetlands.

Wetlands are dynamic systems that are influenced by geography and climate. Superimposed on, or overriding, these influences are anthropogenic effects. Processes that are involved in the degradation and loss of wetlands and groundwater-dependent ecosystems include:

- altered hydrological regimes
- land use change
- nutrient enrichment
- acidification
- the impacts of fire.

Rainfall has the greatest impact on groundwater and wetland depth under natural conditions. However, other factors influence long-term groundwater levels, such as urbanisation, groundwater abstraction, and agriculture, which has increased the amount of water abstracted for public and private use.

In response to a drying hydrological regime and falling levels of groundwater, changes in vegetation that is dependent on groundwater will vary, depending on their tolerance of the amount, seasonality and availability of moisture in the soil. Wetland sediments will also be affected by a drying hydrological regime.

Land use changes include clearing, filling and fragmentation of wetland habitats. A rise in the water levels of wetlands can occur in some areas following clearing of native vegetation for urbanisation, while excessive groundwater abstraction can reduce water levels in other areas.

Wetlands are parts of landscapes where nutrients generated in the catchment can accumulate, where they are involved in biological metabolic processes, biomass growth, decomposition and deposition. An oversupply of nutrients results in an altered ecology and diminished ecosystem services, such as diminished water quality which in turn affects wetland invertebrates and vertebrates, and build up of toxins that can have consequences for the health of human and birdlife.

Acidification (drought-induced, from dewatering or from direct disturbance by excavation) is a change in water chemistry that has biological consequences; it can change aquatic plant and algal communities, cause declines of sensitive species (i.e. fish, crustaceans and molluscs), leading to characteristics reductions in macroinvertebrate richness and abundance.

Fire is part of the ecology of wetlands, and some riparian species depend on fire for recruitment. However wetlands are also places where fire sensitive species and processes occur, and changed fire regimes particularly fires that are intense or frequent, can alter wetland ecologies. The hydrology can change as a result, water chemistry can change (even toward acidification), sediments in the water can increase, species of flora and fauna can disappear (or dominate, as is the case with some opportunistic or weedy species), and organic rich sediment can be consumed by fire, releasing carbon into the atmosphere and toxic materials into the groundwater. These impacts are exacerbated by, and perhaps linked to, recent declines in rainfall, groundwater levels and reduced saturation of sediments, such as peat.

Seven threatened ecological communities are either wetland in nature or are groundwater dependant ecosystems. Threats to these communities include altered hydrological regimes, clearing, fire, climate change and invasive weeds.

Hydrological change on the Swan Coastal Plain is therefore the underlying feature of wetland change, but it has been driven by several processes, mostly anthropogenic:

- climate variability (e.g. inter-annual variability in temperature and rainfall)
- patterns of land-use change (e.g. clearing for urbanisation and agriculture)
- patterns of water regulation (e.g. damming of water courses)

- patterns of groundwater extraction (e.g. extraction for private bores)
- water infrastructure (e.g. collection of stormwater runoff)
- distal societal drivers (e.g. lack of adequate water pricing).

Management methods for restoring wetland systems, including groundwater-dependent wetland systems, or for preserving them, include shutting off nearby extraction bores to reduce local drawdown effects, artificially supplementing water, which can also help alleviate some of the impacts of fire on wetlands, and rehabilitation of inner urban wetlands.

## Recommendations

In order to improve understanding of process of degradation for wetlands and wetland recovery and monitoring, it is recommended that:

- addressing the problems of hydrological regimes be an essential component of wetland management and restoration on the Swan Coastal Plain
- that work be undertaken to fill the significant gaps that exist in our understanding of wetland management, and to provide the data needed for early warning of problems, covering all hydrology–biology linkages across all different wetlands types
- that further investigation be undertaken into the advantages and disadvantages of applying water supplementation to wetlands, including looking at the advantages and disadvantages of using groundwater, stormwater or recycled water
- further work be undertaken to fill the need for longer-term restoration strategies for the GSS study area as a whole to ensure wetland values are maintained in the region.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter Six: Habitat Loss and Fragmentation

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Our environment, our future

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Department of Water
Department of Agriculture and Food WA
Department for Planning and Infrastructure
Department of Environment and Conservation

Water Corporation

CSIRO

Forest Products Commission Western Australia

# 6. HABITAT LOSS AND FRAGMENTATION

## Key points

- Habitat loss has been found to produce negative effects on biodiversity. However the evidence of the negative impact of habitat fragmentation per se is weaker.
- Fauna species respond to habitat loss and fragmentation in a diversity of ways. Declines in species richness of bird and mammal communities and species abundance are common, but consequences for reptiles are poorly understood.
- In the past, the methods of dealing with the adverse effects of fragmentation have focussed on management that will benefit many species and processes simultaneously, such as the establishment of ecological linkages.
- Clearing of remnant vegetation in the study area has been extensive and while this has been identified as a major threat to biodiversity, information on the specific impacts is limited.
- Pine plantations provide only low quality habitat for many native species. However, populations can persist in remnant patches of native vegetation, if the habitat and patch connectivity are of a suitable quality.
- The remnant native vegetation (~ 101 000 ha) in the study area is significant as it retains some of the largest contiguous areas of native vegetation on the Swan Coastal Plain.
- The degree of retention of native vegetation varies considerably between the vegetation complexes, with only three having > 30% of their remnant vegetation protected.
- The ecological linkages proposed for the study area consist of discrete patches of remnant vegetation a minimum of 500 m wide, connecting areas with > 60% remnant vegetation and key biodiversity assets.
- There is a need to finalise the protection status of remnant vegetation identified by Bush Forever and provide additional protection for Beermullah, Guildford, Swan, Yanga, and Coonambidgee complexes.
- The requirements of proposed ecological linkages for fauna need to be investigated further, particularly for species such as honey possum (*T. rostratus*), Carnaby's black-cockatoo (*C. latirostris*), rakali (*H. chrysogaster*) and quenda (*I. obesulus*).

- An ecological linkage framework has been developed for the GSS study area. This needs to be used to prioritise areas for development of corridors based on cost-benefit analyses and information on requirements for fauna.

# Introduction

Habitat loss from clearing is recognised as a major threat to biodiversity in Australia and throughout the world (Saunders *et al.* 1991). In Australia, clearing of native vegetation for agriculture and urbanisation has been extensive, particularly in temperate grassland, forests and woodlands (Burgman and Lindenmayer 1998; Hobbs and Yates 2000). There is strong evidence that when the amount of habitat within a landscape declines, species richness and abundance declines (Fahrig 2002; Lindenmayer and Luck 2005; MacArthur and Wilson 1963). In addition, the population size of remaining species decreases until many species exist only in small, isolated populations (Huggett 2005). While habitat loss has been found to consistently produce negative effects on biodiversity, the evidence for the effects of habitat fragmentation per se (changes in configuration and breaking apart of habitat) is much weaker and may be positive or negative depending on the species (Fahrig 2003).

Habitat loss and fragmentation of remaining remnant vegetation also causes significant changes in the physical and biophysical environment. Following habitat clearing, remnant native vegetation is typically surrounded by a matrix of agricultural, urbanised or other developed land (Saunders *et al.* 1991). Major changes in the surrounding land use have subsequent impacts on the remnant vegetation by altering ecosystem processes such as water and nutrient movement across landscapes (Saunders *et al.* 1991). The size of the remnant habitat is a critical determinant of species richness and population abundance (MacArthur and Wilson 1963; Rosenzweig 1995). In addition, the size and shape of the remnant vegetation is important in determining impacts from edge effects (e.g. spread of weeds, altered light levels, increased predation) and influences the diversity and integrity of remaining biota (Rosenzweig 1995). Furthermore, populations in a fragmented landscape have an increased risk of extinction, and connectivity between fragments is often considered critical for successful recolonisation (Fahrig 2002).

Throughout much of the world, conservation of regional biotas now depends entirely on the retention and management of remnant habitats. The size and integrity of reserves is often positively related to increasing distance from urban areas, with smaller reserves typically close to metropolitan areas. The management of reserves with a high perimeter to area ratio is often very difficult (Panetta and Hopkins 1991). Small remnant patches may be at greater risk from disturbance and degradation via increasingly frequent, or high intensity fires, weed invasions and fragmentation caused by human use (Hobbs 1993).

Habitat loss in south-west Western Australia has been extensive. In conjunction with the area's biodiversity values, this has contributed to its listing as a biodiversity hotspot (Mittermeier *et al.* 2004). Approximately 70% of vegetation in south-west Western Australia has been cleared (Mittermeier *et al.* 2004) with some regions (e.g. Wheatbelt), having experienced up to 90% habitat loss (Hobbs 1993). On the Swan Coastal Plain ~ 65% of habitat has been cleared (as of 2007). Of the remaining remnant vegetation on the Swan Coastal Plain ~ 20% is located within the GSS study area.

In the GSS study area, clearing of remnant vegetation for urban development, agriculture and pine plantations has been extensive, especially in the south-west and eastern boundaries. In total, 53% of native vegetation (113 352 ha) is estimated to have been cleared in the GSS area as of 2005-2006. The majority of remnant vegetation patches in the urban and agricultural parts of the system are thus small and highly fragmented, while larger, more intact, remnants remain on Crown land to the north (Figure 6.1). Most of the biologically significant bushland reserves in the metropolitan area are identified by the Bush Forever Plan (Government of Western Australia 2000b), while larger remnants are protected under DEC-managed estate and on unallocated Crown land (not considered 'conservation' estate). Significantly some of the largest intact areas of remnant vegetation on the Swan Coastal Plain are contained within the GSS study area. For example, remnant vegetation, (Figure 6.1) that encompasses Yanchep National Park, the Wilbinga reserve complex, Yeal Nature Reserve and Whiteman Park.

Habitat loss and fragmentation on flora, fauna and ecosystems have been identified as major threatening processes for the DEC Swan Region (DEC 2009) but information on specific impacts on biodiversity is limited. In their audit of biodiversity of the SWA2

IBRA Bioregion, Mitchell *et al.* (2003) also identified that there is little quantitative data on the impacts of fragmentation.

Wildlife corridors or ecological linkages have been identified as a useful conservation strategy to restore connectivity of fragmented landscapes. There is extensive literature on corridors and various reasons are given for developing or retaining them. The benefits of ecological linkages include enabling wildlife to colonise new sites or move out of sites that become unsuitable. They also may permit recolonisation of extinct populations, promote movement of species between different areas and increase the overall extent of habitat within an area. Whilst we may not have all the information on how every species will utilise, or benefit from such linkages, they are likely to benefit many species and processes simultaneously, and can be part of an adaptive management approach (See Chapter 12).

The aims of this chapter were to:

- collate and review literature on the distribution of clearing and fragments
- review information on the impacts of habitat loss and fragmentation on biodiversity
- identify gaps in our knowledge of the occurrence and impacts on the Gnangara groundwater system.

These aims were addressed by reviewing and assessing literature on:

- the impacts of habitat loss and terrestrial fragmentation on biodiversity (hydrological fragmentation has been addressed in Chapters 4 and 5);
- impacts at a landscape level
- consequences for communities, ecological interactions
- conservation and management in fragmented landscapes
- the role of ecological linkages.

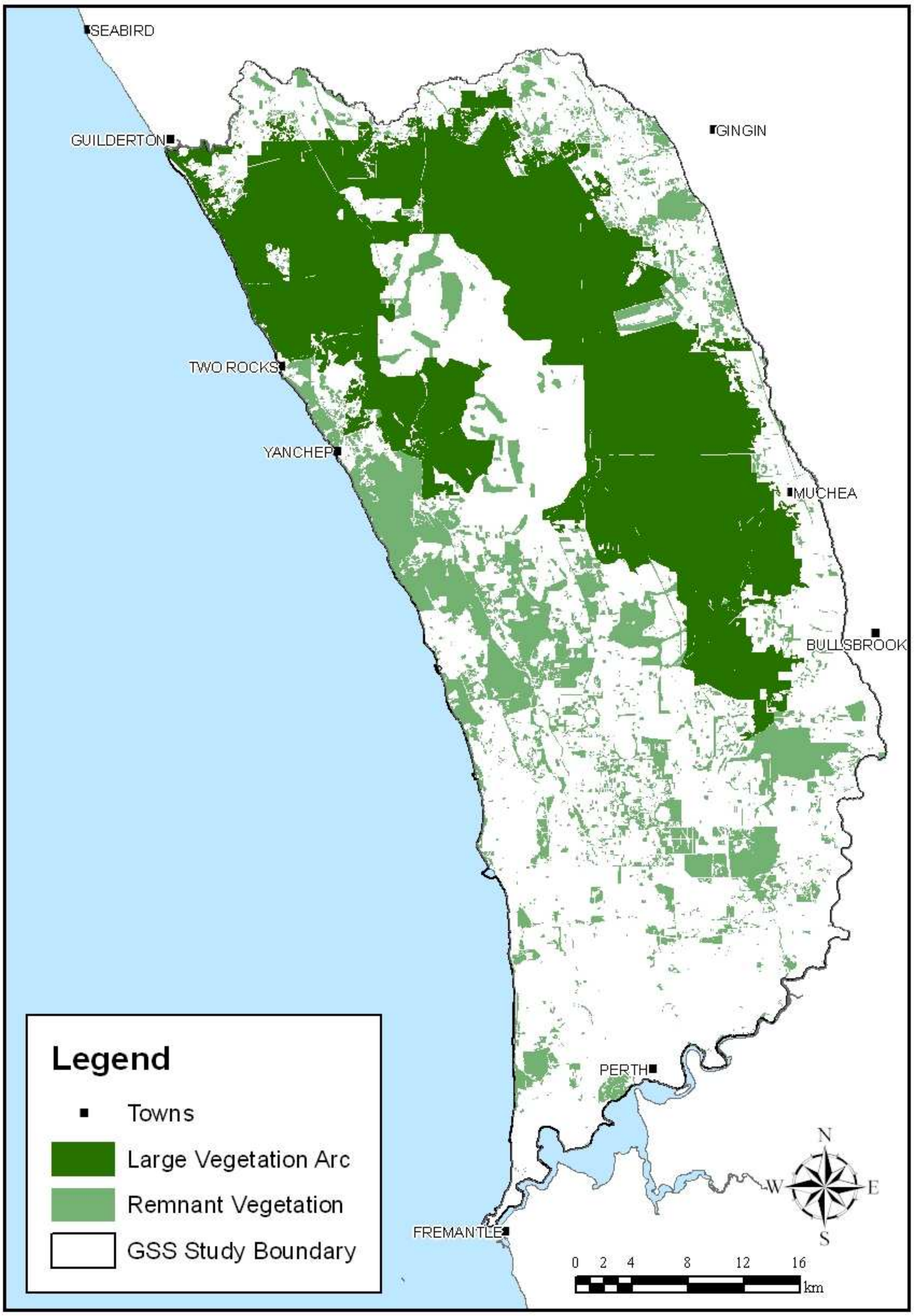


Figure 6.1: Distribution of remnant vegetation within the GSS study area, illustrating the large arc of remnant vegetation.

In addition, a number of DEC GSS projects were undertaken to address identified knowledge gaps for the GSS study area. These projects are outlined below.

**1) Levels of retention, fragmentation and representation of remnant vegetation and vegetation complexes**

This project involved a spatial evaluation of the historical and current distribution of remnant vegetation in the study area. The objectives were to determine what vegetation complexes occur entirely within the study area, quantify the level of representation of complexes at both the regional and study area scale, and to identify which complexes require additional retention and protection within the GSS study area.

**2) Ecological linkages for the pine plantations and the GSS study area**

The aim of this project was to develop plans to retain or promote ecological connectivity between the remnant native vegetation across the landscape. The primary objectives were to design ecological linkages at the landscape, sub-regional or local level, focused on key assets. Ecological linkages proposed for the 23 000 ha of pine plantations, post-pine removal, were also incorporated.

**3) Spatial links connectivity analysis**

The purpose of this project was to analyse the landscape and highlight those areas that may have the greatest potential as fauna corridors. Spatial links analyses were employed to determine pathways of adjacent grid cells across the landscape which would be suitable for biodiversity corridors.

The major findings of these projects have been summarised in this document. Full technical reports are provided in technical reviews (see Appendix 2).

## Habitat loss and fragmentation

Habitat loss and fragmentation occur when contiguous habitat, such as forest or grassland, is cleared and divided into separate areas, thus resulting in a progressive reduction in the amount of habitat (i.e. loss), and fragmentation per se (Forman 1999). Major types of changes that result during the process include:

- an overall loss of original habitat area
- reduced size of remaining patches
- increased distance between patches
- decreased connectivity between patches
- the introduction of different land-uses around the remnants (Bennett 2006).

Early research to understand the impacts of habitat loss and fragmentation were strongly influenced by the equilibrium theory of island biogeography (MacArthur and Wilson (1963), which accounted for the observation that the number of species on islands was lower than in mainland areas of comparable size. It proposed that species richness on islands represents a balance between the rate of colonisation of new species and of extinction of species already present. The rate of colonisation is determined primarily by isolation from the mainland, the rate of extinction mainly by island size. A small island will have fewer species than a larger island of comparable isolation and for islands of similar size, those that are distant will have fewer species than those close to a mainland 'source'.

This theory was then extended to mainland isolates and habitat fragments (Brown 1971; Diamond 1975), and was a framework for studying the effects of habitat fragmentation. The theory that species richness and individual abundance will decrease with reduced patch size has been supported by studies worldwide (Dunstan and Fox 1996; Fahrig 2001). However, it is now recognised that the biota of fragmented habitats are also influenced by management within the remnant habitat and land-uses in the surrounding landscape. Further impacts vary according to the biotic group of interest. The concept of isolation has also changed somewhat from distance to a 'mainland' to distance between neighbouring patches (Haila 2002).

There are different approaches to assessing the effects of landscape changes on biodiversity, one focusing on impacts on multiple species (the species richness and composition approach), and another which addresses impacts on individual species. While the first approach can inform our understanding of impacts on patterns of change in species assemblages, it does not provide detailed species-specific responses to changes in habitat loss, fragmentation, patch size, connectivity and habitat suitability (Lindenmayer and

Fischer 2006). A key issue for management is to determine when it is appropriate to concentrate on a detailed assessment for an individual species, and when it is necessary to undertake a multispecies approach. In some cases both approaches may be complementary and thus both may be required.

Fahrig (2003) has proposed that the processes of habitat loss and fragmentation be treated separately because the merging of the two has often obscured empirical results or interpretations as to their relative impacts. Thus, habitat fragmentation would be used to describe changes in habitat configuration across a landscape resulting from breaking apart of habitat, independent of habitat loss. While habitat loss always reduces the original biodiversity, the evidence for habitat fragmentation per se is much weaker and may be positive or negative.

McIntyre and Hobbs (1999) recognised that the process of habitat destruction occurs over time and results in variable changes in the type and arrangement of habitats. According to their classification of human-induced landscape transformation, four categories can be identified (intact, variegated, fragmented, or relictual). The categories represent stages along a continuum of destruction of intact landscape. The variegated landscape mosaic is dominated by predominantly natural habitats, with varying degrees of disturbance or degradation, for example grazing areas converted to exotic pasture but surrounded predominantly by a native grassland and woodland landscape matrix. The fragmented landscape has a clearer distinction between remnant areas of original habitat and a highly modified surrounding environment, or matrix, as exemplified in the Wheatbelt of Western Australia which has discrete patches of native vegetation in a matrix of agriculture (Hobbs and Saunders 1993). Finally, the relictual landscape contains small, distinctly isolated remnant areas which are under extreme pressure from the intensively used matrix.

Methods for examining the impacts of fragmentation on landscapes vary. Some focus on assessing spatial patterns of vegetation cover across the landscape, while others concentrate on assessing alterations to ecological processes (Lindenmayer and Fischer 2006). The former approach concentrates on the composition and spatial configuration of habitat components at a landscape scale (Turner *et al.* 2001; Wiens 1999). The designation of fragments and classification of landscape features into areas such as patch, corridor and matrix are often based on human perceptions and do not necessarily reflect responses of organisms. However, there is a need to have more explicit classifications based on the organism(s) being targeted and to focus on the processes and mechanisms of organisms in fragmented landscapes, such as movement between habitats, selection of suitable habitat and resources, and the impacts of predators and competitors (Fischer *et al.* 2005).

## Impacts on ecological processes

Fauna and flora species respond to habitat loss and fragmentation in a wide variety of ways. Some species become more abundant while others decline to local or regional extinction. Attributes that influence the vulnerability of species to fragmentation include tolerance to environmental change, population size, and vulnerability to habitat isolation (Bennett 1990b; Wilson and Friend 1999).

Changes to habitats within remnant vegetation, especially degradation of the ground layer, often results in impacts on animal populations as they influence the availability of suitable habitat for foraging, shelter and refuge (Brown *et al.* 2008; Knight and Fox 2000). Altered landscape processes play an important role in changes to habitat quality and availability in fragmented landscapes, particularly in the form of ‘edge effects’. The high proportion of edge habitat in small remnants and linear strips of road verges allows potentially negative edge effects to penetrate further into the fragment and affect the environmental conditions of plant populations (Bennett 2006; Honnay *et al.* 2005). Changes in landscape processes such as temperature regimes, radiation and wind fluxes, and altered hydrological and nutrient cycles have all been implicated as potentially degrading forces at fragment edges (Hobbs 1993; Saunders *et al.* 1991). For example, strong winds can result in uprooting of trees, pruning (leaf and stem damage), increased evapotranspiration and desiccation, and can aid the dispersal of external nutrients from adjacent land uses (e.g. agriculture) and invasive weeds into the fragments (Hobbs 1993; Hobbs and Atkins 1988; Saunders *et al.* 1991).

Population size is an important attribute of remnant areas because it affects vulnerability to stochastic or chance processes such as decline, extinction and loss of genetic diversity (Gilpin and Hanski 1991). Random variation in survival or reproductive output (demographic stochasticity) can lead to fluctuations in per capita growth rates that may threaten persistence of small populations (Stephens *et al.* 1999; Wilson and Bradley 2006). Fluctuations in environmental conditions and catastrophes (e.g. floods, wildfire) are also more threatening to small populations (Gilpin and Hanski 1991; Hanski and Gilpin 1997).

The concept of the metapopulation was developed by Levins (1970), and later refined by Gilpin and Hanski (Gilpin and Hanski 1991; Hanski 1999; Hanski and Gilpin 1997). The metapopulation is a collection of subpopulations interconnected by dispersal. Maintenance of the metapopulation dynamics of species in fragmented landscapes is important for management and conservation (Gilpin and Hanski 1991; Hanski 1999; Hanski and Gilpin 1997). While separate populations may become extinct, recolonisation from other populations may occur if they are connected by suitable habitat. The importance of maintaining structural and functional connectivity in habitat networks that preserve dispersal for metapopulations has been recognised.

As populations decline in size or density, fitness also decreases, often resulting in a reduced rate of growth for the population (Stephens *et al.* 1999). These effects may be experienced by plants when populations are too small or isolated to attract the required pollinators (Aizen and Feinsinger 1994; Groom 1998) or arise from reduction in fertility and viability associated with high levels of inbreeding (Wilson and Bradley 2006). Allee effects were observed in plant populations of *Banksia goodii*, north of Albany, whereby small patches suffered lower seed production due to a lack of effective pollination (Groom 1998; Lamont *et al.* 1993).

## Impacts on plant populations

The effects of fragmentation on plant communities have been studied extensively in woodlands in the wheatbelts of south-west Western Australia (Hester and Hobbs 1992; Hobbs and Atkins 1988; 1991; Norton *et al.* 1995) and south-east Australia (Cunningham 2000; Prober and Thiele 1995); as well as in the tropical to temperate forests of Australia, Europe, South and North America (e.g. Aizen and Feinsinger 1994; Honnay *et al.* 2005). Fragmentation effects on plant communities have also been the topic of several reviews (Hobbs 1993; Hobbs 2001; Hobbs and Yates 2003; Honnay *et al.* 2005; Saunders *et al.* 1991).

The effects of fragmentation on plant communities have been difficult to generalise because expression depends on the type of ecosystem (e.g. forest, woodland, heath or grassland, temperate, tropical, montane) and extent and type of ecosystem transformation (e.g. varying degree of modification to the natural landscapes in an agricultural, plantation forestry or urban landscape) (Hobbs and Yates 2003; Honnay *et al.* 2005). Additionally, the original distribution of plant populations (e.g. rare or patchily distributed) before fragmentation and the species' life-history characteristics (e.g. reliance on seed for regeneration) also play a role in the outcomes attributable to fragmentation (Hobbs and Yates 2003).

Species richness has often been used as a measure of the impact of fragmentation on fauna communities (correlated to patch size and degree of remnant isolation) (Davis *et al.* 2001; Dunstan and Fox 1996) but for few floral communities (Prober and Thiele 1995). Many botanical studies instead investigate impacts on plant reproduction, (e.g. pollen and seed production, viability, fecundity) (i.e. pollen and seed production, viability, fecundity; Aizen and Feinsinger 1994; Cunningham 2000), individual species abundance or distribution (Norton *et al.* 1995; Yates and Broadhurst 2002), and changes to native and non-native community structure (Hester and Hobbs 1992; Milberg and Lamont 1995). Fragmentation-driven declines in species richness have not been so readily identified for plant communities, possibly due to the delayed effects of fragmentation and the sessile nature of plants (Norton *et al.* 1995).

The delayed effects of fragmentation on plant population extinctions, and thus species richness (Honnay *et al.* 2005; Saunders *et al.* 1991) occurs in remnants which may contain more plant species than they are able to sustain. However, the consequent reduction in species is not immediate and species are lost over time ('species relaxation') (Bennett 2006; Saunders *et al.* 1991). The long-lived nature of some plant species may delay 'species relaxation' further as populations persist for a longer period than expected because of the longevity of individuals or seed banks (Aizen and Feinsinger 1994; Hobbs and Yates 2003; Honnay *et al.* 2005; Saunders *et al.* 1991). These longer-lived species are considered part of the community's 'extinction debt', which can persist for hundreds of years after fragmentation has occurred (Bennett 2006; Honnay *et al.* 2005; Saunders *et al.* 1991). Bennett (2006) suggests that many regions in Australia still have an 'extinction debt' and are therefore yet to experience the full ecological consequences of fragmentation.

The relative size of vegetation fragments influences the level of plant species richness where, as habitat size decreases, plant species richness decreases (Prober and Thiele 1995). Factors which contribute to lower species richness in small patches include:

- a decrease in the original habitat area
- a reduced diversity of habitats
- lower population sizes resulting in a decreased ability to maintain viable populations (Connor & McCoy 1979 cited in Bennett 2006).

External factors also have a greater influence on smaller patches because these patches are more susceptible to edge effects (Saunders *et al.* 1991). Prober and Thiele (1995) found that in the fragmented Wheatbelt woodlands of south-east Australia, species richness generally increased with remnant size. However, smaller remnants were more vulnerable to edge effects such as weed invasion which can affect species survival (Hobbs and Atkins 1991). This was observed in *Banksia* woodland roadside remnants where fire increased the number, abundance and cover of weed species, usually to the detriment of native vegetation (Milberg and Lamont 1995).

### *Altered species interactions*

Habitat fragmentation and land uses within the intervening landscape are major threats to plant-pollinator interactions (Aizen and Feinsinger 1994; Groom 1998). As fragments and populations become smaller and more isolated, rates of visitation by pollinators and thus plant fecundity may decline (e.g. Groom 1998). Plant-pollinator interactions may be disrupted if the local pollinator population declines within a fragment, the isolation of the fragment is greater than the foraging range of the pollinator, or wide-ranging pollinators avoid small plant populations or isolated fragments (Hobbs and Yates 2003). Hobbs and Yates (2003) reviewed numerous studies that have tried to test these hypotheses with varying results, illustrating that it may be difficult to generalise or predict the impacts of fragmentation on pollinator abundance, pollination success and thus fecundity.

Changes in diversity, abundance, diet or behaviour of seed predators and dispersers may affect the quantity of seed entering fragments. As many as 1500 native dry heath and sclerophyllous plants in Australia are myrmecochorous whereby they are regularly dispersed by ants (Berg 1975). As with pollination, it has been hypothesised that the collapse of dispersal mutualisms with fragmentation could result in the local decline of plant species within fragments.

Burbidge and Whelan (1982) studied seed dispersal in *Macrozamia riedlei* and found that an absence of animals resulted in diminished dispersal. Without animal dispersal, ripe seeds dropped directly beneath the parent, no more than 40 cm from the base. They recorded several animals eating the fleshy exterior, including the brush-tailed possum (*Trichosurus vulpecula*), which was recorded as transporting the seeds up to 24 m from the parent plant. For plants such as the *Macrozamia riedlei* which have large, heavy seeds it is unlikely any effective dispersal would occur in the absence of animals (Burbidge and Whelan 1982; McGrath and Bass 1999).

The emu (*Dromaius novaehollandiae*) has been recorded consuming a wide range of plants (Davies 1978), which combined with their long gut retention times, their ability to pass seeds intact through the gut and their mobility makes them excellent long distance dispersal agents (Calvino-Cancela *et al.* 2006). They too have been recorded consuming the seeds of *Macrozamia riedlei* (Carter 1923; Sargent 1928). Emus are not strictly frugivorous and hence their role as seed dispersal agents is not limited to species with fleshy structures (Calvino-Cancela *et al.* 2006). This means that many native plant species can benefit from their long distance transport capabilities (McGrath and Bass 1999). Emus are also likely to be of ecological significance in natural regeneration after disturbances have depleted local seed banks, and in providing a vital connecting link which maintains genetic diversity in an increasingly fragmented environment (McGrath and Bass 1999). In contrast, the southern cassowary (*Casuarius casuarius johnsonii*) is a keystone species of the tropical rainforests of northern Australia upon which a large number of plant species depend for seed dispersal (Latch 2007; Lindenmayer *et al.* 2000). However, it is listed as Endangered under the *EPBC Act*. Threatened by fragmentation itself, the potential collapse of this vital animal-plant mutualism, and the threat posed to the plants that rely upon this species as a disperser can be foreshadowed (DEWHA 2009).

### *Changed disturbance regimes and increased incidence and abundance of invasive species*

Tree clearing, grazing by livestock, and altered fire regimes are disturbances that have been linked to an increase in the abundance of invasive weeds, resulting in reduced native species richness (Hobbs and Atkins 1988; Prober and Thiele 1995). Fire in fragmented landscapes can be eliminated or decline if there is a loss of continuous vegetation cover or if active fire management is not continued. Alternatively, fire frequency may increase, due to proximity to roads or humans where the chance of fire accidents can increase. These scenarios can be detrimental to a range of species. The removal of fire can affect species that require it to stimulate flowering, seed release or germination. Increased fire frequency, through regular escapes from prescribed fires or arson which is prevalent in urban and peri-urban areas, can lead to failure of regeneration or death of native plants, and increased weed growth, particularly if the frequency and intensity is not conducive to native species (Baird 1977).

Fire is a disturbance that facilitates the invasion and increased growth of exotic annuals (Baird 1977; Fisher *et al.* 2006; Hobbs and Atkins 1988). Although fire in a large reserve may have little effect on invasion by non- native species, it can have a severe effect on small remnants such as road verge vegetation, remnants surrounded by roads and firebreaks and other remnants with a high perimeter to area ratio, by allowing increased weed invasion.

## Impacts on vertebrate fauna

In Australia, studies that have examined the effects of habitat loss and fragmentation of forests and woodlands on birds have identified declines in the status of many species (Antos and Bennett 2005; Barrett *et al.* 1994; Mac Nally *et al.* 2000; Major *et al.* 2001). Similarly, declines in native mammals in fragmented forest environments have been found (Bennett 1990a; Law *et al.* 1999; van der Ree 2002). In contrast, reptiles have received less consideration and the consequences of landscape change are inadequately understood (Brown 2001; Driscoll 2004; Fischer *et al.* 2004). Significant relationships between the size of habitat remnants and number of species have been shown for birds, mammals and reptiles (Kitchener and How 1982). Species richness is also correlated with the size of conservation reserves (Kitchener *et al.* 1980a; Kitchener *et al.* 1980b).

Reptiles are considered to be vulnerable to habitat loss, fragmentation and degradation characteristically associated with landscape change in agricultural environments (Brown *et al.* 2008; Kitchener *et al.* 1980a; Smith *et al.* 1996). Most reptile species are sedentary and of low mobility, suggesting that they may have limited capacity to move between patches of habitat isolated by clearing or land-use (Driscoll 2004; Fischer *et al.* 2005; Fischer *et al.* 2004; Sarre *et al.* 1995). Several studies have identified remnant size, shape and degree of connectivity as significantly influencing the occurrence and composition of the reptile fauna in fragmented landscapes in southern Australia (Driscoll 2004; Mac Nally and Brown 2001; Smith *et al.* 1996).

At the microhabitat level, structural complexity of vegetation and habitat components at ground-level are important to many reptile species (Downey and Dickman 1993; Fischer *et al.* 2003; Greer 1997; Hutchinson 1993; Kanowski *et al.* 2006; Shine 1987). Reptile species that forage and live in the ground layer, litter or subsoil, are likely to be severely affected as their microhabitats are degraded by activities such as stock grazing, weed invasion, and soil compaction (Brown 2001).

Despite extensive clearing of vegetation threatening reptiles in Australia, recent studies have shown that small and isolated bushland reserves can support a high proportion of the original reptile fauna for a period of time following urbanisation (Gardner *et al.* 2007; Jellinek *et al.* 2004; Tait *et al.* 2005). This suggests that remnants continue to retain sufficient resources to support populations following isolation and that reptiles may be resilient to urbanisation, or slow to be adversely affected by landscape-level changes.

There is evidence that while some species are able to adapt to urban environments, others are restricted to remnant vegetation patches or else may disappear locally. Species that have the ability to move through and live within the built matrix are thus more resilient in the urban setting (Garden *et al.* 2006). Species that are sensitive to the built environment due to unsuitable habitat, lack of food, barriers for movement and increased predation, become restricted to vegetation patches of suitable habitat. This results in fragmentation of populations and increased risk of local extinctions. Large reptile species, particularly snakes, have been identified as the most adversely affected species (e.g. Cooper 1995; How and Dell 1994; 2000).

In Australia fragmentation resulting from historical disruption of regional landscapes for settlement and farming has resulted in substantial changes in the species richness and composition of mammalian fauna in remnants of temperate forests (Bennett 1987; 1990a; b; Lunney and Leary 1988; Recher *et al.* 1987); rainforests (Laurance 1990; 1997); woodlands and shrublands (Bennett *et al.* 1994; Kitchener *et al.* 1978; Kitchener *et al.* 1980b). In temperate forest Bennett (1990b) found that six native species had become extinct and six introduced species had established populations following early settlement and clearing. Similar changes were recorded in south-eastern New South Wales, and in rainforest of the Atherton Tablelands (Laurance 1990; Lunney and Leary 1988). In addition to the local extinction of a number of mammal species, many others became rare and now have restricted distributions.

Some species seem to be far more vulnerable to the effects of fragmentation. While species such as the spotted-tailed quoll (*Dasyurus maculatus*), and Atherton antechinus (*Antechinus godmani*), decline or disappear rapidly following fragmentation, other species such as the bush rat (*Rattus fuscipes*), and brown antechinus (*Antechinus stuartii)* are more resilient and present in all patch sizes. (Bennett 1987; Laurance 1990; 1991; 1993; 1994; Pahl *et al.* 1988; Terborgh 1992). Traits and factors that underlie the variable responses of species include a species' natural rarity, its tolerance of the habitat matrix and edge effects, and its dispersal capacity (Laurance 1990; 1991; Terborgh 1992).

Terrestrial mammal species have a low probability of survival in urban environments due to their limited dispersal abilities across the urban matrix and between highly fragmented and isolated remnant patches (Dickman and Doncaster 1987; 1989; Goldingay and Sharpe 2004; How and Dell 1993; Tait *et al.* 2005). Roads and fences, for instance, present significant barriers to dispersal and there is increased risk of predation by cats, dogs and red foxes (Banks 2004; Barratt 1997; Dique *et al.* 2003; Forman 1999; Smith and Smith 1990).

The conversion of native habitat to agricultural or urban environments obviously alters the availability of resources in an environment, with consequences to the abundance and range of remaining fauna, including birds (Hobbs 1992; Saunders and Ingram 1997). The capacity of remnant vegetation patches to provide suitable habitat for birds depends upon the patch size, shape and connectivity (Garden *et al.* 2006; Radford *et al.* 2005), and the availability of species specific requirements (e.g. nesting and foraging). Overall, there is strong evidence that species richness increases in larger patches (Radford *et al.* 2005; Yamaura *et al.* 2008), particularly when comparing patches of vastly different sizes. However, in urban environments, a single patch of habitat may not be large enough to support the long-term persistence of some species. In areas with low overall habitat cover, the degree of connectivity may influence species richness to a greater extent than small increases in patch size (Radford *et al.* 2005).

In highly fragmented habitats, species persistence may depend upon the occurrence of several populations and dispersal between them (Fahrig and Merriam 1994). For example, the occurrence of connected habitat patches is important for the persistence of the Blue-breasted Fairy-wren in the fragmented Wheatbelt habitat (Brooker and Brooker 2001). In addition to population maintenance, birds may utilise different components of the landscape and thus require the ability to frequently traverse habitats in response to fluctuating resources, such as fruiting plants for frugivorous birds (Price *et al.* 1999 ).

## *Impacts on fauna of the Swan Coastal Plain*

Following their studies of remnant bushland on the Swan Coastal Plain, How and Dell (2000) concluded that most reptiles still retain viable populations. Despite the persistence of reptiles, they found that all reptile groups, except skinks, occur in greater numbers on the larger remnants of vegetation, and that large reptile species such as goannas and snakes with large home ranges have been the most affected species (Cooper 1995; How and Dell 1994; 2000). Although these studies provide evidence that herpetofauna are resilient to the changes, they may reflect the short temporal span of the studies. Long-term persistence of the populations in urban fragments may not be assured, particularly if barriers to dispersal result in reduced ability to recolonise a patch if extinction occurs.

In the GSS study area, reviews have indicated that up to 80 species of birds have declined in number since European settlement, including 13 species that have not been recorded since 1997 and can be considered as locally extinct or very rare (Chapter 3). In addition, in urban remnant fragments, species such as the scarlet robin (*Petroica multicolor*), golden whistler (*Pachycephala pectoralis*), splendid fairy-wren (*Malurus splendens*) and rufous treecreeper (*Climacteris rufa*) are now largely absent or locally extinct (How and Dell 1993). Of the 70 species of naturally occurring passerines on the Swan Coastal Plain, How and Dell (1993) identified 46 as having decreased as a result of habitat loss and fragmentation. The large-scale removal and alteration of wetlands has substantially affected wetland-associated bird species, such as the black bittern and Australasian bittern, both of which are now very rarely observed in the GSS study area. In addition, the removal of woodlands has reduced the breeding areas for several birds of prey, including the collared sparrowhawk and whistling kites (How and Dell 1993).

The impacts of fragmentation on mammals in urban remnants on the Swan Coastal Plain differ markedly from those described in small remnants of native vegetation in the Western Australian Wheatbelt where mammals persist in remnant vegetation despite widespread clearing of land for agriculture (How and Dell 2000). How and Dell (2000) observed only three ground dwelling mammal species, honey possum (*Tarsipes rostratus*), western pygmy possum (*Cercartetus concinnus*) and yellow-footed antechinus (*Antechinus flavipes*), with each of the last two species present only at one site located south of the Swan River. This contrasts with the situation observed for reptiles.

A comprehensive survey undertaken in 1978 confirmed the persistence of only 12 (of the historical 28 non-bat species) native mammal species in areas considered to be those most likely to still support extant species. This included large tracts of relatively undisturbed vegetation (Kitchener *et al.* 1978). These results indicate that fragmentation alone is not the primary factor contributing to the decline of mammals in the GSS study area. Significant factors identified as causing the lower survival of mammals on the Swan Coastal Plain, compared to reptiles, included higher human populations and the associated changes to fire regimes (high intensity or frequency), feral cat predation and increased impediments to dispersal and recolonisation of isolated patches (How and Dell 2000). The surveys undertaken for the GSS (2007–08) identified that at least some priority-listed mammals (e.g. quenda and rakali) are still persisting in some urban remnants in the GSS study area (see Chapter 3). However, without habitat connectivity between these urban remnants, the long-term persistence of these populations is unknown.

## Habitat loss and ecological thresholds

There is evidence that once habitat loss crosses a ‘threshold’, a substantial number of species are then lost from the landscape (Andren 1994; Fahrig 2002; Huggett 2005; Radford *et al.* 2005), potentially leading to an ecological change in state (Lawton *et al.* 1994). Although the concept of an ecological threshold has been predominantly driven by theoretical models, there is an increasing amount of research that supports it (Radford *et al.* 2005). Modelled simulation studies suggest a major ecological change of state occurs when habitat cover declines to between approximately 10% and 30% of the landscape (Andren 1994), and empirical studies have shown very strong evidence for sharp decline in species richness in landscapes with less than 10% habitat cover (Radford *et al.* 2005). However, the threshold level will undoubtedly vary according to taxon, community type, configuration and condition of remnant vegetation and the extent of additional disturbances (Fahrig 2002; Lindenmayer and Luck 2005).

The use of the ecological threshold concept in land-use policy is gaining popularity (Lindenmayer and Luck 2005). For example, in semi-arid regions in Queensland, a reduction in remnant vegetation to 30% at a regional scale has been estimated to result in the loss of 25% to 35% of the vertebrate fauna (McAlpine *et al.* 2002). This has lead to the proposal of minimum vegetation retention thresholds of 50% at a bioregional scale and 30% at a regional ecosystem scale (vegetation categories defined by landform type, soil and dominant plant species) (McAlpine *et al.* 2002). A 30% minimum retention threshold has also been adopted in the *National objectives and targets for biodiversity conservation 2001 to 2005* (Commonwealth of Australia 2001). The retention and protection of ecological communities also contributes to the establishment of a ‘comprehensive, adequate and representative’ (CAR) system of protected areas, as required under the National Strategy for the Conservation of Australia’s Biological Diversity (Commonwealth of Australia 1997).

Although the use of ecological thresholds in land use policy is increasing, land use decision makers need to be aware of issues associated with their use (Lindenmayer and Luck 2005). For example, even above the threshold, species richness will still decline with habitat loss. Thus the threshold represents the point at which species loss is exacerbated (Andren 1994), or an end point of species decline (Radford *et al.* 2005). The configuration of remaining vegetation in the landscape will also affect the threshold level as the degree of fragmentation strongly influences species loss (Fahrig 2002). Further, the retention of habitat needs to be representative of the variety of vegetation communities in the landscape (Lindenmayer and Luck 2005).

In the Perth metropolitan region, the representation and retention of vegetation complexes were key criteria in identifying significant sites as part of the Bush Forever process. In this planning process, a 10% threshold was used due to the considerable constraints associated with protecting areas in a heavily populated urban region (Government of Western Australia 2000b). Inherent in the Bush Forever process was the general presumption against clearing any vegetation complex with less than 10% remaining in the Perth metropolitan region portion of the Swan Coastal Plain (Government of Western Australia 2000b).

## Pine plantations and habitat fragmentation

Areas cleared for exotic softwood plantations have an impact on individual taxa and communities as they provide only low quality habitat for many native species. This is indicated by pine plantations being depauperate in native wildlife (Friend 1980). However populations can persist in remnant patches of native vegetation within the plantation matrix (Lindenmayer *et al.* 1999). Ecological and genetic studies have been undertaken on a range of native taxa to understand the impacts of fragmentation on species survival in plantations of south-eastern Australia (Banks *et al.* 2005; Lindenmayer *et al.* 1999; Lindenmayer and Peakall 2000; Peakall *et al.* 2003). These studies showed that although the remnant habitat patches comprised approximately 2% of the study area, they can support non-isolated populations. However, the distribution of species is dependent on habitat quality and patch connectivity (Banks *et al.* 2005).

In Victoria, pines, particularly in younger aged stands (< 5 years) support few bird species and their richness, abundance and species composition are lower than in eucalypt woodlands (Friend 1982). Pines support generalist species with life histories that include an insectivorous diet, foraging from the ground or low shrubs and the ability to nest in open areas. Canopy feeders and eucalypt specialists are absent (Friend 1982).

A recent study in New South Wales confirmed these trends, finding a lower frequency of occurrence of birds in pines compared to nearby eucalypt woodland (Lindenmayer *et al.* 2003). Proximity to native vegetation is important, with bird frequency and diversity in pines being higher where pine stands bordered native forest. Lindenmayer *et al.* (2003) found that seven native bird species had higher detection rates in pine plantations and a further 15 generalist species were more common in pine plantations than native vegetation and that a number of other species used pines as part of their foraging range, although most still required bushland for breeding, feeding or shelter.

Another study also showed reduced bird diversity in pines as opposed to native vegetation and reported that the area of surrounding eucalypt forest had a more important influence on the avifaunal assemblage of pines than any factors within the pine forests (Lindenmayer *et al.* 2002).

## Conservation management in fragmented landscapes

The likelihood and practicality of managing all taxa and ecological processes in fragmented habitats and landscapes is very low. Approaches to mitigate the adverse effects of human landscape changes have focused on employing management that is beneficial to many species and processes simultaneously, because even in the absence of detailed knowledge such approaches are likely to provide many benefits.

A primary approach has been to maintain and increase areas of large intact and structurally complex native vegetation and habitat. Other approaches involve improvement or restoration of habitat quality, and enhancement or restoration of the matrix, with buffers of habitat quality. There has also been a focus on increasing landscape connectivity where different types of landscape components may contribute (Lindenmayer and Fischer 2006). These include wildlife corridors, which involve physical linkages between patches of native vegetation (Bennett 1990a), and ‘stepping stones’ represented by small patches of native vegetation scattered over the landscape that together may facilitate habitat connectivity for species. In addition the role of a ‘soft’ matrix which has similar vegetation structure to the native vegetation is considered important.

Corridors have been identified as a useful way to restore the connectivity of fragmented landscapes and as a conservation strategy. There is extensive literature on corridors and various reasons for developing or retaining corridors. (Andrews 1993) described the objectives of a corridor as being to:

- enable colonisation of new sites as they become suitable
- allow wildlife to move out of sites that become unsuitable
- permit recolonisation of extinct populations
- promote species to move between different areas as required in their lifecycle
- increase the overall extent of habitat within an area.

The purpose and usefulness of corridors have been debated extensively. Existing patterns of landscape structure constrain design and application of corridors as their configuration is often determined by the patterns of remaining habitat, or fragments that appear to be connected by humans in an often simplified binary assessment of the landscape (remnant habitat, non-remnant) (Chetkiewicz *et al.* 2006). This is often because fragments are tangible structural entities which are easy to identify in the landscape. However, such human perceptions are not necessarily those of other organisms (McIntyre and Hobbs 1999). This means that there are often large differences between a human-defined landscape and the landscape perceived by other species.

Furthermore, there is evidence that linear strips of vegetation (traditional corridors) have a number of aspects detrimental to biodiversity conservation. Rates of nest predation from bird predators are higher in linear remnants as opposed to large remnants (Major *et al.* 1999b). The abundance and density of birds is often also higher in large, non-linear remnants as opposed to narrow corridors (Major *et al.* 1999a). However, some studies have found that linear corridors are used extensively by bushland-dependent migrants (Bentley and Catterall 1997). It has also been proposed that long, narrow remnants may lie across a range of environmental gradients and therefore contain more vegetation types and habitats than a more circular remnant of similar area (Saunders *et al.* 1991).

There is consensus that corridors do provide a useful function in connecting fragmented populations, for some species, and tend to increase the diversity and abundance of species in connected patches (Arnold and Weeldenburg 1998; Drinnan 2005; Haas 1995). Whilst information on the movement of individual species across the landscape is required to create successful landscape connectivity, this is difficult and time-consuming to obtain. Therefore an adaptive management approach should be undertaken based on the approach that corridors do have value for biotic movement (Saunders *et al.* 1991). Whilst the required data are gathered, attempts should be made to create and retain corridor networks wherever possible. This is supported by an increasing number of studies which indicate that corridors are of value for movement, at least for a subset of biota.

Many factors influence the use of habitat corridors by wildlife (Lindenmayer and Fischer 2006). Some of these factors are:

- the individual characteristics of species, such as their ecology, patterns of behaviour and social structure
- the landscape characteristics of corridors, including corridor size and shape, their topographic location and the size of connected patches
- the vegetation attributes within the corridor and the availability of food resources.

It is essential to consider the purpose of the corridor including the species it is intended to benefit. In addition, management needs to carefully consider how to maintain or apply appropriate disturbance regimes, how to minimise or control threatening processes such as invasive species and predators.

The function of a corridor needs to be decided prior to its design. While all corridors increase the habitat values of an area, narrow corridors may only be suitable for rapid movements, whereas a wide corridor allows for the restoration of a complete range of community and ecosystem processes and enables movement of organisms between areas (Hess and Fischer 2001). These wide corridors are often termed landscape linkages and provide regional connectivity (Hess and Fischer 2001). If properly designed they can allow a complete range of community and ecosystem processes and also the movement of organisms between areas, over generations.

Conservation strategies have examined the development of habitat networks that comprise relatively large patches of intact habitat that are sufficiently connected to each other to facilitate gene flow and create viable metapopulations (Hanski 1994). Recent studies to develop such linkages and networks have focused on maximising the benefits to biodiversity and minimising costs by building efficient habitat networks. For example, Drielsma *et al.* (2007b) describe a spatial habitat modelling methodology for evaluating the contribution and potential contribution of connecting paths to landscape connectivity based on a spatial links tool, which maps link value across a region. This tool combines connectivity measures from metapopulation ecology with the least-cost path algorithm from graph theory. It thus produces a map of link value. It does not quantify the potential benefit to conservation but potential corridors identified can be assessed as part of an overall land-use scenario.

Another approach has utilised cost–benefit modelling that combines the costs of movement to organisms with the benefits of access to habitat (Drieslma *et al.* 2007a). The model employs habitat suitability mapping to determine benefits to the species, and habitat permeability that reflect the species' movement abilities. Permeability involves determining the probability of movement success. The methods can be employed to assess potential habitat linkages within a landscape or to evaluate connectivity across regions.

## Ecological linkages on the Swan Coastal Plain and the Gnangara groundwater system

The remnant vegetation of the GSS study area retains some of the largest intact areas of native vegetation on the Swan Coastal Plain. Although these tracts offer high quality habitat for a range of native species, current and proposed land-uses have fragmented, or will fragment, this vegetation into smaller remnants that are further isolated by urban development, agricultural land, major roads and highways. Management aimed at limiting the impacts of such human landscape changes is currently focused on managing in a manner that benefits many species and ecological processes simultaneously, such as increasing landscape connectivity through ecological linkages.

The concept of ecological linkages is not new to the GSS study area, as various linkage schemes have been previously proposed. Over the last ten years local and regional linkages have been identified in various biodiversity conservation and land use planning strategies. One example is the *Avon Arc Sub-regional Strategy* which included a regional greenway system which aimed to 'protect and secure existing major vegetated areas, principal rivers and river valleys and to establish linkages between the major components'

(Western Australian Planning Commission 2001, pg 72). Another is the Perth Biodiversity Project which established principles for ecological linkages within the Perth region at both a local and landscape scale. This project recognised that effective regional linkages needed to incorporate variation in faunal and floral diversity that is typical of the region so that linkages could be utilised by the greatest range of species possible (Del Marco *et al.* 2004). These principles formed the basis of the proposed network of regional linkages and enabled local governments to identify and create local linkages that supported them.

Ecological linkages currently proposed for the GSS study area generally consist of discrete patches of remnant vegetation amongst a matrix of land uses. The GSS proposes to protect these existing blocks of remnant bushland and to connect them with strategic ecological linkages, thereby strengthening the ecological resilience of the system to multiple threats. This would be achieved by forming ecological linkages that create a connection between larger areas of remnant vegetation (often conservation reserves), and create connectivity in both a north-south and east-west direction, whilst also conserving the higher biodiversity assets.

To assess the benefits and disadvantages that may be associated with the implementation of ecological linkages, research should be undertaken both prior to, and in conjunction with their implementation and restoration. These studies could be conducted at different time intervals after implementation to help gain a greater understanding of what species use the linkages and of any successional changes. As it is difficult to assess the use of linkages by all fauna and flora species, the use of a suite of 'focal species' will help identify which species might be using, and benefiting from, the linkages. The requirements of species from this sub-group should encompass those of the rest of the biota in the area (Lambeck 1997). Including species with a range of different spatial and compositional attributes, will allow an accurate representation of the range of biota in the area to be made. Additionally, species richness and species composition can be used to determine the advantages and disadvantages associated with ecological linkages.

Several species are likely to benefit from the connectivity provided by ecological linkages. These include some species which have already been studied through the GSS, including the variety of nectar-dependent bird species that disperse throughout the northern *Banksia* woodlands and the endangered Carnaby's black-cockatoo (*Calyptorhynchus latirostris*)

(See case study below). In addition, the native honey possum (*Tarsipes rostratus*), rakali (*Hydromys chrysogaster*), quenda (*Isoodon obesulus*) and the long-necked tortoise (*Chelodina oblonga*) are also species likely to benefit from increased connectivity of remnant vegetation. Studies which assess the use of linkages by these particular species may aid in the determination of benefits from a broad management scheme such as ecological linkages.

### *Case study of Carnaby's black-cockatoo*

There is a need to assess the potential impacts on the removal of pine plantations on Carnaby's black-cockatoo, and any benefits from the connectivity provided by the proposed ecological linkages. A GSS consideration is rehabilitating sections of the pine plantations (see DEC–projects section below). Remnant patches of native vegetation are scattered throughout the pine plantations, and these may provide a focal point for rehabilitation efforts.

Carnaby's black-cockatoo (*Calyptorhynchus latirostris*) is a threatened species, endemic to south-west Western Australia, and is listed as Schedule 1 under State (*Wildlife Conservation Act 1950*) legislation and as endangered under Commonwealth (*Environment Protection and Biodiversity Conservation Act*) legislation. Since the 1950s, the range of Carnaby's black-cockatoo has contracted by more than 30%, the species has disappeared from more than a third of its breeding range, and the population is estimated to have halved (Garnett and Crowley 2000; Mawson 1995; Saunders and Ingram 1998). The major threatening process affecting Carnaby's black-cockatoo is the removal of nesting and food resources via habitat loss and fragmentation (Cale 2003).

Carnaby's black-cockatoo is principally a migratory parrot, spending the breeding months (July–December) in the Western Australian Wheatbelt region and moving southwards and/or westwards towards the coast during the non-breeding months (Saunders 1980; 1990). Breeding pairs exhibit nesting site fidelity (returning to the same nest site area), and utilise nest hollows within large *Eucalyptus* species in woodland habitat (Saunders 1990). However, the Western Australian Wheatbelt has lost nearly 90% of native vegetation (Hobbs 1993), and much of the nesting and food resources now available for Carnaby's black-cockatoo remain in small fragmented patches.

Carnaby's black-cockatoos have been observed feeding from ~ 60 native plants, but principally they forage on seeds from the Proteaceae and Myrtaceae family, and occasionally flowers, nectar and insect larvae (Higgins 1999; Valentine and Stock 2008). Important native food plants in the GSS study area include *B. attenuata, B. menziesii, B. grandis, B. sessilis, B. prionotes, B. ilicifolia, Corymbia callphyla* and *Eucalyptus marginate*. However, Carnaby's black-cockatoo also forage upon a number of introduced plant species, such as *Erodium* spp. (wild geranium) in the Wheatbelt, and crop or forestry species such as *Pinus pinaster* (maritime pine).

The GSS study area represents a critical non-breeding foraging area for Carnaby's black-cockatoo (Saunders 1980; Shah 2006; Valentine and Stock 2008). However, urbanisation and agricultural development has resulted in the removal of large sections of native vegetation. In addition, there has been a major shift in the dietary composition of Carnaby's black-cockatoo, from a traditional diet of mostly native seeds and nectar to seeds of pine from plantations (Higgins 1999; Perry 1948; Saunders 1974; 1980). The strong reliance on the pine plantations in the GSS study area as food and roosting habitat (Finn *et al.* 2009; Saunders 1980; Shah 2006), has lead to concern that the removal of pine plantations will impact populations (Cale 2003; Garnett and Crowley 2000).

The standing crop of pine plantations in the GSS study area is estimated to support ~ 2800 birds for six months and *Banksia attenuata* woodland in the Bassendean and Cottesloe North vegetation complexes of DEC ~ 4900 birds for six months (see Valentine and Stock 2008). The pine plantations represent an important foraging resource as there is a minimum estimated population of 4500 birds in the Perth region (Shah 2006), and 3000–4000 within the GSS study area (Finn *et al.* 2009). Behavioural observations of the birds within the GSS study area pine plantations indicate that ~ 3000 birds are concentrated within and adjacent to the pines between January and March (Finn *et al.* 2009). Although the studies did not concentrate within the large remnant arc of *Banksia* spp. woodland, foraging trace evidence indicates that these food resources are also exploited (Valentine and Stock 2008). In order to assess the potential impacts of the removal of pines on Carnaby's black-cockatoo, we need to understand the availability and spatial variability of food resources in remnant native vegetation.

Another GSS consideration following the removal of pine plantations is rehabilitating the portion of current pine plantations (> 20 000 ha) with native vegetation. Remnant patches of native vegetation are scattered throughout the pine plantations and these may provide a focal point for rehabilitation efforts. Creating 'Corridors for Carnaby's' may be necessary to offset the loss of food and nesting resources currently provided by the pine plantations. To benefit Carnaby's black-cockatoo it is likely that such corridors will need to provide both foraging plant species and potential roost species. In addition, rehabilitation should ideally begin in conjunction with the removal of pine plantations.

## Summary of DEC–GSS projects (2007-09)

### Project 1. Status of remnant vegetation

Technical report reference: Kinloch, J., Valentine, L.E. and Sonneman, T. (2009) *Status of vegetation complexes in the Gnangara Sustainability Strategy study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

This project assessed the current extent and levels of protection of remnant vegetation across the GSS study area and evaluated the size and shape of the remnant vegetation patches. We also quantified the pre-European and current level of protection and retention of vegetation complexes, at both the regional (Swan Coastal Plain IBRA region) and GSS study area scale, to identify what vegetation complexes require additional areas to be retained, protected and rehabilitated within the GSS study area.

Remnant vegetation mapping was combined into a single spatial dataset and modified to ensure polygons represented the boundaries of remnant vegetation patches. For each patch of remnant vegetation area (ha), perimeter (m) and perimeter to area ratio were calculated. Area and perimeter to area ratios were divided into classes allowing the patches to be ranked.

The vegetation complexes defined by Heddle *et al*. (1980) (see Chapter 2) were accepted as a base for quantifying and interpreting representation of biodiversity. The vegetation complexes delineate broad vegetation classes and have associated spatial mapping depicting pre-European extent, enabling the level of representation and retention for each complex to be assessed over its natural state. However, it is likely that some of the vegetation complexes found within the GSS study area occur in unmapped areas, and hence the level of representation may be underestimated.

### Level of fragmentation – patch size and shape

Analysis of remnant vegetation patches revealed that most remnants in the south-west and east of the GSS are small and highly fragmented, whereas large intact areas remain in the north and central areas with the largest patches occurring in an arc to the north and east of the pine plantations (Figure 6.2). The areas and shapes of the remnant vegetation patches varied greatly and hence their perimeter to area ratios did too (Figure 6.3). Many of the very small patches which appear in Figure 6.2 (very small area and very high perimeter to area ratio) are actually artefacts of the spatial analysis (Kinloch *et al* 2009). Figure 6.3 shows the distribution of remnant vegetation patches by area and by perimeter to area ratio.

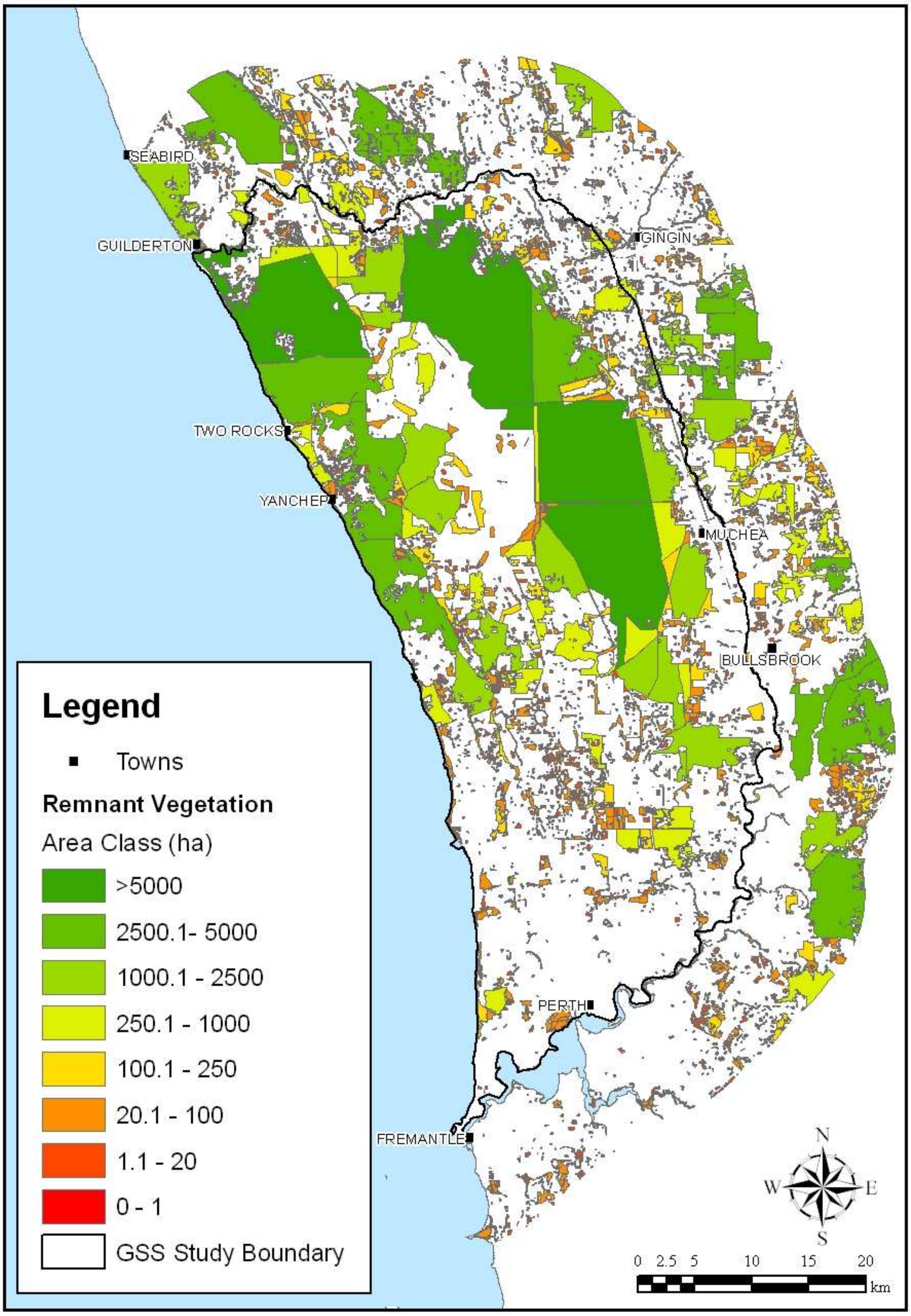


Figure 6.2: Ranking of remnant vegetation patches according to area class within the GSS study area and additional 10 km buffer.

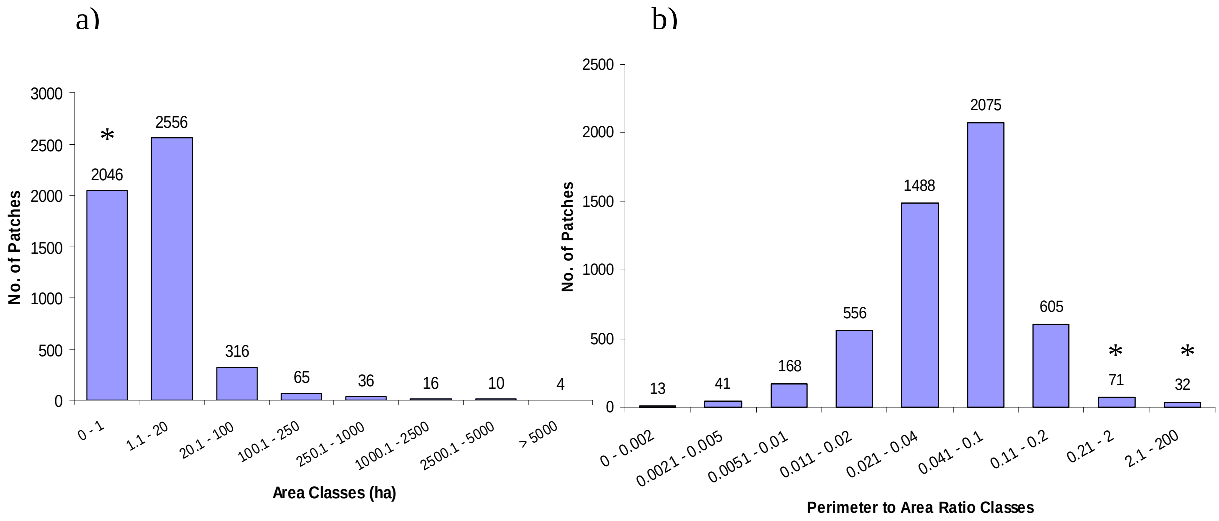


Figure 6.3: Distribution of remnant vegetation patches by area and perimeter to area ratio classes (* majority are artefacts of the spatial analysis).

## Current extent and levels of protection of remnant vegetation

One-third of the native vegetation remaining in the GSS study area is protected. An additional 31% occurs within tenure that precludes large scale clearing (state forest and Unallocated Crown Land) and a further 11% is proposed to be protected (Table 6.1). Of those areas protected, most are managed by the DEC. Bush Forever identified > 8000 ha of remnant vegetation, occurring outside existing protected areas, as being regionally significant (Government of Western Australia 2000a). These Bush Forever sites are on public and private land and the protection status of only a few is yet to be finalised.

Table 6.1: The extent of remnant vegetation and protection status in the GSS study area in 2005–2006.

| Level of protection | Tenure | Total area of remnant vegetation (ha) | Proportion of remnant vegetation (%) | Proportion of total area of GSS (%) |
|---|---|---|---|---|
| Protected for conservation | Major parks, nature reserves, other reserves vested in the Conservation Commission (CALM Act) | 25 950 | 25.6 | 12.1 |
| | Bush Forever – additional to above | 8195 | 8.1 | 3.8 |
| | **Sub-total** | **34 145** | **33.6** | **15.9** |
| Some level of protection | Bush Forever (additional nominations) | 69 | 0.1 | 0.0 |
| | State forest and other DEC managed lands (vesting purpose not conservation) | 11 809 | 11.6 | 5.5 |
| | Proposed for vesting as a conservation reserve (currently state forest) | 11 490 | 11.3 | 5.3 |
| | Unallocated Crown Land | 19 218 | 18.9 | 8.9 |
| | **Sub-total** | **42 586** | **41.9** | **19.8** |
| Not protected for conservation | Other Crown reserves outside Metropolitan Regional Scheme | 670 | 0.7 | 0.3 |
| | All other tenures (not protected through Bush Forever) | 24 143 | 23.8 | 11.2 |
| | **Sub-total** | **24 813** | **24.4** | **11.5** |
| **Total area of remaining remnant vegetation** | | **101 544** | **100.0** | **47.3** |
| **Total area cleared** | | **113 352** | | **52.7** |

### Levels of retention of vegetation complexes at the regional (Swan Coastal Plain) and GSS study area scale

Two complexes (Beermullah and Guilford) have < 10% retained and an additional seven (Bassendean Central and South, Herdsman, Karakatta Central and South, Pinjar, Southern River, Swan and Yanga) have < 30% retained across the Swan Coastal Plain (Table 6.2). To prevent loss of species within these ecological communities and to satisfy current policy, all of these complexes require additional protection on the Swan Coastal Plain, including within the GSS study area. Of particular importance, for retention and protection in the GSS study area, are those vegetation complexes with > 40% of their pre-European extent, across the Swan Coastal Plain, within the GSS study area (Herdsman, Karrakatta Central and South, Pinjar and Yanga complexes). Very small amounts (< 10 ha) of the largely cleared Beermullah and Guilford complexes are unprotected in the GSS study area (Table 6.2). Rehabilitation of these complexes is needed to increase their levels of retention and protection.

Although nearly 50% of remnant vegetation remains within the GSS study area, the degree of retention varies considerably between vegetation complexes. Eleven vegetation complexes have < 30% remaining, and typically occur in areas that have been largely cleared for urban and agricultural development (e.g. Pinjarra Plain, southern Bassendean dunes, central Wetlands; Table 6.2; Figure 6.4). Nine of these have inadequate levels of retention at the Swan Coastal Plain scale (see above) and three (Vasse Complex, Beermullah Complex and Swan Complex) have < 10% remaining in the GSS. These communities require protection of all of the remaining remnant vegetation within the GSS.

Only three of the 21 GSS vegetation complexes have > 30% of their remnant vegetation protected (Figure 6.2). These are the Bassendean Central and South Transition, Bassendean North Transition and Karakatta North Transition complexes.

Table 6.2: Current (2005–06) and pre-European extent of vegetation complexes in the Swan Coastal Plain and GSS study area and ranking in order of priority for retention, protection and rehabilitation. Values in brackets are proportions (%) of the pre-European extent for each category.

| **Landform** | **Vegetation complex** | **Current extent in the Swan Coastal Plain** | | **Pre-European extent in the GSS** | | **Current extent in the GSS** | | **Current extent protected in the GSS** | | **Current extent with some level of protection in the GSS** | | **Current extent not protected in the GSS** | | **Extent proposed for formal conservation protection in FMP[(1)]** | **Ranking** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **ha** | **%** | **ha** | **%** | **ha** | **%** | **ha** | **%** | **ha** | **%** | **ha** | **%** | | |
| Quindalup Dunes | Quindalup | 30 129 | 58 | 15 843 | 30 | 9614 | 61 | 1804 | 11 | 1973 | 12 | 5837 | 37 | 0 | 3 |
| Spearwood Dunes | Cottesloe - central and south | 17 529 | 39 | 21 593 | 48 | 8381 | 39 | 3575 | 17 | 889 | 4 | 3917 | 18 | 0 | 3 |
| | Cottesloe - north | 25 304 | 58 | 21 399 | 49 | 15 461 | 72 | 5038 | 24 | 8802 | 41 | 1621 | 8 | 5644 | 3 |
| | Karrakatta - central and south | 12 791 | 26 | 24 284 | 49 | 3484 | 14 | 1348 | 6 | 323 | 1 | 1813 | 7 | 0 | 2 |
| | Karrakatta - north | 19 586 | 44 | 15 365 | 35 | 5868 | 38 | 778 | 5 | 4050 | 26 | 1040 | 7 | 411 | 3 |
| | Karrakatta – north transition | 4751 | 90 | 5260 | 100 | 4751 | 90 | 2102 | 40 | 2648 | 50 | 0 | 0 | 0 | 2 |
| Marine (estuarine and lagoonal) deposits | Vasse | 3778 | 34 | 549 | 5 | 6 | 1 | 5 | 1 | 0 | 0 | 1 | 0 | 0 | 3 |
| Wetlands | Herdsman | 2559 | 26 | 4144 | 43 | 996 | 24 | 770 | 19 | 0 | 0 | 226 | 5 | 0 | 2 |
| | Pinjar | 1140 | 23 | 4893 | 100 | 1140 | 23 | 905 | 18 | 63 | 1 | 172 | 4 | 0 | 2 |
| Combinations of Quindalup/Spearwood/ Bassendean Dunes | Moore River | 2979 | 35 | 797 | 9 | 267 | 34 | 0 | 0 | 0 | 0 | 267 | 34 | 0 | 3 |
| Bassendean Dunes | Bassendean - central and south | 24 678 | 28 | 10 437 | 12 | 1923 | 18 | 1566 | 15 | 97 | 1 | 260 | 2 | 0 | 2 |
| | Bassendean - central and south transition | 2176 | 100 | 2178 | 100 | 2176 | 100 | 2175 | 100 | 1 | 0 | 0 | 0 | 0 | 4 |
| | Bassendean - north | 57 054 | 72 | 51 920 | 66 | 34 705 | 67 | 10 194 | 20 | 18 878 | 36 | 5633 | 11 | 4738 | 2 |
| | Bassendean – north transition | 18 510 | 89 | 7789 | 37 | 6687 | 86 | 2845 | 37 | 3643 | 47 | 199 | 3 | 6 | 4 |
| | Caladenia | 5309 | 55 | 277 | 3 | 49 | 18 | 0 | 0 | 0 | 0 | 49 | 18 | 0 | 3 |
| Combinations of Bassendean Dunes / Pinjarra Plain | Southern River | 12 238 | 21 | 7490 | 13 | 1429 | 19 | 1047 | 14 | 0 | 0 | 382 | 5 | 0 | 2 |

| Landform | Vegetation complex | Current extent in the Swan Coastal Plain | | Pre-European extent in the GSS | | Current extent in the GSS | | Current extent protected in the GSS | | Current extent with some level of protection in the GSS | | Current extent not protected in the GSS | | Extent proposed for formal conservation protection in FMP(1) | Ranking |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Pinjarra Plain | Beermullah | 436 | 6 | 1000 | 15 | 87 | 9 | 81 | 8 | 0 | 0 | 6 | 1 | 0 | 1 |
| | Guildford | 4870 | 5 | 486 | 1 | 91 | 19 | 83 | 17 | 0 | 0 | 8 | 2 | 0 | 1 |
| | Swan | 2239 | 13 | 1741 | 10 | 83 | 5 | 48 | 3 | 0 | 0 | 34 | 2 | 0 | 1 |
| | Yanga | 5164 | 20 | 16 321 | 62 | 3680 | 23 | 482 | 3 | 73 | 0 | 3125 | 19 | 36 | 1 |
| Gingin Scarp | Coonambidgee | 2865 | 46 | 448 | 7 | 336 | 75 | 0 | 0 | 0 | 0 | 336 | 75 | 0 | 1 |
| | **Total** | **256 085** | | **214 214** | | **101 212** | | **34 846** | | **41 441** | | **24 925** | | **10 835** | |

(1) Extent of state forest proposed for formal conservation protection in the Forest Management Plan (Conservation Commission of Western Australia 2004).

Vegetation complexes can be ranked in terms of need for retention, protection and rehabilitation based on current levels of retention and protection. Ranks were assigned using the following criteria:

- Rank 1 < 10% retained on the Swan Coastal Plain, or < 10% in the portion of the Perth metropolitan region. Retention and protection of remaining areas is a high priority. Rehabilitation should be especially considered along ecological linkages.
- Rank 2 < 30% retained Swan Coastal Plain, or < 400 ha retained Swan Coastal Plain, or > 60% of pre-European extent is within the GSS. Retention and protection of remaining areas is a priority. If rehabilitation is going to be undertaken, ecological linkages should be targeted.
- Rank 3 < 30% protected in the GSS. Retention and protection of remaining areas is a priority.
- Rank 4 do not meet any of above criterion. No additional protection is required.

Ranking 1 contains five vegetation complexes. These are Beermullah, Guildford, Swan, Yanga and Coonambidgee, which all occur in the extensively cleared areas in the east of the study area on the Pinjarra Plain and Gingin Scarp (Table 6.2, Figure 6.4). Additional protection is vital in order to maintain ecological linkages with the existing conservation estate to the west and east. The forest management plan is proposing that 36 ha of the Yanga vegetation complex be protected for conservation. Opportunities to rehabilitate Beermullah, Guildford and Swan complexes along Ellen Brook and the Swan River should be investigated.

Four of the vegetation complexes in rank 2 have both low levels of retention and limited scope for protection of additional areas. These are Herdsman, Bassendean Central and South, Karrakatta Central and South and Southern River. Formal protection of unprotected areas is a priority for these complexes and could be achieved by retaining and rehabilitating areas within ecological linkages and surrounding existing remnants. The Pinjar complex is also in rank 2 and has low levels of retention. This complex occurs only within the GSS and therefore is a high priority for protection. The high conservation significance of this vegetation complex was recognised by Bush Forever and the Western Australian Planning Commission subsequently purchased land in Lake Pinjar to increase the area protected. Also in rank 2, the Karrakatta North Transition and Bassendean North vegetation complexes both have high levels of retention in the GSS and have adequate levels of protection. These complexes are also a high priority for protection within the GSS as they either occur only within the GSS (Karrakatta North Transition) or occur largely within the GSS (Bassendean North). Additional areas of Bassendean North have been proposed to be protected in the forest management plan.

Many of the vegetation complexes ranked 3 require better protection (Table 6.2). Worthy of mention are the Quindalup and Cottesloe Central and South complexes that currently have sufficient levels of retention within the GSS but most of the unprotected areas are zoned for urban or industrial development. Therefore the protection of these vegetation complexes should be carefully considered during the development of structure plans to ensure that the 30% protection threshold can be satisfied, and that areas are retained in a configuration that maintains connectivity of existing protected areas.

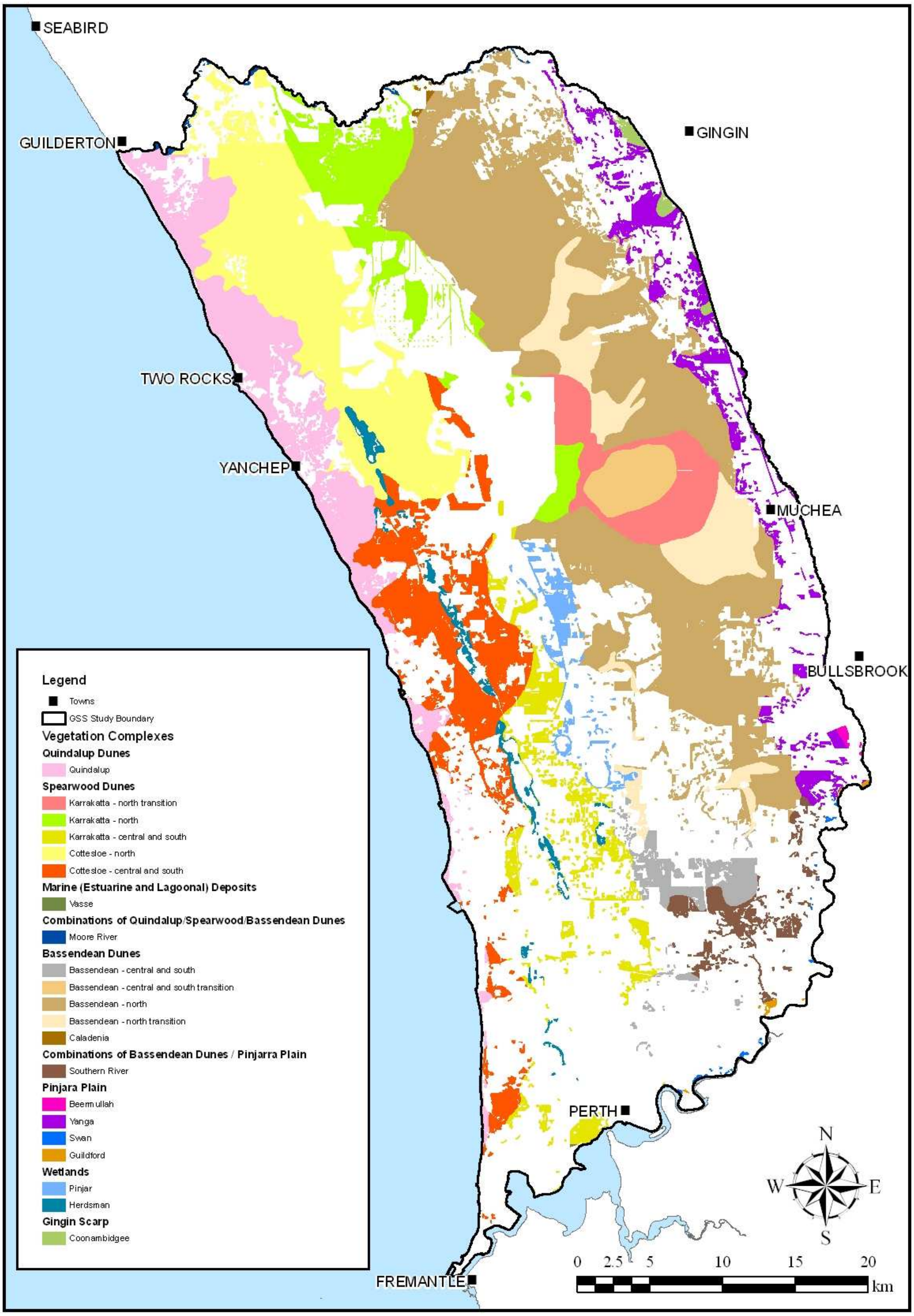


Figure 6.4: Current extent of remnant vegetation complexes within the GSS study area.

# Project 2. Ecological linkages

Technical report references:

1) Brown, P.H., Davis, R., Sonneman, T. and Kinloch, J. (2009). *Ecological linkages proposed for the Gnangara groundwater system*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

2) Brown, P.H., Sonneman, T. and Kinloch, J. (2009). *Ecological linkages within the pine plantations on the Gnangara groundwater system*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

The remnant vegetation of the GSS study area offers high quality habitat for a range of native fauna species. However, current and proposed land uses (e.g. urban development, agriculture) are likely to lead to further fragmentation. Additionally, the removal of 23 000 ha of pines by 2029 will result in land requiring either rehabilitation, or reassessment for other land use. A significant consideration of the GSS is the proposal to design ecological linkages that promote landscape-level connectivity at a sub-regional level, by improving connectivity of remnant vegetation across the landscape, whilst focusing on key biodiversity assets. The ecological linkages proposed for the pine plantations, post-harvest, have been included.

As part of this project, a review compiled all linkages previously identified by land-use planning and biodiversity strategies on the Gnangara groundwater system and highlighted gaps in the linkage network. For example, few linkages were designated in northern areas outside the Bush Forever study area (Shires of Chittering and Gingin), and there were few in the coastal areas north of Alkimos and south of Guilderton. Although the latter areas currently have > 60% remnant vegetation, the land is privately owned and zoned for urban development. Linkages therefore need to be identified so they can be included in future planning.

Information on the landscape requirements of sensitive avifauna species on the Swan Coastal Plain (Davis *et al.* 2008) identified that a threshold of ~ 60% total vegetation cover within a 2 km square area was required to support the most sensitive species (scarlet robin) and there was evidence that this threshold would provide adequate habitat for most bush birds. Therefore, areas within and adjoining the GSS study area that have > 60% remnant vegetation cover were designated ‘core’ areas. The project thus identified ecological linkages to join these.

A desktop assessment of spatial information was undertaken to identify new linkages. This process considered:

- remnant vegetation cover
- vegetation complex type
- the level of retention and protection of vegetation complexes
- waterways
- protection status of land parcels
- zoning of land under the Perth metropolitan region and town planning schemes
- orthophotos
- in the pine plantations
    - the condition of remnant vegetation patches
    - other information relating to ecological value of these patches.

Rehabilitation costs were also compiled for the vegetation patches.

A number of other factors were considered when plotting new ecological linkages. These included maximising the number of viable remnants (conservation reserves, Bush Forever sites and other bushland patches) along the linkage so as to minimise the need for re-vegetation. Where possible ecological linkages were routed so that the ends of the linkage were in an area protected within the DEC conservation estate or a Bush Forever site. Vegetated waterways and drainage lines were identified as linkages because they are unique ecosystems in the GSS, which form natural linkage corridors. Remnants of a vegetation complex which has low levels of retention and protection across the Gnangara groundwater system and Swan Coastal Plain were also given priority for inclusion within an ecological linkage. Linkage sites within pine plantations were chosen to include the larger high ecological value remnant vegetation patches within them.

In total, 19 ecological linkages are proposed (Figure 6.5), covering 15 500 ha. Of this, 60% requires complete rehabilitation. Establishment techniques and costing for such rehabilitation are still to be determined, and the adaptive management details will require further investigation. The type of vegetation established in the 'parkland' areas in between the linkages (post-pine) is of particular importance to the success of these linkages.

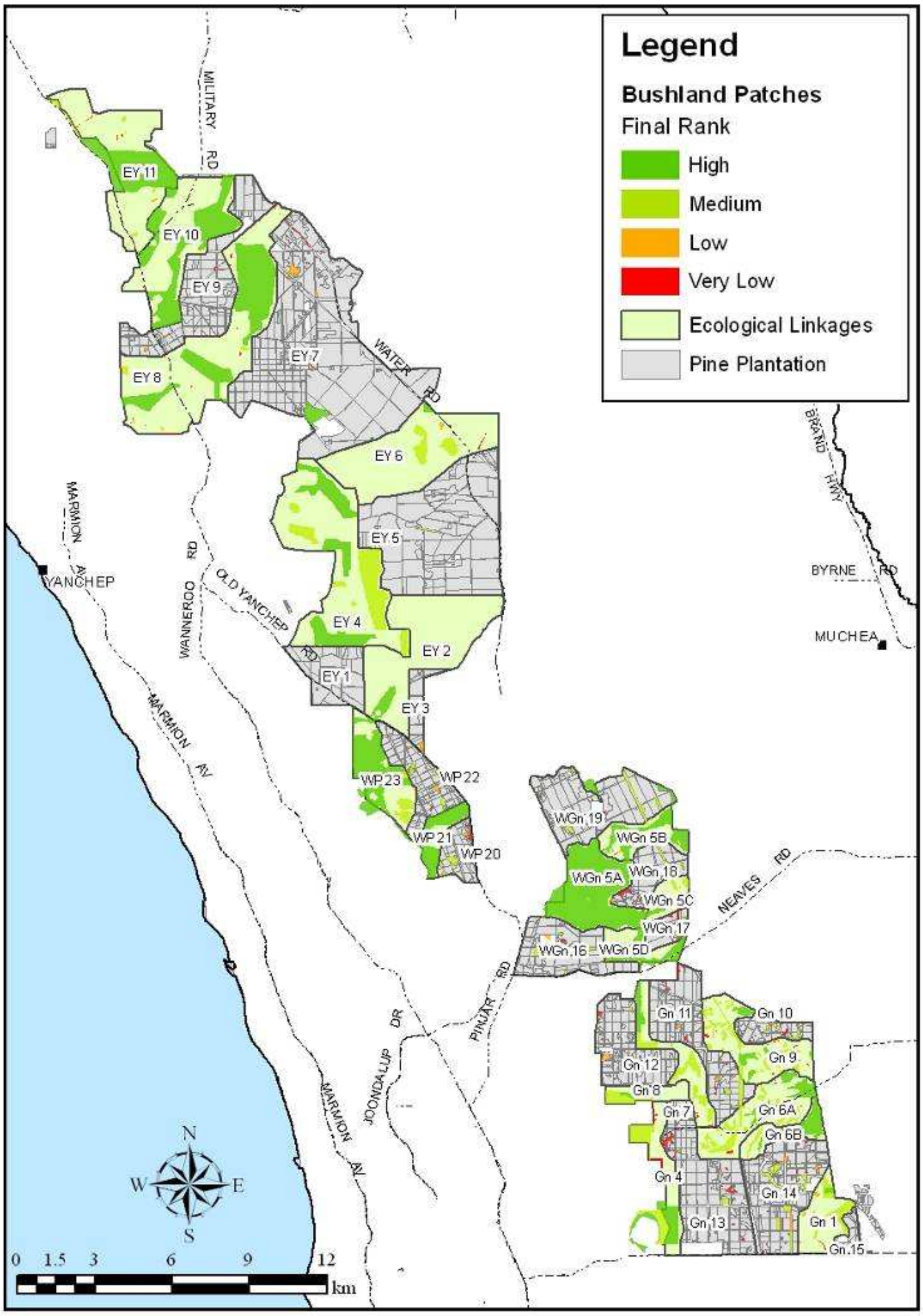


Figure 6.5: Proposed ecological linkages and associated remnant vegetation patches following removal of pine plantations.

The resulting proposed ‘Gnangara ecological linkage framework’ (Figure 6.6) thus has four major components:

- Core – areas of the landscape that have > 60% remaining native vegetation
- Linkage sites – Bush Forever sites associated with ecological linkages
- Conceptual linkage – proposed ecological linkages based on past studies and new linkages. The two sites labelled ‘area for conceptual linkage’ require more work to determine the preferred alignment of the linkage.
- Post-pine linkage – ecological linkages for the 23 000 ha of pine plantation.

The 500 m wide ‘conceptual linkages’ (blue lines in Figure 6.6) are the components of the proposed ecological linkages that are not currently protected (in reserves or Bush Forever sites). Therefore they will require acquisition, covenants with land owners and often, rehabilitation.

These conceptual linkages will need to be assessed individually to determine the exact on-ground boundaries, based on remnant vegetation, land use and availability for purchase. Where possible any remnant vegetation or local natural areas should be retained in their entirety, rather than just the portion of these areas which fall within the mapped 500 m wide linkage.

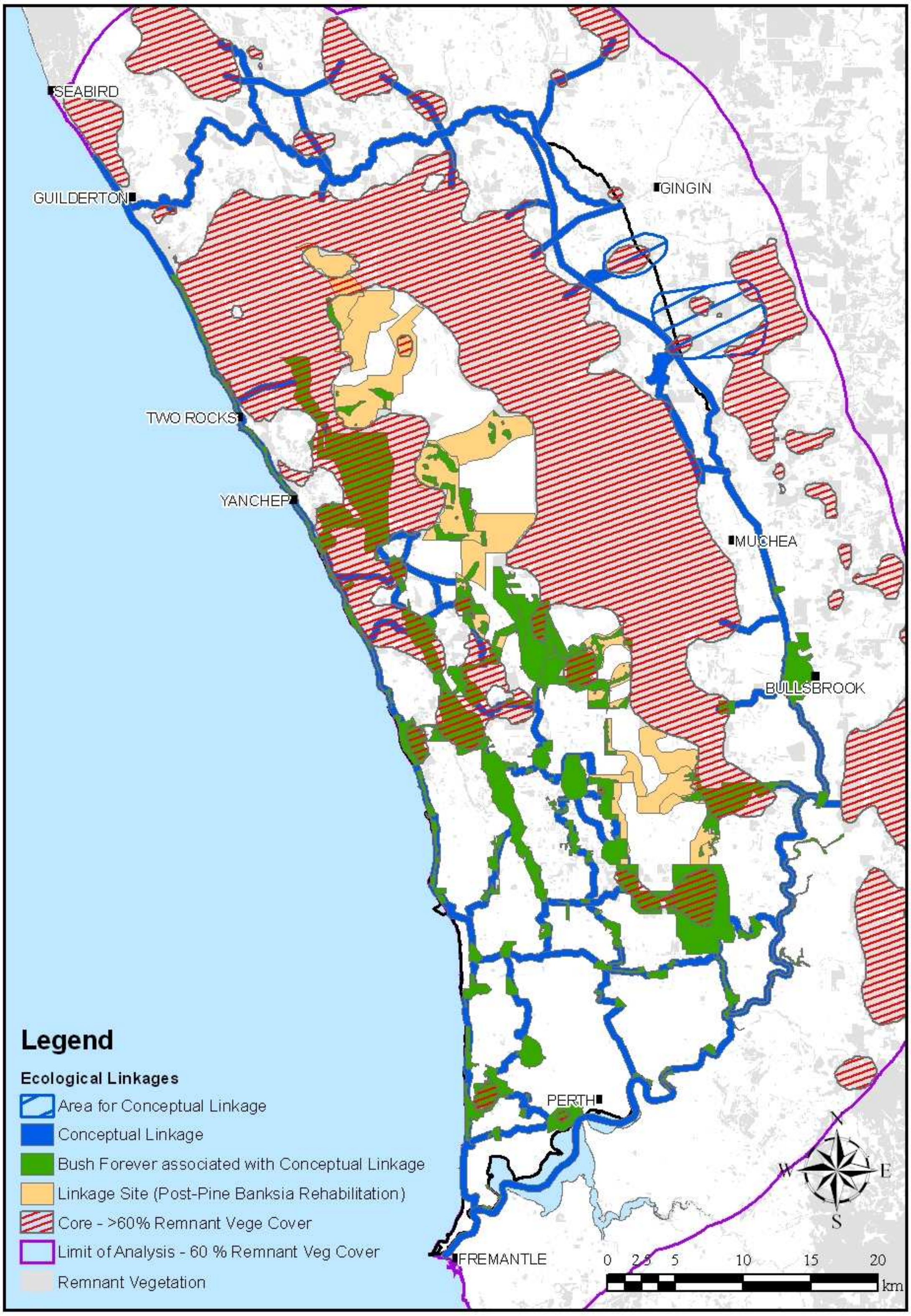


Figure 6.6: Ecological linkages for the GSS study area incorporating Bush Forever sites and highlighting > 60% remnant vegetation 'core' areas.

# Project 3. Connectivity and corridor analysis

Technical report reference: Eco Logical Australia Pty Ltd (2009). *Methodology and recommendation for connectivity/corridor analysis for the GSS study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

In this project, a connectivity analysis identified potential biodiversity corridors by highlighting areas in the landscape that could act as fauna corridors. This was undertaken using the Spatial Links tool and ESRI ArcMap GIS, which maps the GSS study area as a grid and creates pathways across the landscape according to input datasets and a set of rules. The datasets used were vegetation type, patch size and land use. Identification of habitat areas was also required, with the aim of connecting these areas. In this analysis, the habitat grid was based on vegetation within 500 m of a recorded location of a threatened species, and having native vegetation patches > 4 ha in size within the land-use score 2 or 3 (e.g. rural, parks and recreation and reserves). Additionally, the maximum corridor length was set at 50 km, based on a review of the input data and the nature of the study area.

The resulting connectivity model (Figure 6.7) represents a least-cost path where the minimum possible barrier to movement is identified based on the pre-determined cost grid. These potential corridors should be reviewed, to identify those that are feasible and best meet the aims and objectives of linkages. This analysis should be incorporated with the work completed by the DEC–GSS team (Project 2), including the ranking of biodiversity assets (Chapter 11) and information on the distribution of key fauna, to build an appropriate corridor network.

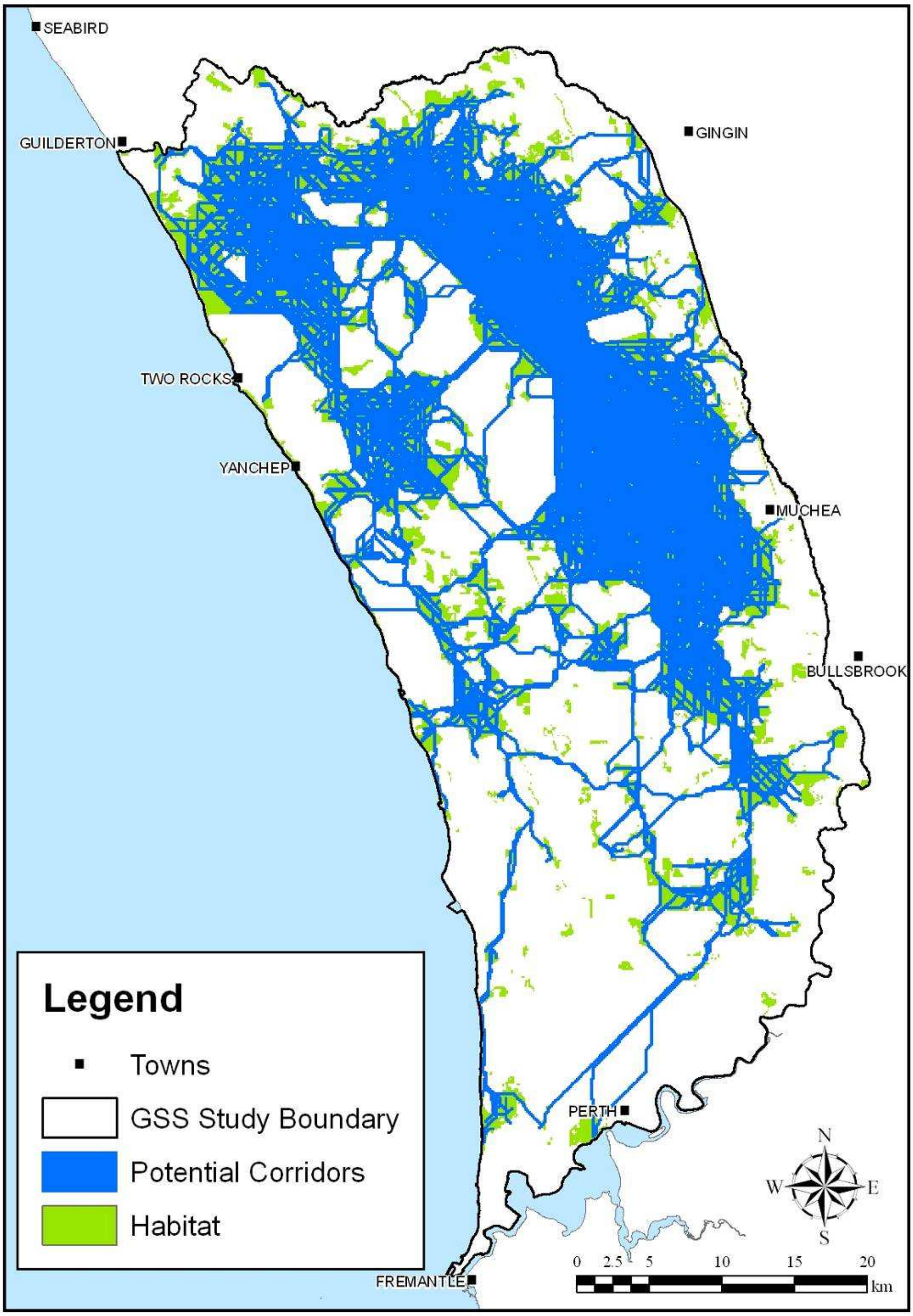


Figure 6.7: Potential biodiversity corridors identified by the Spatial Links connectivity analysis – corridor width is 100 m.

# Discussion

Habitat loss and fragmentation have been identified as major threatening processes for biodiversity and ecosystems. The aims of this chapter were to review information on the impacts and consequences of habitat loss and fragmentation on biodiversity, conservation and management in fragmented landscapes and the role of ecological linkages. The chapter also summarised the outcomes of projects undertaken to assess levels of retention and representation of remnant vegetation and to design ecological linkages.

The review confirmed that habitat loss and fragmentation is widely distributed on the Gnangara groundwater system and that clearing of remnant vegetation for urban development and agriculture has been extensive, especially in the south-west and central areas. The impacts on biodiversity and communities however, are currently poorly understood and information has been limited to impacts on susceptible plant species, threatened ecological communities, and impacts on fauna in urban fragments.

Analyses of the occurrence and distribution of the fragments and remnant vegetation established that there is approximately 101 000 ha of remnant native vegetation and that the GSS study area retains some of the largest intact areas of native vegetation on the Swan Coastal Plain. Whilst 47% of the GSS study area is remnant vegetation, only 37% of the SWA2 IBRA region is, ~20% of which is located within the GSS study area. Although nearly 50% of the GSS study area retains native vegetation, the degree of retention varies considerably, with only three of the 21 vegetation complexes within the GSS study area having > 30% of their remnant vegetation protected.

On the Swan Coastal Plain, the majority of studies on the impacts of fragmentation and habitat loss on fauna have been carried out in urban fragments where mammals have been found to have largely disappeared, all reptile groups, except skinks occur in greater numbers on larger remnants, and large reptiles such as goannas and snakes have been detrimentally affected (Cooper 1995; How and Dell 1994; 2000). Although the studies provide some evidence that herpetofauna are resilient to fragmentation the long-term persistence of the populations is not assured, particularly if barriers prevent recolonisation of patches.

The recent fauna studies undertaken for the GSS (2007–08) in the larger intact remnants in the north of the study area recorded 9 native mammal species. Historically, up to 33 mammal species have been recorded on the northern Swan Coastal Plain, however, only 11 – 12 species are considered to be currently extant. The capture rate of mammals was exceedingly low. The most abundant native species was the honey possum (*Tarsipes rostratus*) and priority listed species such as the rakali (*Hydromys chrysogaster*) and the quenda (*Isoodon obesulus*) occurred in low abundance and at very few sites (Valentine *et al.* 2009). Due to their low abundance and distribution maintenance of connectivity is likely to be significant for these mammal species. A rich and diverse reptile fauna was recorded. Although there was no evidence of the impacts of fragmentation on reptiles, this was not specifically addressed.

Many species of birds in the GSS are highly dispersive, especially nectar-dependent honeyeaters that follow resources (Davis 2009). Maintaining connection of *Banksia* woodlands in the GSS study area with northern sandplains is likely to be crucial for continued bird movement. Numerous wetland bird species have declined due to the loss of coastal plain wetlands (Chapter 4). Protection of wetland areas is crucial for these species.

The GSS make several recommendations to strengthen ecological resilience on the groundwater system including protection of remnant vegetation from clearing, fragmentation and other threats, and the development of regional ecological linkages to complement remnant vegetation and conservation reserves. This approach to mitigate against the adverse effects of landscape changes is intended to benefit many species and processes simultaneously. In the absence of detailed knowledge, such approaches are presumed to provide many benefits, however the effectiveness is not guaranteed.

In the GSS study area there is little information about the locality, distribution or movement patterns of fauna, or the likelihood that fauna will utilise the proposed ecological linkages. However, the results of the DEC-GSS projects provide direction for the future development of ecological linkages. Priority-listed fauna, such as the rakali and quenda, and the most abundant native mammal, honey possum, are taxa that are likely to benefit from landscape connectivity. In addition, highly dispersive bird species and the endangered Carnaby's black-cockatoo may benefit from ecological linkages. Designing linkages that target specific taxa is an appropriate first step. For example, the creation of 'corridors for Carnaby's' that focus on providing food and roosting resource is likely to be a very significant requirement due to the impacts of pine removal on these resources.

Although the development of linkages is recommended, further work is required to assess the potential use of linkages by particular species and the resources that they will require. In addition, the GSS study area has numerous other threatening processes (see Chapters 7 to 10), and their impacts on linkages need to be determined. For example the mammal taxa of the GSS are threatened by introduced species (see Chapter 8), and the creation of ecological linkages, without appropriate predator control, are unlikely to provide tangible benefits to them.

The GSS recommends that 9 000 ha of ecological linkages be established across the pine plantation area (post-harvesting) and a large portion will require revegetation and rehabilitation. The cost of revegetation is likely to be significant and cost-benefit analyses should be undertaken to build an appropriate effective corridor network with priority areas, and targeting specific taxa.

## Recommendations

In order to manage the effects of habitat loss and fragmentation it is recommended that:

- the protection status of remnant vegetation identified by Bush Forever (~ 8000 ha), occurring outside existing protected areas, be finalised
- the regional significance of remnants in the portion of the GSS covered by the Shires of Gingin and Chittering be assessed
- additional protection for remnant vegetation of Beermullah, Guildford, Swan, Yanga and Coonambidgee complexes be provided in order to maintain ecological linkages to the existing conservation estate to the west and east

- opportunities to rehabilitate Beermullah, Guilford and Swan complexes along Ellen Brook and the Swan River be investigated
- the proposed ecological linkage framework be used to develop guidelines for establishing and prioritising ecological linkages, based on recent biodiversity data and investigations of costs and benefits
- the proposed ecological linkages be assessed for requirements of species such as the honey possum (*Tarsipes rostratus*), Carnaby's black-cockatoo (*Calyptorhynchus latirostris*), the rakali (*Hydromys chrysogaster*) and the quenda (*Isoodon obesulus*)
- further analyses of connectivity corridors be undertaken based on recent data on fauna distribution and habitats obtained from studies carried out for the GSS and incorporating the ranking of biodiversity assets (Chapter 11)
- areas for development of corridors within the GSS study area be prioritised by incorporating cost-benefit analysis.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter Seven: Disturbance and Threats to Biodiversity Values – Fire Regimes

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Department of Environment and Conservation

Department of Environment and Conservation

Our environment, our future

September 2009

Government of Western Australia
Department of Water
Department of Agriculture and Food WA
Department for Planning and Infrastructure
Department of Environment and Conservation
Water Corporation
CSIRO
Forest Products Commission Western Australia

# 7. FIRE REGIMES

## Key points

- Fire has been an integral part of the Australian environment for millions of years and although many wildlife have traits that enable them to survive fire; often, they are adapted to specific fire regimes, determined by intensity, frequency, season and scale.
- Inappropriate fire regimes may have undesirable consequences including declines or local extinctions of biodiversity.
- Although the impact of inappropriate fire regimes has been identified as a major threat to biodiversity conservation on the Swan Coastal Plain (DEC 2007, 2009 SNCA Plan) information on the impacts of fire on biodiversity are limited.
- Evidence for post-fire seral responses of reptiles and mammals in the Gnangara groundwater system was obtained from fauna studies thus providing strong support for maintenance of a diverse range of post-fire aged habitat including long-unburnt habitat.
- Burning regimes need to ensure different fire ages in the long term, including retention of long-unburnt *Banksia* and *Melaleuca* communities that are important to fauna species such as honey possum, quenda, rakali, *Neelaps calonotus*, and *Menetia greyii.*
- Several projects examining fire management and history in the GSS study area have been initiated. This work on fire of the GSS study area will continue only until June 2010, and preliminary results are discussed in this chapter.
- Reviews of DEC fire management operations have been initiated (2009-10). They include: the planning framework; the impact of weather conditions, the specific requirements for maintaining wetlands, threatened species, reference areas, and constraints following the replacement of pine plantations with strategic ecological linkages and parkland.
- Assessments of fire frequency, area and distribution based on DEC fire records and analyses of Landsat imagery are currently underway (2009-10).
- Development of ecological burning regimes based on plant vital attributes, frequency distributions of different vegetation complexes and habitat requirements of fauna in the Gnangara groundwater system are in progress (2009-10).

- Approaches for fire management need to incorporate diverse requirements, including: (i) maintaining diverse fire regimes based on vital attributes and life histories of focal species; (ii) applying diverse fire regimes for habitat diversity; and (iii) regimes that aim specifically to manage fuel accumulation.

# Introduction

Fire occurs as a natural disturbance via lightning in many ecosystems and plays a key role in modifying landscapes and promoting ecosystem changes (Whelan 1995). Fire-induced ecosystem changes subsequently influence environmental and biological heterogeneity, vegetation floristics, and faunal assemblages (Bond and Van Wilgen 1996; Brawn *et al.* 2001; Whelan 1995). Around the globe, humans have a history of altering natural fire regimes by employing fire as a land management tool, or for recreational purposes (Kauffman *et al.* 1993; Russell-Smith *et al.* 2003). Although prescribed burning may be useful for restoring conservation values in some ecosystems (Angelstam 1998; Brawn 2006; Davis *et al.* 2000; Peterson and Reich 2001), inappropriate fire regimes may have undesirable consequences for native biodiversity (Barlow and Peres 2004).

The arrival of humans in Australia, some 45 000 years ago (O'Connell and Allen 2004), heralded a change in fire-disturbance patterns, although the extent of aboriginal-mediated disturbances, including the use of fire, on Australian landscapes is still a contentious issue (Bowman 1998; Flannery 1990; Johnson 2006; Kershaw *et al.* 2002; Singh *et al.* 1981). It is likely that human-mediated fire has been an important disturbance that has influenced the Australian environmental landscapes for at least the last 5000 years (Bowman 1998; Johnson 2006; Russell-Smith *et al.* 1997). European settlement in Australia dispossessed the Aboriginals of their land and consequently substantially modified traditional burning practices in large parts of Australia. Fire was originally viewed as a hazard and burning was suppressed, leading to the build up of high fuel levels that eventually resulted in devastating wild fires (Gill 1981c). Today, fire is employed as a land management tool for a variety of reasons including human safety, asset protection (e.g. houses and property), pastoral and agricultural management, weed control, conservation management and traditional aboriginal land management. However, there is often a gap between the knowledge of fire behaviour and fire ecology and its on-ground application and use in the management of fire (Burrows 2008).

# Gnangara sustainability strategy and fire

Several processes threaten biodiversity in the Gnangara groundwater system, including climate change, decreasing rainfall, habitat destruction through clearing of native vegetation, fragmentation, disease (e.g. *Phytophthora cinnamomi*), and inappropriate fire regimes (DEC 2009; Government of Western Australia 2009b; Mitchell *et al.* 2003).

Maintaining biodiversity is fundamental to maintaining ecosystem processes and is an environmental priority of both Commonwealth and State Governments in Australia. However, current understanding of biodiversity values in the GSS study area, ecosystem processes and the dynamics of the Gnangara groundwater system, particularly at landscapes scales, is inadequate (Government of Western Australia 2009b). Without improved knowledge, justification of changed management actions in the face of potentially degrading impacts on biodiversity is difficult.

One of the challenges involved in developing a land and water use management plan for the GSS study area is the strong interconnectedness between land uses and hydrological balance, which in turn affects consumptive water yields and the ecological integrity of water-dependent ecosystems and other terrestrial ecosystems. Modification of fire regimes on Crown land has been proposed as a cost effective option to enhance water yield to the Gnangara Mound (Canci 2005; Yesertener 2007). In order for this to become a management option, the biodiversity consequences must be understood and the water yield and biodiversity balance quantified. The GSS seeks to address these gaps by improving our knowledge of the impacts of fire on biodiversity values on the Gnangara groundwater system.

The aims of this chapter are to:

- review information on the role of fire in ecological communities and the response of biodiversity to fire in Australia
- review information on the impacts of fire on biodiversity on the Gnangara groundwater system
- examine the current management policies regarding fire in protected estate
- review fire management policies applied to the DEC - managed estate in the GSS study area (Swan Region).

In addition, a number of DEC GSS projects were undertaken to address identified knowledge gaps for the GSS study area. These included:

1) **Effects of time since fire on ground-dwelling vertebrates**
   In this study we surveyed the terrestrial fauna of the northern GSS study area, concentrating on the largest contiguous areas of remnant bush land, and the main landforms and differences in fire history.

2) **Fire and the B*anksia* woodlands of the Swan coastal plain – fuel reduction burns and water recharge on the Gnangara groundwater system**
   Alterations to the fire regime to enhance water yield is likely to have consequences for biodiversity on the Gnangara groundwater system. This on-going project seeks to improve our knowledge and measurement of the impacts of fuel reduction burns on biodiversity values and groundwater recharge on the Gnangara Mound. This report provides a summary of DEC biodiversity work completed to date (July 2009).

The major findings of these projects have been summarised in this document. The full technical reports are provided as a list in Appendix 2.

# Fire regimes

Australia has been the most fire-prone continent on Earth for a long time (Bowman 2003), yet fire and the Australian environment remains a complex issue. On the one hand, fire has been a part of the Australian environment for millions of years, and consequently has influenced the structure, function and sustainability of fauna and flora. Adaptations of flora and fauna to tolerate and survive in an environment where fire is prominent and frequent have developed over millions of years (Bowman 2003); indeed some biota have adaptations that lead them to rely on fire (e.g. the seeds of some species of *Hakea* are released by heat during bushfire (Attiwill and Wilson 2003). On the other hand, the arrival of humans in Australia has altered this balance between the environment and fire.

Before humans arrived, Bowman (2003) suggests that fires were started by lightning and burned in large areas infrequently and created a broad-based mosaic in eucalypt savannas.

The arrival of Aboriginal people presumably led to changes in fire patterns, using ‘firestick farming’ (Abbott 2003; Bowman 2003), whereby fires were lit frequently and burnt small areas. This regular and deliberate burning of parts of the landscape probably maintained a fine-scale vegetation mosaic of varying time since last fire across the landscape (Bowman 2003). For example, this regime was thought to be used by the Noongar Aboriginal people in the south-west of Western Australia (Bowman 2003).

Every fire requires three components: fuel, a source of ignition, and oxygen. However, fire behaviour and its impact depend on factors such as weather conditions, topography, fuel loads and distribution, as well as suppression activities (Whelan *et al.* 2006). Fire intensity (rate of heat energy release) is used to measure the severity of a fire (Burrows *et al.* 2008). The intensity and rate of spread of the fire depends on the quantity of fuel, the moisture content of the fuel and the weather conditions (Attiwill and Wilson 2006).

The long-term effect of fire on a landscape varies according to sequences of fire events, rather than to a single fire event. Sequences of fires are known as fire regimes, and are determined by four factors: intensity (how severe fires are), frequency (how often fires occur), season (the time of year fires occur) and scale (the extent and patchiness of a fire). It is important to understand the fire regime in order to define risks to people and property, and to make management decisions (Bradstock *et al.* 2002). In terms of biodiversity, inappropriate fire regimes (e.g. long periods of fire exclusion, sustained frequent burning, large and intense wildfires and post-fire grazing (Burrows and Wardell-Johnson 2003) may lead to local extinctions of plants and animals (e.g. (Woinarski 1999), and may result in a loss of biodiversity and structural complexity over time (Burrows and Wardell-Johnson 2003). However, the term ‘inappropriate’ is relative – what may be an inappropriate fire regime for one species may be beneficial to another species (Whelan *et al.* 2006).

Fire management involves the prevention of fire, the suppression of existing fires, and the introduction of fire where appropriate. Fuel quantity and weather conditions are the most influential factors of fire intensity, but fuel quantity is the only factor that can be effectively controlled in fire management (Bowman 2003; Burrows *et al.* 2008). Therefore, fire regimes should be planned to reduce fuel loads (Attiwill and Wilson 2006) through the use of prescribed burning. Prescribed burning refers to the planned use of fire to achieve specific land management objectives, where fire is applied under specific environmental conditions to a predetermined area.

# Climate change and fire regimes

Wildfire is a global issue, and the key factors involved – climate/weather, fuels, ignition agents and people – will continue to change as they respond to global changes in climate (Flannigan *et al.* 2009). Primarily through the release of greenhouse gases, the global climate is changing rapidly, which may have significant and possibly unexpected impacts on global fire activity (Flannigan *et al.* 2009).

The implications for an interaction between the changing climate and fire, in terms of fire frequency and severity, are vast. Generally, climatic changes, in terms of increased temperatures, declining rainfall and longer drought periods, are expected to interact with fire primarily though an increase in fire-weather risk, with the number of very high or extreme fire danger days projected to increase significantly in the next 100 years (Hennessy *et al.* 2007). Predictions are that climate change will foster a general increase in area burned and fire occurrence (however there will be some areas with no change and some areas with decreases) and longer fire seasons for temperate and boreal regions. Changes in fire intensity and severity in the future are difficult to predict and this area needs further research. Overall, it is expected that global fire activity will continue to increase as a consequence of climate change (Flannigan *et al.* 2009).

Climate change may lead to complications of future fire management and prescribed burning, with such potential changes as reduced rainfall and higher temperatures, which in turn could affect the amount of leaf litter and soil moisture. A report by CSIRO found that if the average summer temperature increases in south-eastern Australia, the frequency of very high and extreme fire danger days will increase by 4-25% by 2020 and by 15-70% by 2050 (if model projections are correct) (Hennessy *et al.* 2007). Changes such as these are likely to be greatest in inland areas. For example, savannas are the most fire-prone biome on Earth, and many flora and fauna species in these ecosystems will be vulnerable to extensive and frequent fires, especially fauna that have small home ranges and are relatively immobile and longer-lived obligate seeder flora species (Yates *et al.* 2008). Ideal conditions/seasons for prescribed burning may also become restricted due to weather conditions that pose higher wildfire risk in spring and autumn (Hennessy *et al.* 2007).

More frequent, high intensity, large scale fires, as a result of climate change, will have implications for the biodiversity of the GSS study area. Firstly, the *Banksia* woodland of the GSS study area are generally adapted to fire (Enright *et al.* 1998a). In one GSS study area species, *Banksia prionotes,* adults are killed by fire, but fire stimulates seeds to germinate. This type of life history strategy is thought to be particularly vulnerable to frequent, widespread fire events, as seed regeneration may be insufficient to replace adults lost in the fire if the canopy seed bank has not had sufficient time to recover from previous fire (Wooller *et al.* 2002). Wooller *et al.* (2002) go on to suggest this will be particularly true when fires are widespread, since *Banksias* have limited dispersal potential. In the GSS study area, there are several species that are not killed by fire, but instead resprout from the original plant, such as *B. attenuata*, *B. grandis*, *B. ilicifolia*, *B. littoralis*, *B. menziesii* (Enright *et al.* 1998a). In these species adult trees can sometimes survive low to medium intensity fire due to their thick bark, and regenerate from lignotubers, which resprout following fire. (Note: a full description of classification of plants into spouter and seeder species is discussed in the next section.) Enright *et al.* (1998a) suggest that too frequent fires can still result in the local extinction of these species that resprout, but at a much slower rate than species where adults are killed by fire. If climate change conditions increase the frequency of fires, regenerating species, such as *B. prionotes*, are likely to be most at risk of decline. However, if fires become extremely frequent and/or the intensity of fires occurring is severe, other species, such as resprouters, will be unlikely to recover.

A number of studies also indicate that frequent, widespread, and/or severe fires will impact on priority fauna in the GSS study area. For example, following a major summer fire in 1985, which was followed by a series of other minor fires, the population size of a number of bird species, including the splendid fairy wren (*Malurus splendens*), western thornbill (*Acanthiza inornata*) and scarlet robin (*Petroica multicolor*) declined (Brooker 1998). Population declines were observed for eight years after fire and resulted in temporary cessation of breeding in western thornbills (*Acanthiza inornata*) and increased nest predation and parasitism in splendid fairy wrens (*Malurus splendens*). Capture rates of the honey possum (*Tarsipes rostratus*) also decline markedly after fire, typically remaining low for more than five years post fire, with maximum abundances recorded 20-30 years post-fire (Everaardt 2003).

# Responses of flora to fires

Fire has a direct effect on plants by affecting their growth, survival and reproduction (Burrows and Wardell-Johnson 2003). Fire itself affects the structure of vegetation by consuming live and dead vegetation (Bond and Van Wilgen 1996), and the frequency of fire affects both the structure of vegetation and its floristic composition (Burrows and Wardell-Johnson 2003; Muir 1987). Changes in vegetation structure and composition can affect light penetration, soil moisture, and soil nutrient levels. Plants themselves have many adaptive vegetative and reproductive traits that enable them to persist in fire-prone environments (Gill 1981a).

Whelan (1995) states that the survival of plants during a fire has two components: (i) a plant surviving the direct effects of the fire during the passage of flames; and (ii) a plant tolerating the changes to the environment after the fire. Many plants may tolerate the actual fire but cannot tolerate the stresses of the post-fire environment. Plant tissue dies during a fire due to the high temperature that the plant cells are exposed to, which can cause protein denaturation, lipid mobility, chemical decomposition or metabolic changes at continuously high temperatures (Whelan 1995). Less severe fires may only cause a temporary disruption to some biochemical pathways, and can depend on the length of time of heat exposure and the state of the cells (e.g. hydrated or dehydrated) (Whelan 1995). In addition, some parts of the plant are more important than others in terms of survival post-fire. The survival of woody plants after fire can vary according to the level of protection of the bud by soil (e.g. subterranean buds, lignotubers) or bark (stem buds located beneath the bark) during a fire. In some plant species, reproduction may be enhanced as a result of fire through a flowering response (e.g. *Xanthorrhoea australis*), or through seed that is held on the plant being released, or through germination that is stimulated by fire (Gill 1981a).

Plant species are classified at the most basic level into two classes: sprouters and seeders (Table 7.1, Gill 1981b; Whelan 1995). Seeders (or seed regenerated, non-sprouters or obligate seeders) are species in which mature plants are killed by fire and depend on seed for regeneration or germinate in woody capsules on the plant, called bradysporous or serotinous species. This latter group is more susceptible to population decline through inappropriate fire regimes. If a second fire kills a population of regenerating bradysporous plants before it reaches reproductive maturity, then it may decline and become locally extinct. Sprouters (or vegetatively regenerated) are species in which mature plants survive fire by re-sprouting (Gill 1981b). However, this classification does not take into account that different types of fires with different intensities, frequencies and seasons of occurrence are likely to have different effects on plants (Gill 1981b).

Table 7.1: A classification of plant species in relation to their response to fire (adapted from Burrows and Wardell-Johnson 2003; Gill 1981a; Gill 1981b).

| **Response Class** |
|---|
| **Ephemerals -** life span 1-3 years post-fire, regenerate from seed. |
| **Seeders -** reproductively mature plants that die following stem girdling or 100% leaf scorch.<br>a. Seed stored on plant (serotinous)<br>b. Seed stored in soil<br>c. No seed stored in burnt area (depends on dispersal) |
| **Sprouters -** reproductively mature plants that survive stem girdling or 100% leaf scorch.<br>a. Regenerative buds subterranean and present as:<br>i. Root suckers or horizontal rhizomes<br>ii. Basal stem sprouts or vertical rhizomes<br>b. Regenerative buds aerial and present as:<br>i. Epicormic buds grow out<br>ii. Undamaged active pre-fire buds (continued outgrowth of active aerial pre-fire buds) |

For seeder species, the only source of regeneration is the seed, and therefore this group is classified further based on the different modes of regeneration from seed (Gill 1981a):

(1) Species that accumulate a store of seed on the plant between fires and are released at the time of the fire. These species do not usually have seeds stored in the soil (e.g. *Banksia ornata*)
(2) Plants that store seed in the soil as dormant seed (e.g. *Acacia genistifolia*)
(3) Species in which there is no seed store on the plant or in the soil – these plants rely on the dispersal of seed into the burnt area from an unburnt area (e.g. *Atriplex vesicaria*).

For sprouter species, as most of the heat from fire rises, protection of buds below ground is a very effective survival mechanism from above-ground fires (i.e. not peat fires). Even though trees may endure 100% leaf scorch during fire, these trees survive because their buds are protected by the bark. For the species that have subterranean buds, species can either produce basal buds that grow out to form shoots (i.e. still one individual plant) or possess root suckers that produce multiple stems after fire (Gill 1981a).

Two other categories include geophytes and fire ephemerals (Shedley 2007; Whelan 1995). Geophytes are a group of species that avoid the main impact of fire in time or space as they have bulbs, corms, tubers or rhizomes, and their above-ground growth takes place outside the normal season for fires (summer-autumn) (Bell *et al.* 1984). Fire ephemerals are also short-lived species that germinate in large numbers following fire (and utilise the nutrient-rich post-fire site) and often avoid fire by completing their life cycles within one year and before the next fire event. Fire ephemerals also produce seed that is stored in the soil, which germinates in response to heavy rainfall or disturbance (Bell *et al.* 1984; Shedley 2007).

### *Fire regimes and post-fire vegetation dynamics*

The ability to flower and produce viable seed in inter-fire periods is fundamental to the persistence of vascular plants in fire-prone environments, especially the species that depend on seed stored on the plant (Burrows *et al.* 2008). Therefore it is vital to choose an appropriate fire regime to ensure the persistence of all species. The frequency of fires is important, and while single fire events rarely have an effect on species composition, repeatedly exposing the ecosystem to the same fire regime may be detrimental. For example, if the juveniles of species that store their seed on the plant (serotinous species) are exposed to fire before their first flowering, this species may be lost locally. On the other hand, species with short-lived seed or serotinous species that only regenerate after fire may decline in long unburnt areas (Bond and Van Wilgen 1996; Burrows and Wardell-Johnson 2003). The intensity of fires is also an important factor, with low intensity burns unable to germinate some species and high intensity burns damaging epicormic buds and viable seeds (Burrows *et al.* 1990; Yates *et al.* 1994).

Changes occur in floristic composition and structure of the vegetation with time since last fire. One way to describe the distribution of vegetation age classes post-fire (or post-fire seral stages) is to characterise them according to their functional habitat traits, such as floristic composition, live and dead vegetation structure and surface litter cover, all of which are a function of time since last fire (Burrows 2008).

Post-fire vegetation dynamics tend to have similar patterns (Ashton 1981; Bell and Koch 1980; Burrows 1994; Burrows and Wardell-Johnson 2003; Gill *et al.* 1999; Gould *et al.* 2007; Hobbs and Atkins 1990; Hobbs *et al.* 1984; McFarland 1988; Noble and Slatyer 1980; Russell and Parsons 1978; Specht 1981; Specht *et al.* 1958). Firstly, plant species richness is greatest in the first few years following fire before stabilising or decreasing. The cover and height of understorey vegetation increases rapidly post-fire before stabilising for a period of time and then declining. Total biomass also increases rapidly post-fire before stabilising and ultimately declining to a steady state. The proportion of dead vegetation increases with time since fire and then stabilises.

Burrows (2008) described an approach that characterises 'functional habitat' types within each major spatial unit (i.e. landscape unit, habitat type or vegetation complex) based on time since last fire and the changes in floristics and structure of vegetation that occur post-fire. Three broad functional habitat types (or post-fire seral stages) that are based on the rate of change of the understorey vegetation and floristics in forests of south-west Western Australia are described in Burrows (2008):

(1) Early seral stage – characterised by high plant species richness, vigorous regeneration, low but increasing vegetation biomass, cover and height (Burrows 1994; Burrows and Wardell-Johnson 2003);
(2) Intermediate seral stage – characterised by reduced plant species richness, slower increase or peak in the change in vegetation biomass, cover and height; and
(3) Late seral stage – characterised by lower plant species richness; decline in vegetation cover and height (as understorey plants thin out and senesce), high depth and quantity of litter, and stabilising or declining of above ground biomass (Burrows 1994; Gould *et al.* 2007).

The rate of change of vegetation structure and floristics is affected by the type of fire (i.e. intensity of fire). The three seral stages are not distinct stages; rather, vegetation dynamics change continuously after fire (Burrows 2008). Using indicators of the post-fire rate of change in floristic composition and structure for a given ecological unit can help to decipher the transition between the three seral stages (Burrows 2008). An example is the juvenile period of the slowest maturing fire sensitive plant species within the major vegetation type (Tolhurst and Friend 2001).

## Responses of fauna to fire

The interaction between Australian fauna and fire has received considerable attention (Catling and Newsome 1981; Fox 1996; Friend 1993; Whelan 1995; Wilson 1996) although the focus has been the impact of fire upon birds and mammals. Behavioural patterns and requirements for shelter and food are two factors that affect the responses of taxa to fire (Friend 1993). For example, species that nest in tree hollows may avoid the acute effects of fire. Conversely there may be limited food resources for sedentary species in the early post-fire period, whilst mobile species can migrate to unburnt patches to obtain food and shelter.

Similar to changes in the vegetation composition and structure over time following a fire, the composition of fauna (birds, mammals and invertebrates) that use post-fire habitat can also change (Burrows 2008). A ‘habitat accommodation’ model developed to describe post-fire succession of small mammals describes how succession occurs in response to vegetation changes (Fox 1982; Fox 1996). Species enter the succession as their specific requirements are met, and decline in abundance as conditions become suboptimal. This model is generally supported by the results of studies in southern heathlands, healthy woodlands and arid grasslands (Masters 1993; Newsome *et al.* 1975; Recher *et al.* 1974; Wilson 1996; Wilson *et al.* 2001). Similarly succession of reptiles has been documented for arid Spinifex landscapes, where there is a strong relationship between shelter and foraging requirements of species and their abundance in successional ages (Cogger 1969; Dickman *et al.* 1999a; Letnic *et al.* 2004; Masters 1996; Pianka 1996).

### *Invertebrates*

The impact of fire on invertebrates and the response patterns that invertebrates exhibit can be highly variable and difficult to detect, often more so than for vertebrates and plants (Campbell and Tinton 1981; Friend 1995; Whelan 1995; Whelan *et al.* 2002). This is due to several reasons. Most invertebrate studies lack robust experimental design, sufficient replication and adequate sampling of sites before and after fire (Friend 1995; Whelan 1995; Whelan *et al.* 2002). Similarly different sampling protocols (pitfall traps, sweep net sample, litter samples, soil cores) have been employed, and each of these protocols give different results (Friend 1995; Whelan 1995; Whelan *et al.* 2002). In addition, invertebrates are a diverse group and exhibit a wide range of life histories and morphologies and are found in many different habitat types. Patterns and fire-related responses may not be apparent when data is analysed at broad classifications, for example phylum, class, order (Friend 1995; Whelan *et al.* 2002). Likewise, invertebrates found in the soil and litter are sometimes grouped and analysed together, but these two groups include a wide range of species that may have very different patterns of fire response due to differing impacts that fire has on the litter and soil (Whelan 1995). Finally, fire history such as intensity, frequency and extent needs to be taken into account as its' impact upon invertebrates can alter accordingly(Whelan 1995; Whelan *et al.* 2002).

Fire directly impacts invertebrates by killing them, as well as indirectly by affecting their habitat. Some invertebrate species survive the direct effect of fire by either moving ahead of the fire front, by being protected in the soil or other refugia (e.g. termite mounds), or if the fire coincides with a dormant part of their life cycle (Whelan *et al.* 2002). Some species may also survive due to the patchiness of a fire, providing refugia in the unburnt pockets (Whelan *et al.* 2002). Most invertebrates subsequently recolonise burnt areas from unburnt patches (Whelan *et al.* 2002), dense crowns of plants (Gandar 1982; Main 1981; Whelan *et al.* 1980), thick layers of leaf litter (Andrew *et al.* 2000), thick bark on trees, and soil under rocks and in burrows (Main 1981; Warren *et al.* 1987). Species recolonise at different rates, depending on their dispersal ability. The patchiness, intensity, extent and season will all influence the recolonisation capacity of invertebrates (Whelan *et al.* 2002). The post-fire burnt environment will also influence recolonising invertebrate survival due to altered habitat structure, food availability, surface temperature and soil moisture (e.g. Blanche *et al.* 2001; Tap 1996).

Many invertebrate taxa appear to decline after fire and then recover quickly (Friend and Williams 1996; Whelan 1995), with little change in subsequent abundance (e.g. Abbott *et al.* 1985; Collette and Neumann 1995). Friend (1995) reviewed several studies that examined the response of invertebrates to fire. This review indicated that Araneae, Lepidoptera, Isopoda, Blattodea and Thysanura are sensitive to fire and exhibit consistent patterns in response to fire across a variety of habitats. Some studies have found that populations of soil and litter arthropods will not recover to pre-fire population numbers during a five year inter-fire period, but other researchers have stated that populations will recover quickly. Some species may depend on the amount of litter layer (e.g. wood crickets, Dolva 1993), which takes time to build up after a fire.

York (1996) found that frequent burning significantly decreased the abundance of ticks and mites, insect larvae, flies and beetles. These groups are associated with leaf litter, and York (1996) concluded that these results were probably due to the removal of leaf litter (and changes in moisture) due to frequent fire. The substantial increase in bugs, ants and spiders was probably due to a response to changes in habitat suitability and a reflection that these species were easier to catch in a simplified post-fire environment.

Most of the impacts of fire on invertebrate communities are relatively short-term (2-3 years) and there are greater impacts from high intensity wildfires than from lower-intensity prescribed burns, and greater impacts from spring prescribed burns than autumn ones (Friend and Williams 1993). In order to overcome some of the aforementioned inconsistencies, further studies are required to examine the effect of fire on the survival, growth, reproduction and population changes of invertebrates (Friend 1995; Whelan *et al.* 2002).

## *Reptiles and frogs*

There have been very few studies on the impacts of fire on frogs. Most frog species are probably indirectly affected by fire due to their utilisation of pools, ponds and subterranean shelter, sites essentially protected from the direct effects of fire (Friend 1993). Bamford (1986; 1992) investigated the responses and adaptations of three frog species to fire and their relationship to vegetation structure, litter density and potential food supply. He found no relationship between time since fire and number of species or total abundance of frogs. However, the abundance of two species (*Limnodynastes dorsalis* and *Myobatrachus gouldii*) was greater in long unburnt areas. These abundance changes were not related to litter and vegetation changes, or prey abundance. The differences were proposed to be the result of nett movement away from burnt areas after fire in these nomadic species. Frogs appear to be influenced more by proximity to water than by time since fire (Bamford 1986). Arnold *et al.* (1993) found little impact of fire on frogs immediately after the fire, but found lower densities on plots three years post fire. Investigations of fire impacts on

*Geocrinia* spp. in forests found 30% declines in populations two years after fire, but no differences between burnt and control plots seven years after fire (Conroy 2001; Driscoll and Roberts 1997).

Thus although many frogs can survive the event of fire, there may be declines some time after fire, followed by recovery. Fires *per se* may not be the most important factor, but rather the consequential changes in habitat and microclimate that impact on frog habitat, such as water levels, quality or open vegetation and patchiness of fire (Bamford and Roberts 2003).

The effect of fire on reptile and frog communities is still largely unknown in Australia (Bamford and Roberts 2003; Friend 1993). Reptile information is based on studies in mallee woodlands, heathlands and savanna forests where reptilian diversity is high (Caughley 1985; Cogger 1969; 1989; Dickman *et al.* 1999a; Letnic *et al.* 2004; Masters 1996; Pianka 1996; Trainor and Woinarski 1994; Valentine and Schwarzkopf 2009). Few studies have been undertaken in southern temperate areas (e.g. Humphries 1992; Lunney *et al.* 1991). Studies suggest that many species of snakes and lizards are resilient to the short-term effects of fire, due to their preference for open microhabitats and use of burrows, whereas arboreal or surface-dwelling species less protected (Fox 1978; Friend 1993). Although the longer-term relationships between reptiles and fire regimes are still uncertain, species respond in variable ways and the type of fire regime imposed may be critical in determining species response (Braithwaite 1987; Valentine and Schwarzkopf 2009).

Seral responses of reptiles have been investigated in a number of studies (Bamford 1986; Braithwaite 1987; Caughley 1985; Cheal *et al.* 1979; Cogger 1969). For example, the mallee dragon (*Ctenophorus fordi)* was recorded at higher density in 10-year, post-fire regrowth than in unburnt areas (Cogger 1969). In mallee areas, there was similar species richness in sites aged four to 60 years since burn, although relative abundance of species was markedly different (Caughley 1985). Similar changes occurred in lizard communities in broombrush communities (Woinarski 1989). In Spinifex hummock grassland reptile assemblages changed apparently following fire, because the Spinifex-dependent taxa did not survive where all Spinifex had been combusted (Cogger 1984). Similarly, lizard community composition at Spinifex (*Triodia*)-dominated sites aged from 0 to > 25 years post-fire could be arrayed along a single continuum of vegetation structure and conformed to a directional model for post-fire succession (Letnic *et al.* 2004). Changes in the abundance of early to late succession species are considered to reflect the differing thermoregulatory, shelter and dietary preferences of the species.

## *Birds*

Previous studies in a variety of habitats have observed the response of different bird assemblages in grasslands (Pons *et al.* 2003), tropical savannas (Mills 2004; Valentine *et al.* 2007; Woinarski 1990; Woinarski *et al.* 1999), oak savannas and forests (Artman *et al.* 2001; Brawn 2006), conifer and pine forests (Hutto 1995; Saab *et al.* 2005), and rainforests (Barlow *et al.* 2006). The response of bird species to fire is often related to changes in vegetation structure and the availability of resources in the post-fire environment (Brawn *et al.* 2001; Davis *et al.* 2000; Woinarski and Recher 1997). This post-fire environment is influenced by the fire regime, the intensity or patchiness of fire and the time since last fire, consequently altering nutrient availability, food resources and floristics (Saab and Powell 2005; Smucker *et al.* 2005; Whelan 1995; Woinarski 1990; 1999; Woinarski and Recher 1997).

Many of the impacts from fire on birds are indirect. Direct impacts are influenced by where in the vegetation a particular species occupies as well as the species' mobility. In dry sclerophyll forests of Western Australia, the higher in the forest layer that a bird species inhabits, the less effect fire will cause (Catling and Newsome 1981). Sedentary species in heathlands tend to have a close association with shrub canopies or ground vegetation, and therefore are the most vulnerable to being killed by fire. However, sedentary species that live in burrows or less fire-prone microhabitats (such as rock outcrops) and more mobile species are more likely to survive (Keith *et al.* 2002).

The high abundance of birds in recently burnt habitat is often observed in the first year post-fire (Hutto 1995; Smucker *et al.* 2005; Woinarski 1990; Woinarski *et al.* 1999), and may be related to an increase in food or greater accessibility to food resources (Brawn *et al.* 2001; Whelan 1995; Woinarski and Recher 1997). Bird abundance has been found to increase in jarrah and karri forests in south-west Western Australia after prescribed burning under both dry and moist soil conditions (spring in jarrah forests, early summer in karri forests). However after wildfire (in autumn), species richness and total bird abundance decreases (see studies in Abbott 1999)). Although burning may reduce the amount of vegetation, remaining plants often exhibit traits that enhance survival in the post-fire environment, including flushes of new growth via epicormic shoots, vegetative regrowth and re-sprouting, and flowering (Gill 1981a; Whelan 1995). New foliage on remaining vegetation may attract arthropods (Force 1981; Recher *et al.* 1985; Swengel 2001), temporarily increasing food resources for some insectivorous birds (Barlow and Peres 2004; Hutto 1995). A short-term increase in the abundance of some granivorous bird species has been attributed to the release of seeds following fire (Hutto 1995).

Vegetation recovery following fire is important for bird recolonisation, especially those species that inhabit the ground layer and shrub vegetation (Recher *et al.* 1987; Smith 1985). Recovering vegetation must be able to provide food, nest sites and protection in order to be of use (Whelan 1995). Birds with specialised habitat requirements, including prey and vegetation structure, may decline following burning if their preferred resources have been adversely affected by fire (Artman *et al.* 2001).

Fire can alter the nesting behaviour of certain birds of heath vegetation. In south-west Australia, in the immediate post-fire period, three bird species (splendid fairy wren *Malurus splendens*, western thornbill *Acanthiza inornata*, and yellow-rumped thornbill *A. chrysorrhoa*) build their nests in shrubs that regenerate by sprouting following fire (Brooker and Rowley 1991). Splendid fairy wrens (*Malurus splendens*) mainly build their nests in low shrubs, although some nests are located in small trees. The mean height of nests of western thornbills does not change following a fire, although the placement of their nests in a favoured plant, *Xanthorrhoea preissii*, increases with time since fire. On the other hand, nests of yellow-rumped thornbills (*A. chrysorrhoa*) in burnt areas tend to be found higher and in a more restricted range of shrubs than existed before a fire.

### *Mammals*

The effects of fire on mammalian fauna vary depending on the fire regime including frequency, intensity, season and spatial scale (Catling 1991; Friend 1993; Wilson 1996; Wilson and Friend 1999). While the impacts to some species and communities have been comprehensively studied, we have little knowledge for others. The responses of small mammals to wildfire and low intensity burns have been investigated in south-east

Australian heathlands, woodlands and forests; south-west Western Australian forests; central arid grasslands and tropical savannas (e.g. Andersen *et al.* 2005; Catling and Newsome 1981; Christensen and Kimber 1975; Dickman *et al.* 1999b; Fox 1982; 1983; Leonard 1972; Letnic *et al.* 2004; Masters 1996; Newsome *et al.* 1975; Recher *et al.* 1974; Wilson 1996; Woinarski *et al.* 2001). Other studies have employed the chronosequence approach to compare populations and communities in areas of different post-fire successional ages (Cockburn *et al.* 1981; Fox 1990; Fox and McKay 1981).

Changes in post-fire abundance of mammal species are strongly related to habitat requirements (Fox 1982; Fox 1996; Fox and McKay 1981; Newsome *et al.* 1975; Wilson 1991; Wilson *et al.* 1990). For example, *Pseudomys novaehollandiae* (New Holland mouse) were captured in low numbers initially after fire, and then in higher numbers 2-4 years post-fire, and then they declined. In contrast, *Antechinus swainsonii* (dusky antechinus) were not captured until 6-10 years post-fire (as they require complex, dense ground-cover) (Attiwill and Wilson 2003). Catling *et al.* (2001) developed a sequence of 'habitat complexity scores' that describe changes in vegetation structure over time since fire, and used the scores to examine the distribution of fauna across the landscape and the response of mammals to changes in vegetation structure and habitat complexity since fire.

Wildfires of high intensity frequently result in substantial reductions of populations, although as animals sometimes survive in small unburnt patches, they can recolonise (Catling and Newsome 1981; Newsome *et al.* 1975; Recher *et al.* 1974). Low intensity fires that leave substantial unburnt areas, may produce no immediate declines as survival is high (Leonard 1970). Individual species respond differently to fire and fire regimes due to their different shelter and food requirements, and their behavioural patterns (Friend 1993). Taxa also have different successional responses and preferences. Macropods such as the western grey kangaroo (*Macropus fuliginosus*) and the western brush wallaby (*Macropus irma*) favour burnt forest (Christensen and Kimber 1975). Species such as the eastern chestnut mouse (*Pseudomys gracilicaudatus*), which prefer open, floristically rich vegetation, recolonise early in the post-fire recovery period, while species such as the swamp antechinus (*Antechinus minimus*), which require dense ground cover, exhibit low population numbers up to 20 years after fire (Fox 1982; 1983; Wilson *et al.* 2001; Wilson *et al.* 1990).

There have been few studies on the impacts of repeated fire or sustained fire regimes on mammal species. Studies of the effects of consecutive wildfires on small mammals in south-eastern Australia found that responses can differ in extent or time, but patterns are similar and are strongly related to the structure of the vegetation (Catling 1986; Fox 1982; 1990; Fox 1996; Newsome *et al.* 1975; Recher *et al.* 1974). There is also information that cumulative impacts of fires including intensity and frequency can impact on vegetation dynamics involving composition and structure (Bradstock *et al.* 2002). Response of fauna to repeated fires will also be impacted as a result of such changes (Bradstock *et al.* 2005). Similarly, relatively little research has been directed at the impacts of fire at a landscape level. There is little information on factors such as the role of unburnt patches and refugia, the relationship between fire extent and recolonisation, and the process of recolonisation of burnt isolates (Bradstock *et al.* 2005). The small scale of available studies has implications for our understanding of the applicability of results at the landscape level (Andersen *et al.* 1998; Fox 1996; Gill *et al.* 1990; Whelan 1995). For individual species we need to understand the dynamics of the persistence of metapopulations, which are related to the survival and extinction rates of individual populations, the recolonisation rates for habitat patches and the dispersal rates of the species (Gilpin and Hanski 1991; Hanski 1991; Harrison 1991).

Overall, mammal communities are adapted and resilient to fire and species are adapted to specific fire regimes. There is an identifiable successional response to fire for small mammal communities that occur in fire-adapted communities, and this is predominantly in response to structural vegetation changes. Our understanding of the impacts of changed fire regimes and impacts at the landscape level are limited. Early studies of the impacts of fire on mammals in south-west Australia focused on jarrah and karri forests, and species such as the tammar wallaby (*Macropus eugenii*), quokka (*Setonix brachyurus*) and woylie (*Bettongia penicillata ogilbyi*) (Christensen and Abbott 1989; Christensen 1980; Christensen and Kimber 1975). However, there have been few studies of impacts of fire on mammals on the Swan Coastal Plain.

# Fire impacts on the biodiversity in the GSS study area of the Swan Coastal Plain

Studies of the impacts of fire on biodiversity in the GSS study area and on the Swan Coastal Plain have been limited, particularly in comparison to studies in Jarrah and Karri forests of south-west Western Australia (e.g. (Abbott 1999; Abbott *et al.* 1985; Adams *et al.* 2003; Burrows 2008; Burrows and Wardell-Johnson 2003; Burrows *et al.* 2008; Christensen and Kimber 1975; Kimber 1974; McCaw *et al.* 2003; Robinson and Bougher 2003; Van Heurck and Abbott 2003; Wooller and Calver 1988).

## *Vegetation*

Despite the extent of the *Banksia* woodlands and their proximity to Perth (and extensive clearing and disturbance in the area), there have been few studies investigating vegetation dynamics of *Banksia* woodlands on the Swan Coastal Plain and the impact of fire on vegetation in these communities, especially in comparison to studies in the forests of south-west Western Australia. Studies that have investigated various aspects of the impact of fire on vegetation on the Swan Coastal Plain include Baird (1977), Lamont and Downes (1979), Cowling and Lamont (1985), Hopkins and Griffin (1989a), Hobbs and Atkins (1990), and Lamont and Markey (1995). In addition, there are some studies on the impacts of fire on vegetation that have been conducted just outside the GSS study area (e.g. Hayward *et al.* 2008; Lamont *et al.* 2000).

In contrast, several studies have examined the impact of fire in heath and scrub-heath vegetation on the Geraldton Sandplain, 250-350 km north of Perth (e.g. Cowling and Lamont 1985; Cowling and Lamont 1987; Enright and Lamont 1989; Enright *et al.* 1996; Enright *et al.* 1998a; b; Groeneveld *et al.* 2002; Meney *et al.* 1994).

### Fire responses of flora on the Geraldton Sandplain

Several studies have examined fire responses of *Banksia* species and other species near Eneabba, 235 km north of Perth. Lamont *et al.* (2007) found that after two burns in ten years, the numbers of *B. attenuata* increased with each fire and *B. menziesii* decreased, due to different levels in seed production and fire tolerance. A post fire study of *B. attenuata*, *B. leptophylla*, *B, menziesii* and *B. prionotes* (Cowling and Lamont 1987) found that seed release in serotinous species of *Banksia* is largely fire-dependant; however *B. menziesii* and *B. prionotes* both exhibit regular spontaneous follicle rupture in summer. In a seed bank study (Meney *et al.* 1994), Restionaceae showed, regardless of fire response, a marked depletion of the seed bank after fire even though seed count in unburnt vegetation was relatively high. However, Epacridaceae indicated that seed banks persist in soil after fire regardless of fire response or life history. The exception to this was *Lysinema ciliatum*, which had low residual (no residual seeds one year after fire) seed density after fire. Between 90-100% of annual seed production of obligate seeder and resprouter Epacridaceae species deteriorate within two years. There was also no evidence of recruitment of any of the species studied after 10 years since last burn (Meney *et al.* 1994).

Serotiny (the canopy storage of seed for a prolonged period) is common in Australian sclerophyll vegetation (Cowling and Lamont 1985), and 76% of *Banksia* species are serotinous (George 1981). A correlation between a decrease in annual rainfall and an increase in average temperature with a decrease in plant height and an increase in the degree of serotiny was found in three *Banksia* species (*B. attenuata*, *B. menziesii* and *B. prionotes*) along a climatic gradient extending 500 km north of Perth (Cowling and Lamont 1985). This study concluded that the degree of serotiny in these three *Banksia* species is related to the fire characteristics of the site, which depend on plant height. In the northern-most site (Northampton), with a xeric scrub-heath, plant height was lowest and entire canopies of the *Banksia* species would be consumed by fire, promoting a massive release of seed. In the south-most site (King's Park), with a mesic woodland, cones would rarely come into contact with flames due to a greater plant height, and seeds are released spontaneously (Cowling and Lamont 1985).

## Fire responses of flora on the Swan Coastal Plain

### *Responses of individual species*

In a review by Hopkins and Griffin (1989a), the *Banksia* woodland on the Swan Coastal Plain was found to contain 13 long-lived perennial species that regenerated only from seed after 100% crown scorch. Six of these species were identified as fire sensitive and as having seed storage on the plant in bradyspores. Species that stored seed on plant include *Banksia prionotes*, *B. sessilis*, *Hakea trifurcata*, *Hakea obliqua*, *Beaufortia elegans*, and *Beaufortia squarrosa*. Species with seed storage in the soil include *Adenanthos cygnorum*, *Astroloma xerophyllum*, *Leucopogon striatus*, *Leucopogon cordatus*, *Lysinema ciliatum*, *Andersonia heterophylla*, and *Acacia pulchella* (Hopkins and Griffin 1989a). In *B. prionotes*, adults are killed by fire, but fire stimulates seeds to germinate. This fire response may be vulnerable to frequent, widespread fire events as seed regeneration may be insufficient to replace adults lost in the fire if the canopy seed bank has not had sufficient time to recover from previous fire (Wooller *et al.* 2002).

Several of the key species in the GSS study area are resprouters, including *Banksia attenuata*, *B. grandis*, *B. ilicifolia*, *B. littoralis*, and *B. menziesii* (Enright *et al.* 1998a). In these species adult trees can sometimes survive low to medium intensity fire due to their thick bark and also regenerate from lignotubers, which resprout following fire. Hobbs and Atkins (1990) suggest that both *B. attenuata* and *B. menziesii* do not depend on fire for recruitment in the *Banksia* woodlands on the Swan Coastal Plain. This concept is also supported by Cowling and Lamont (1985) who observed that neither species has serotinous cones, but rather they release their seed shortly after cone maturation.

Out of 1337 native GSS plant species, 42 have had their fire response recorded in studies that have been conducted on the Swan Coastal Plain. The 438 vascular flora species that are found in the GSS study area have been divided into their recorded fire response (note that there have been different fire responses recorded for the same species, therefore the total number of records in the table is 494) (**Error! Reference source not found.**). From these records, 37% of the native vascular plants are killed by 100% scorch with 53% surviving fire by utilising basal sprouts, epicormic growth, apical buds or soil suckers.

### *Responses in plant populations and communities*

There is little literature on the responses of plant populations and communities to fire on the Swan Coastal Plain. In the *Eucalyptus-Banksia-Casuarina* woodland of King's Park (Baird 1977), the first plants that grow after a mid-summer fire are *Xanthorrhoea* spp., followed closely by sedges (particularly *Tetrariopsis*). A few weeks after the fire, new leaves of the cycad *Macrozamia* appear which reach their full length in approximately four months. Some deep-rooted shrubs sprout within 2-3 weeks. With the start of the winter rains there is a flush of growth of herbaceous plants and annual weeds, as well as an increase in growth of shrub species and seedlings of trees and shrubs. Shrub species are erect and vigorous for the first 2-3 years after the fire, and the percentage of dead wood and litter from trees increases with time since fire. In stands not burnt for 20 years or more, Baird (1977) found a suppression of the undergrowth and a large amount of leaf and twig litter build-up.

Using a series of stands, within remnant areas of low woodland dominated by *B. attenuata* and *B. menziesii*, ranging in age since last fire from 1 to > 44 years, Hobbs and Atkins (1990) examined long-term vegetation development post-fire. Species richness increased for the first five years after fire, and many shrub species reached their greatest density two years after fire, thereafter declining in density. The shrub *Eremaea pauciflora* became more dominant with increasing age since last fire.

Flowering response post-fire can differ between species. For example, two species of grass trees showed a different flowering response post-fire; unlike *Xanthorrhoea preisii, Kingia australis* showed an immediate flowering response to fire, with a marked increased in the number of *K. australis* plants that flowered post-fire (Lamont and Downes 1979).

### Season of fire and flora responses

The season of a fire can have an effect on the rate and type of recovery of vegetation (e.g. growth, germination, flowering and fruiting) post-fire. For example, within remnant areas of low woodland dominated by *B. attenuata* and *B. menziesii* near Perth, autumn fires can promote seedling germination and regeneration (and may therefore be beneficial especially for seeder species), while spring fires may result in rapid vegetation recovery and greater species diversity (Hobbs and Atkins 1990). Autumn burns may result in less vegetation regrowth and may also increase invasion by non-native plant species (Hobbs and Atkins 1990). From this Hobbs and Atkins (1990) suggest that spring burning may be preferable in these remnant patches of *Banksia* woodlands.

Seasonal differences in the recovery of vegetation post-fire have also been recorded in the Jarrah woodlands in King's Park (Baird 1977). There was vigorous growth of *Xanthorrhoea* spp., fibrous monocotyledons and shrubs after a spring to early summer fire, with shoots of shrubs appearing within 3-6 weeks of the fire, and then growing more rapidly into the summer. While autumn burns are not necessarily unfavourable to the growth of shrubs, the growth of herbaceous plants was greater in autumn burns as compared to spring-early summer fires (Baird 1977).

North of the GSS study area, the responses of vegetation to fire in different seasons have also been studied. Cowling and Lamont (1987) examined the effects of autumn and spring burns on the recruitment of four *Banksia* species (*B. menziesii*, *B. prionotes, B. leptophylla* and *B. attenuata*) in a 15-year old stand that co-occur in the sandplain scrub heath 350 km north of Perth. The rate of seed release from burnt cones of all four species was significantly slower after the spring burn compared to the autumn burn. In addition, the number of seedlings recruited per parent of all four species was less than half as high after the spring burn than the autumn burn (after the first winter) (Cowling and Lamont 1987). Similarly, after a hot fire in late summer-autumn, seed release, germination and establishment was best for *Banksia burdettii* at Watheroo National Park (east of Jurien) (Lamont and Barker 1988).

### *Fire responses of threatened ecological communities and declared rare flora*

Several threatened ecological communities and declared rare flora occur in the GSS study area (see Chapter 2 for a detailed discussion). Interim Recovery Plans have been written for five of the ten declared rare flora that occur in the GSS study area. The Interim Recovery Plans, Evans *et al.* (2003) and Brown *et al.* (1998) were consulted to review known responses to fire of the ten species of declared rare flora (Table 7.3).

A number of adaptive management projects have been undertaken in the DEC Swan Coastal District that have examined the burn response of several threatened ecological communities and declared rare flora on the Swan Coastal Plain. These include: *Banksia mimica* (fire response at different fire intensities); *Caladenia huegelii* (examined the Fraser Road population after a wildlife occurred in the 2007-2008 fire season); *Melaleuca huegelii- Melaleuca systena shrublands on limestone ridges* (community type 26a described by Gibson *et al.* 1994) (fire response and percentage cover before and after a prescribed burn); *Macarthuria keigheryi* (fire response after a prescribed burn), *Perth to Gingin Ironstone Association (Northern Ironstones)* (examining this threatened ecological community after a major wildfire burnt the entire community), and *Muchea Nature*

*Reserve* (fire response of two species of declared rare flora (*Darwinia foetida* and *Grevillea curviloba*) were examined).

Table 7.2: Plant species found in the GSS study area with fire response observations from Swan Coastal Plain (SWA2) studies (Baird 1977; DEC 2008b; Hopkins and Griffin 1989a; Muir 1987), and summary of fire responses of 438 vascular flora in the GSS study area (Baird 1977; DEC 2008b; George 1981; Hopkins and Griffin 1989b; Lamont and Markey 1995; Meney *et al.* 1994 1994; Muir 1987). Some species were observed to have multiple fire responses, which are reflected in this table by a total of 494 records. Fire Response definitions have been based on Burrows *et al*. (2008).

| **Fire Response** | **# Taxa** | **Species Response Observed in the Swan Coastal Plain** |
|---|---|---|
| 100% scorch kills | 31 | *Calytrix fraseri; Conospermum triplinervium; Dodonaea hackettiana; Grevillea crithmifolia; Leucopogon racemulosus* |
| 100% scorch kills, in soil seed storage | 131 | *Acacia cyclops; Acacia saligna; Adenanthos cygnorum; Anthocercis littorea; Astroloma xerophyllum; Eremaea pauciflora; Gompholobium tomentosum; Homalosciadium homalocarpum; Lysinema ciliatum; Podotheca chrysantha; Templetonia retusa; Trachymene pilosa* |
| 100% scorch kills, no seed storage | 3 | |
| 100% scorch kills, on plant seed storage | 17 | *Acacia pulchella; Banksia leptophylla; Banksia prionotes; Banksia sessilis; Banksia telmatiaea; Beaufortia elegans; Hakea trifurcata* |
| Ferns and Allies (spores) | 1 | |
| Geophyte (Survives 100% scorch) | 50 | |
| Survives 100% scorch | 8 | *Hybanthus calycinus* |
| Survives 100% scorch, basal sprouts | 131 | *Calytrix leschenaultia; Eremaea beaufortioides; Hypocalymma robustum; Persoonia saccata; Regelia ciliate; Stirlingia latifolia* |
| Survives 100% scorch, epicormics | 16 | *Banksia attenuata; Banksia menziesii; Eucalyptus gomphocephala* |
| Survives 100% scorch, large apical bud | 3 | *Xanthorrhoea preissii* |
| Survives 100% scorch, soil suckers | 103 | *Acacia stenoptera; Amphipogon turbinatus; Grevillea vestita; Schoenus grandiflorus; Tetraria octandra; Thysanotus sparteus* |
| Total | 494 | |

Table 7.3: Responses to fire of declared rare flora on the Swan Coastal Plain.

| Scientific Name | Fire Responses | References |
|---|---|---|
| *Caladenia huegelii*<br><br>(Geophyte: Survives 100% scorch) | • Fire is considered detrimental if fire occurs between July to November (during vegetative and flowering stages).<br>• Fire may be beneficial as summer fires promote flowering.<br>• Field experiment showed that the Fraser road population is in a degraded bush block in Banjup surrounded by sand mines. No prescribed burning is allowed for this species. Wildfire occurred in 2007/2008 season. Recent experiment overlayed 5 x 5 m plots in burnt area. | • DEC (2008a)<br>• Evans *et al.* (2003)<br>• Brown *et al.* (1998) |
| *Darwinia foetida* | • Frequent fires reduce vigour and seed bank. | • Evans *et al.* (2003) |
| *Drakaea elastica*<br><br>(Geophyte: Survives 100% scorch) | • Fire is considered detrimental if fire occurs between April/July to November (during vegetative and flowering stages). Fire may kill plant during active growing period (late April-Oct). Indirect impacts of fire include loss of canopy cover and increased weeds.<br>• Fire may be beneficial if fire occurs between November to June, which may open up the canopy and reduce competition, but species still needs to retain some canopy vegetative cover after disturbance in order to protect plant and its fungus from desiccation. Fire is not likely to impact during the species' dormant period (November to early April).<br>• Field observation: species does not require fire to complete its life cycle. Increased competition with increased density of native understorey vegetation has been observed following fire, leading to a decline in some populations. Species does not generally endure repeated disturbance or the consequential habitat changes (e.g. fire/wildfire). | • DEC (2008a)<br>• Evans *et al.* (2003)<br>• Brown *et al.* (1998) |
| *Eleocharis keigheryi* | • Field Observation: species can grow in areas that have been recently burnt, and can flower in the absence of fire (one plant up to 10 years since last fire). | • Evans *et al.* (2003)<br>• Brown *et al.* (1998) |
| *Epiblema grandiflorum var. cyaneum* | • Fire is considered detrimental if fire occurs between June to December (during vegetative and flowering stages).<br>• Autumn fire is thought to be the most appropriate for this species. | • Stack *et al.* (2000).<br>• Evans *et al.* (2003)<br>• Brown *et al.* (1998) |
| *Eucalyptus argutifolia* | • Fire is considered to be detrimental if fire frequency is less than every 5-8 years (the species flowers 3-4 years after regenerating from rootstock). | • Evans *et al.* (2003)<br>• Brown *et al.* (1998) |
| *Grevillea curviloba subsp. curviloba & G. curviloba subsp. incurva* | • Fire is considered to be detrimental if fire is too frequent, as it can deplet rootstock reserves and soil bank. | • English and Phillimore (2000) |
| *Trithuria occidentalis* | • Fire is considered to be detrimental if fire occurs during flowering (Sept-Nov) | |
| *Maranthius paralius* | • No information available | |

## *Wetlands*

Wetland soils and sediments are becoming more vulnerable to fire on the Swan Coastal Plain due to summer drying of surface sediments, which is affected by recent changes in climate, declining rainfall, increased use of groundwater by an increasing urban population, clearing of bushland, silviculture and horticulture, urban development (Horwitz *et al.* 2003; Horwitz and Smith 2005; Horwitz and Sommer 2005). Wetlands exposed to fire can result in the loss of organic matter and other chemical and physical changes (Horwitz and Sommer 2005). Semeniuk and Semeniuk (2005) state that along with the hydrology of a wetland, the potential for soils and sediments to combust is related to annual fluctuations in the water table, the longer term climatic patterns and the distribution of flammable material within a wetland.

Chapter 4 provides a detailed review of wetlands on the Swan Coastal Plain and in the GSS study area. Also included is a discussion of the impact of fire on wetlands and water quality as well as a discussion of wetland sediments, stratigraphy, hydrology, and other features that are important to consider in the management of fire in wetlands.

## *Invertebrates*

A study by Van Heurck and Abbott (2003) reviewed 31 studies on the impact of fire on invertebrates in south-west Western Australia, including five studies conducted in the coastal woodland that is found on the Swan Coastal Plain. From this review Van Heurck and Abbott (2003) found that invertebrate biodiversity was greatest where habitat heterogeneity is maximised (with habitat heterogeneity represented by a wide range of post-fire successional stages in the vegetation). After a fire, there is a short-term decline in the local species richness but an increase in landscape species richness. For example, beetles in dry sclerophyll forests and coastal woodlands of south-west Western Australia exhibited this pattern (Ladhams 1999; Van Heurck and Abbott 2003).

The woodlands and shrublands of the coastal regions of south-west Western Australia (Swan Coastal Plain) are characterised by a relatively uniform landscape of infertile sand dunes deposited by the receding ocean and a highly predictable climate. Due to these two features, the invertebrate fauna of the coastal woodlands on the Swan Coastal Plain is resilient to more regular and frequent fires (Van Heurck and Abbott 2003).

To conserve invertebrate diversity, Van Heurck and Abbott (2003) recommend a combined precautionary and adaptive approach to fire management across the landscapes of south-west Western Australia due to the high invertebrate diversity and the limited knowledge of their taxonomy and ecology. The authors suggest six strategies, including prescribed burning techniques that create small burnt patches in small remnants and habitats of fire sensitive relict species, and setting aside some long-unburnt areas. Van Heurck and Abbott (2003) also discuss future research priorities for invertebrate species in south-west Western Australia. These include researching the distribution of different species and increasing our knowledge about the fire tolerance of threatened invertebrate species.

## *Reptiles*

Studies of the impacts of fire on reptiles in south-west Western Australia are few (Bamford and Roberts 2003). In contrast to other studies, Bamford (1986; 1992; 1995) did not find a relationship between total species or number of captures, and time after fire in heathland and *Banksia* woodland habitats. He concluded that the overall effect of fire on reptiles was negligible, although a small number of species did exhibit clear post-fire seral responses. Some species were absent from early succession areas, while others were present in increased numbers, apparently favouring the more open ground. Whilst overall the assemblage did not change, fire effects may have been obscured by patterns of abundance across the landscape that were independent of fire history (Bamford and Roberts 2003). These studies focused on time since fire and did not examine fire intensity.

Dell and How (1995) examined the response of reptiles to wildfire at Kings Park and found that the longest unburnt sites supported the highest lizard diversity, while the most recently burnt sites were found to have the lowest lizard diversity. Species and abundance was lower in the first year post-fire but appeared to return to pre-fire levels by the second year post-fire. Migration from burnt to adjacent unburnt sites was apparent. Some reptile species in the wheatbelt of Western Australia appear to be vulnerable to single high intensity fire events, including the arboreal gecko (*Strophurus spinigerus*), an agamid lizard (*Ctenophorus reticulatus*) and a skink (*Ctenotis pantherinus*). By 4–5 years after fire, mature adults of the lizard and skink species recolonised burnt remnants with an adjoining unburnt habitat (Lambeck 1999).

### *Birds*

Burbidge's (2003) review of the impacts of fire on birds in south west Western Australia found that bird species richness is highest in long unburnt vegetation (15 years post-burn) but is also high in habitat for several years following a fire (Davis in prep.). Only honeyeater species richness is reduced in burnt habitat for the first 3 years following fire. In the GSS study area a proportion of long unburnt vegetation (> 15 YSLF) is therefore important to facilitate the maximum species richness possible in a vegetation/habitat type. However post-fire habitat is favoured by species that have a preference for open habitat, e.g. birds of prey. Many species that prefer open habitat will remain in a burnt area and be the dominant species for 2-6 years post fire (Burbidge 2003).

The abundance of birds decreases to very low levels immediately following a fire but usually recovers within 2-3 years (Burbidge 2003). Insectivores generally increase in abundance after fire and can exceed pre-fire abundance for up to 7 years. Conversely, nectarivores decline following fire due to the reduction in the number of flowering *Banksias* in the burnt area. However it is fire intensity that is one of the biggest determinants of post-fire richness and abundance. Low intensity burns have the least impact on bird ecology (Burbidge 2003).

There are no species in south-west Western Australia that only occur in long-unburnt vegetation (Burbidge 2003), however Bamford (1985) found that the western thornbill (*Acanthiza inornata*), shining bronze-cuckoo (*Chrysococcyx lucidus*) and scarlet robin (*Petroica multicolor*) were more common in *Banksia* woodland unburnt for 11-12 years. Variations in abundance are linked to habitat structure and consequential foraging opportunities. Wooller and Calver (1988) noted significant decreases in the abundance of white-breasted robin (*Eopsaltria Georgiana*), golden whistler (*Pachycephala pectoralis*), splendid fairy-wren (*Malurus splendens*) and white-browed scrub-wren (*Sericornis frontalis*) following fire and surmised that this was due to changes in the vegetation structure and prey. Frequent fire causes population decline of splendid fairy-wrens (*Malurus splendens*) by altering habitat and food availability (Brooker 1998) and high intensity fire reduces cover, facilitating an increase in nest predation. In the GSS study area open post-fire habitats favour some species but in the long-term, a proportion of long-unburnt sites are required to maintain some key species that will otherwise decline due to food shortage and higher rates of predation.

Active nests are able to recruit fledglings following a low intensity fire (Kimber 1974) although recruitment success diminishes for some species such as splendid fairy-wrens (*Malurus splendens*) if fire is too intense (Rowley and Russel 1990). Moreover intense fire in small remnants (under 2000 ha) can cause the extinction of some species such as the splendid fairy-wren (*Malurus splendens*) (Brooker and Brooker 1994) and western thornbill (*Acanthiza inornata*) (Recher 1997). Nest predation and brood parasitism can also increase after intense fire (Russell and Rowley 1993). In the GSS study area it is therefore imperative that for prescribed burns less than 2000 ha in size that the intensity of the burn is low.

Following an extensive high intensity fire or burn, species recolonise from surrounding unburnt patches. Recolonisation rate varies, being slow for some species such as the grey shrike-thrush (*Colluricincla harmonica*), which only returns 3 years after fire. Sedentary species in remnants are more likely to become extinct following fire than agile species (Burbidge 2003). Adjacent unburnt areas are essential to provide source populations for recolonisation. Extensive fires risk the local extinction of sedentary species.

In the GSS study area, a mosaic of fire ages, including maintaining a proportion of habitat that is at least > 15 years since last fire is optimal. This mosaic would provide habitat for bird species that benefit from fire (open habitat), as well as species favouring long unburnt habitats, and these unburnt patches would also provide source sites for re-colonisation of post-fire landscapes. High intensity fires or burns as well as too frequent fire in isolated remnants have caused the extinction of sedentary species such as the splendid fairy-wren (*Malurus splendens*). Fire prevention in isolated remnants, possibly through mosaic burning, may be critical to ensuring the regional persistence of these species at a regional level.

### *Mammals*

Studies undertaken by Bamford (1985; 1986) examined the impact of prescribed burns in *Banksia* woodland and assessed areas of varying fire ages at Mooliabeenee Nature Reserve. Survival of most mammal species was high in places of uneven (patchy) burn as species were able to survive by moving to unburnt areas. The mean number of captures in the first year after fire was significantly less than that of all subsequent years. House mice (*Mus musculus*) and little long-tailed dunnarts (*Sminthopsis dolichura*) were more abundant 0 – 3 years after fire, ash-grey mice (*Pseudomys albocinereus*) between 3 – 6 years after fire, and honey possums (*Tarsipes rostratus*) and western pygmy possums (*Cercartetus concinnus*) were more abundant in 11 year old vegetation. The pre-fire abundance of each species and the intensity of the fire/burn were two significant factors determining the impact upon the species present (Bamford 1986).

Kitchener *et al.* (1978) proposed that too frequent burning of vegetation may have threatened the persistence of mammal species on the Swan Coastal Plain. In the 1978 study, ash-grey mice (*Pseudomys albocinereus*) and honey possums (*Tarsipes rostratus*) were only trapped at two sites (Mullaloo and Burns Beach) in patches of vegetation that had remained unburnt during extensive, high intensity fires in 1975 – 1976. The area had been extensively burnt previously (1971-72) and the patches were thus estimated to be six years post-fire age. High intensity fire in small isolated vegetation remnants on the Swan Coastal Plain may lead to local extinction (How and Dell 2000). Those individuals that survive fire are prone to starvation or predation due to lack of cover and food and there is little likelihood of colonisation from elsewhere in the urban matrix.

## Fire management for biodiversity conservation

One of the major challenges in current fire management is the use of appropriate fire regimes to protect and promote biodiversity. The relatively fine-scale burning patterns used by Aboriginal people prior to European colonisation probably maintained relatively high levels of heterogeneity of seral stages and environments, and consequently supported high biodiversity (Woinarski 1999). The impact that rapid change over the last 200 years has had upon biodiversity is not fully understood, although the altered fire regimes implemented since European settlement have certainly contributed to a decline in abundance and range contraction of many plant and animal species, sometimes to extinction. Modified fire regimes may also alter successional patterns (Wilson 2003). Other disturbances such as fragmentation, disease, introduction of exotic plants and animals, and land-uses that restrict the use of fire (Bowman 2003; Hobbs 2003) compound the effect of fire regimes.

Fire can alter species assemblages and composition, vegetation structure, habitat characteristics and nutrient cycling. It may therefore be that a diverse fire regime may promote biodiversity (Bradstock *et al.* 1995; Burrows 2008; Burrows and Wardell-Johnson 2003; Edwards *et al.* 2001; Martin and Sapsis 1992; Panzer 2003; Parr and Andersen 2006). However, the relationship between fire and biodiversity may not be that simple. An increase in fire frequency in unproductive environments (where populations grow and recover slowly, e.g. Kwongan) can lead to a decrease in plant and animal diversity, while in productive environments (e.g. Karri forest), an increase in fire frequency can increase plant and animal diversity (Huston 2003). As the productivity varies greatly in the ecosystems of the south-west of Western Australia, so does the effect of a particular fire regime on the biodiversity of these ecosystems (Huston 2003). In addition, fire can have an impact on nutrient cycling and increase the spatial variability of nutrients in an ecosystem by mobilising nutrients such as nitrogen, phosphorus and calcium (Adams *et al.* 2003). The diversity and abundance of fungi can also affect nutrient cycling and the community structure and species diversity of flora and fauna (Robinson and Bougher 2003). While the role of fire and fungi is not well understood in the south-west of Western Australia, recent research showed that fire significantly impacts the physical environment where fungi is found (Robinson and Bougher 2003).

Therefore a diverse fire regime may promote biodiversity in a landscape because of its diverse influence on species assemblages and composition, on vegetation structures, habitat characteristics and nutrient cycling (Burrows 2008). However, ecological fire management must be based on an understanding of the processes of flora and fauna responses to fire, and should be developed so that no species decline or become extinct (Wilson 2003).

In environments that are fire-prone but support fire-maintained ecosystems, managers should manage fire to both protect human life and property, and protect and conserve biodiversity (Burrows 2008). Therefore fire regimes should be planned to achieve defined goals for the maintenance of biodiversity. The development of a plan for prescribed burning to manage accumulated fuel load and to manage biodiversity should take into account knowledge of both fire behaviour (intensity, type, frequency, patchiness and season) and of the life histories of plants and animals (Attiwill and Wilson 2006).

In practice, however, decisions about fire management are frequently made with limited knowledge. There is often a gap between knowledge of fire behaviour and fire ecology and its on-ground application and use in the management on fire. Managers are often uncertain or confused about which are the most appropriate fire regimes to use for a particular area, as fire ecology is complex, and not complete, and must be translated into simple management options for managers (Burrows 2008). In addition, there is a challenge in translating results from fire and biodiversity research into prescriptions for fire management regimes (Wilson 2003). Burrows (2008) stresses the need to be consistent with the precautionary principle when managing fire – a management agency should not avoid taking action to protect biodiversity (and other assets) from fire even though there is uncertainty associated with the impact of fire on an ecosystem. Rather, the best available and current information should be used to inform good decision making.

As there is no single optimum fire regime that will meet all management objectives, Burrows (2008) discussed several approaches to planned burning that are consistent with the precautionary principle and that can be applied to various ecosystems under various circumstances when the available information is considered. These include the following and will be discussed in more detail below:

- maintaining diverse fire regimes based on vital attributes and life histories of threatened, keystone or focal species
- applying regimes that are diverse in terms of frequency, patchiness, season and intensity, therefore producing greater habitat diversity and creating finer grained habitat mosaics
- applying regimes that aim specifically to manage fuel accumulation.

## Fire regimes based on vital attributes

These regimes use knowledge of the vital attributes and life histories of threatened, keystone or focal species in order to predict the response of vegetation to fire to help establish an ecologically appropriate fire regime (Burrows 2008). Vital attributes include the environmental conditions required for regeneration and re-establishment, the juvenile period and longevity of the longer-lived species, the method of persistence at a site following fire, and post-fire regeneration syndromes (Burrows 2008; Noble and Slatyer 1980). These vital attributes can be used to determine the interval between prescribed fires for a particular ecosystem. For example, Burrows *et al.* (2008) uses juvenile period (time to first flowering after fire) as a biological indicator to guide minimum intervals between fires to conserve plant diversity as well as to manage fuel accumulation. Burrows *et al.* (2008) recommends that this minimum interval should be approximately twice the juvenile period of the slowest maturing fire sensitive plant species in the community (where juvenile period is the time taken for at least 50% of the population to reach flowering age).

In 1998 in Victoria, the Department of Sustainability and Environment and Parks Victoria implemented a new approach to manage fire for biodiversity conservation on public land. This approach used relevant vital attributes of key fire response species (i.e. species with vital attributes that indicate that they are most likely to be affected by very frequent and very infrequent fires) to determine the maximum and minimum time between fires that maintained all species in one or more vegetation communities. This approach, uses the time period of the species that takes the longest time to reach maturity as a minimum fire interval (Department of Sustainability and Environment 2004).

Fire regimes can also be based on the vital attributes of vertebrate fauna (e.g. honey possum, quokka and quenda) (Burrows 2008). However, determining fire regimes based on the vital attributes of fauna is less clear due to factors such as mobility, predation and habitat availability affecting the distribution of fauna species (Burrows 2008). Information regarding the sensitivity to fire of invertebrate fauna is also needed to determine ecologically appropriate fire regimes (Van Heurck and Abbott 2003).

## Fire regimes to create diverse habitats and mosaics

This approach is based on the idea that ecosystem integrity, and therefore biodiversity, is enhanced by the maintenance of a diversity of post-fire ages and fine-scale mosaics of fire history (Bradstock *et al.* 1995; Burrows and Wardell-Johnson 2003; Edwards *et al.* 2001; Martin and Sapsis 1992; Panzer 2003; Parr and Andersen 2006). It has been suggested that a common and stable pattern of vegetation fire ages (time since last fire distribution) includes small patches of older, long unburnt vegetation within a matrix of larger patches of younger vegetation (Burrows 2008; Weir *et al.* 2000).

More recent approaches to fire management include trying to produce mosaic patterns similar to patterns produced by natural (pre-European) disturbance regimes (Tolhurst and Friend 2001; Weir *et al.* 2000), and introducing frequent fires (2-3 year intervals) into the landscape (e.g. south-west Western Australia) to create, maintain and promote fine-scale habitat mosaics that incorporate a range of interlocking post-fire seral stages, which will promote biodiversity and reduce the severity and impact of wildfires (Burrows 2008).

## Fire regimes to manage fuel accumulation

This approach is based on the idea that fuel quantity and arrangement are the only factors that can be practically managed in fire management (Burrows 2008). Therefore the management action taken is fuel reduction burning (prescribed burning).

As mentioned above, the Department of Sustainability and Environment and Parks Victoria implemented a new approach to manage fire for biodiversity conservation on public land in 1998, using vital attributes of vegetation to determine the minimum and maximum fire interval for an ecosystem. In addition, these agencies use information about the fire history in an area to determine the known age-class distribution of a particular vegetation type to determine how often and where to burn. The known age-class distribution of a particular vegetation type is compared to a ‘theoretical’ age class distribution, which is calculated using the vital attributes of the key fire response species. The theoretical age-class distribution helps managers to identify which age classes are under represented (and therefore need protection from burns) or over represented (and therefore are considered for prescribed burning) (Department of Sustainability and Environment 2004).

## DEC Swan Region fire management

Most of the fire management research within DEC has been conducted in the forested areas of south-west Western Australia. In this Mediterranean climate region of Australia, fire is an integral part of the ecology of plant and animal communities due to several factors, including seasonal drought, vegetation that is flammable, and regular ignition of fires from lightning storms (McCaw *et al.* 2003). Prescribed fire has been used in south-west Western Australia since the mid 1960s, initially for management of fire to protect timber in forests, but more recently for biodiversity management and conservation (McCaw *et al.* 2003). Prescribed burning will not prevent wildfires, but it can reduce the intensity of wildfires, the frequency of wildfire and the size of the area that gets burnt (Burrows *et al.* 2008). However, there are concerns that prescribed burning for fuel reduction could be ecologically damaging in the long term (Burrows *et al.* 2008). The history of the research into fire behaviour and fire management in south-west Western Australia was reviewed by McCaw *et al.* (2003) and it emphasised that that fire management is most effective when supported by an understanding of fire behaviour in the area.

To help manage wildfire risk, DEC uses the 'Wildfire Threat Analysis' to inform decision-making by fire managers of the most effective burn plan strategies to minimise the risk of wildfire impacts. The Wildfire Threat Analysis process uses the following information to make decisions: (i) the likelihood of wildfires occurring; (ii) the values at risk; (iii) the potential wildfire intensity; and (iv) the ability to suppress the fire.

A number of guiding documents for burn planning include:

- Code of Practice for Fire Management (which provides a framework for fire management procedure and practice on land managed by the Department of Environment and Conservation in Western Australia)

- Fire Management Guidelines: includes information for land managers on habitats and flora and fauna (for example, honey possum, organic soils, and black cockatoos)

- Fire Management Policy (Policy Statement 19). The objectives this policy are:
    - The Department will manage prescribed fire and wildfires on lands managed by the Department to protect and promote the conservation of biodiversity and

natural values whilst also providing for protection of human life and community assets. The Department will also promote fire management that protects biodiversity on lands not managed by the Department.

- Fire management will be planned and implemented in partnership with other landowners and land managers, fire authorities and the community. The Department will implement an informed and balanced approach to risk management. A variety of fire regimes incorporating different frequency, intensity, season and scale will be applied at the landscape scale on lands for which the Department has a fire management responsibility.

DEC undertakes hand burns for areas < 500 ha and aerial burns for larger areas, whereby aeroplanes deploy cap or gel incendiaries to effectively and efficiently burn out the area. Long unburnt areas are also set aside as Fire Exclusion Reference Areas (FERA), with the intention to prevent fire (be it wildfire or prescribed burning) from impacting these reference sites.

## Prescribed burning

Since July 2003, DEC has assumed responsibility for the coordination and on-ground management of fire risk prevention on 89 million ha of land in Western Australia. To manage this wildfire risk, DEC undertakes prescribed burning across 45 120 ha of native vegetation and 24 620 ha of pine plantation (for Forest Products Commission) on DEC-managed estate and Unallocated Crown Land (UCL) on the Gnangara groundwater system, primarily for the protection of life and property on neighbouring lands and also for the protection of assets on the conservation estate itself (plantation timber, biodiversity, recreational assets). The first step towards implementing a prescribed burning regime is to prepare a Master Burn Plan. This plan identifies the areas planned for prescribed burns, the scheduling of these burns and the season (Autumn or Spring) they will be completed in the forthcoming year, and an indication of the schedule of prescribed burns for the next three years.

Each prescribed burn requires documentation and approval. For each scheduled prescribed burn, a Prescribed Fire Plan is prepared. This plan identifies the preliminary work required to prepare for the burn, including desktop (and sometimes field) surveys for threatened fauna and rare flora species within the burn boundary, determination of fuel loads to assess the viability of the area carrying a burn, and notification of the proposed burn to neighbouring landholders adjacent to the burn boundary. Assets such as recreation sites and threatened flora populations within the burn boundary may need to be excluded from the burn. The Prescribed Fire Plan identifies the weather conditions that are required to achieve the stated objectives of the burn, and a ‘prescription’ is prepared that sets guidelines and constraints for the weather conditions and ignition strategy at the burn site. The Prescribed Fire Plan also requires information to be collected during and after the fire.

## Prescribed burning in the GSS study area

Currently, as part of the master burn plan for DEC’s Swan Coastal District, the native vegetation and pine plantations on DEC-estate, as well as Unallocated Crown Land outside gazetted towns, are burnt during autumn and spring on an 8 to 12 year rotation. Any proposal to increase the frequency of burning of native vegetation (principally *Banksia* woodlands and seasonal wetlands) will require additional research to determine the possible adverse impacts upon the biodiversity of these unique remnant bushland areas.

There are approximately 9700 ha of Defence and 9970 ha of UCL lands on the eastern side of the GSS study area. The Defence Department funds small areas of strategic prescribed burning around their infrastructure and military sites. The fire management (prescribed burning) of UCL has been recently transferred from FESA to DEC although no additional funding has been provided to uphold this responsibility in the outer Perth area. However there is the operational capacity to burn a greater proportion of UCL and Defence land if resources are provided.

Prescribed burning across Swan Coastal District (including the GSS study area) is planned 7 years in advance at both the District and Regional level (Swan Region, DEC). Prescribed burns are strategically planned to try and produce a mosaic of different fuel ages and burn seasons. Indicative Burn Plans are produced by DEC’s Swan Coastal District prior to each burn season, and Prescribed Fire Plans are compiled for each of the imminent prescribed burns.

Three types of burns are undertaken throughout the pine plantations in the GSS study area, including: the burning of redtops (crowns and side branches removed from harvested timber); the burning of needlebed under pine trees stands (to reduce the fuel load); and clearing burns.

## Wildfire in the GSS study area

DEC is the hazard management agency for DEC-managed land outside gazetted areas and the Perth metropolitan area. Plantations and DEC-managed land north of Perth are the responsibility of DEC. There are 100-250 wildfires per year in the Swan Coastal District, with a decrease in the number of wildfires in the last 30 years. Most wildfires result from arson and the burning of stolen cars, as well as escapes from agricultural / horticultural land. Few wildfires (10-15%) occur seasonally from lightening.

The wildfire detection system across south-west Western Australia includes manned fire towers (Pinjar Tower, Bickley Tower and Wabling Tower), which are used to triangulate the position of observed fires, and spotter aircraft (14 scouts in south-west Western Australia) that fly over forested regions to detect fires.

During the fire season, October to April, there are rostered crews on call 24 hours a day. From the Swan Coastal District headquarters in Wanneroo, water bombers are coordinated for the whole of Western Australia. Wanneroo also manages forward bases for these water bombers (i.e. airstrips with water tanks installed for close proximity to plantations).

# Summary of DEC–GSS projects (2007–09)

## Project 1. Effects of time since fire on ground-dwelling vertebrates

Technical report reference: Valentine L. E., Wilson B. A., Reaveley A., Huang N., Johnson B. & Brown P. H. (2009). *Patterns of ground-dwelling vertebrate biodiversity in the Gnangara Sustainability Strategy study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

The GSS fauna survey was undertaken to assess the current occurrence and distribution of terrestrial vertebrate fauna across the GSS study area, to examine patterns in biodiversity with landscape features and to assess the susceptibility of taxa and communities to threatening processes such as declining groundwater levels and fire. One of the specific aims was to examine the response to time since fire by reptile and mammal fauna in the GSS study area. The main findings are summarised here. The relationship between floristics and time since fire in the *Banksia* woodlands of the GSS study area has also been examined but the results are preliminary and will be presented in future reports.

The GSS fauna survey included a ground-dwelling vertebrate trapping program across 40 sites in the major areas of continuous remnant bush land in the northern and eastern areas of the GSS study area. Sites were selected to represent the major landform units (Quindalup, Spearwood and Bassendean), vegetation communities (*Banksia* woodland, coastal scrub, jarrah forest, tuart forest and *Melaleuca* wet or dampland) and time since last fire. As the time since last fire varied considerably among sites (3-36 years since last fire), the sites were grouped into two major categories: Young, those recently burnt (< 11 years since last fire); or Old, those long unburnt (> 16 years since last fire). Where possible, the 40 sites were grouped into 20 paired sites, located between 300 – 500 m distance apart. Sites included pitfall arrays as well as Elliott and Sheffield cage traps, and were surveyed in Spring 2007, Autumn 2008 and Spring 2008.

The results indicate that the response of reptile communities to time since fire varied among different combinations of vegetation type and time since fire. Although vegetation type strongly influenced community composition, time since fire also influenced reptile assemblages. Our surveys showed that overall reptile abundance, as well as the abundance of some specific species, was higher in long unburnt sites (Figure 7.1).

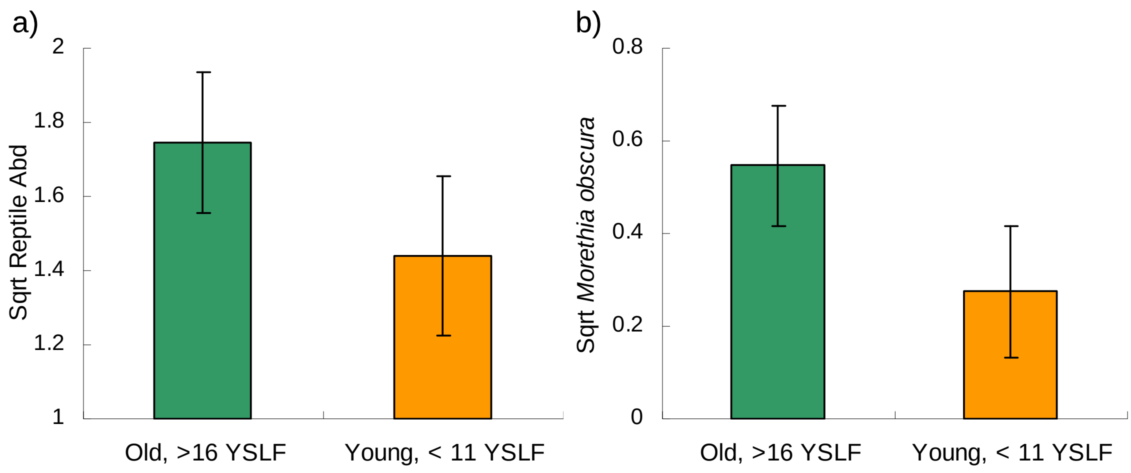


Figure 7.1: Significant differences in the mean (± 95%CI) of a) reptile abundance and b) abundance of *M. obscura* between long unburnt (Old) and recently burnt (Young) sites.

In addition, the majority of the burrowing snake species, including the Priority listed elapid *Neelaps calonotos* were captured at sites of old fuel age. This perhaps reflects a difference in resource availability between recently burnt and long unburnt sites. In previous studies, skink-consuming elapids tend to be absent, or in lower abundances in recently burnt habitat (Valentine and Schwarzkopf 2009).

Specific analyses examining the interaction between vegetation type and fuel age categories were undertaken for *Banksia* and *Melaleuca* sites. Our results indicated that reptile communities varied among different combinations of habitat and fuel age. Different reptile species tend to prefer different habitat attributes (Letnic *et al.* 2004), and these attributes will be in different supply in different vegetation types, and may be altered by burning. The differences between fuel ages were particularly pronounced in *Melaleuca* sites. Young fuel age sites in *Melaleuca* habitat tended to contain fewer reptiles, and had few species associated with them. Although some differences were also detected between fuel ages in *Banksia* sites, they were less pronounced, and young fuel age *Banksia* sites often had species common in old fuel age sites (e.g. *Morethia obscura*). Particular species of fauna were associated with fuel age sites in different ways within the two habitat types. For example, reptile abundance was correlated with fuel age, however, this correlation was only significant within *Melaleuca* sites. This suggests that the response of reptiles to fire age is dependent upon habitat type.

Changes in the abundance of reptiles following burning is often linked to fire-induced changes in the resource availability of the post-fire environment (Friend 1993; Masters 1996). Because reptiles tend to occupy sites with suitable thermal, shelter, and food resources (Friend 1993; Letnic *et al.* 2004; Masters 1996), burning may have modified elements of the habitat in a manner undesirable to some species. The long unburnt sites contained deeper piles of litter, and those species with a preference for deeper litter, were observed in high abundances in the long unburnt sites. Typically, litter-associated lizards respond strongly to the removal of vegetation and are usually observed in high abundance in the least-disturbed sites, and their density is often correlated with variables of vegetation cover (e.g. litter cover; Greenberg *et al.* 1994; Masters 1996).

The succession of fauna with time since fire was also examined and highlighted that the response of reptiles to fire in *Banksia* woodland are fairly complex. Reptile species tended to respond in different patterns to time since fire, with relative abundance estimates peaking at every fire age category for at least one species of reptile (Figure 7.2). Several species preferred recently burnt sites, whilst others were most abundant in intermediate fuel age sites, and still other species in long unburnt sites (Figure 7.2). Furthermore, several species displayed a cyclic response to time since fire, with relative abundances peaking in both recently burnt and long unburnt sites (Figure 7.2). This indicates that a diverse range of post-fire habitat is necessary to cater for the species rich reptile fauna in the GSS study area.

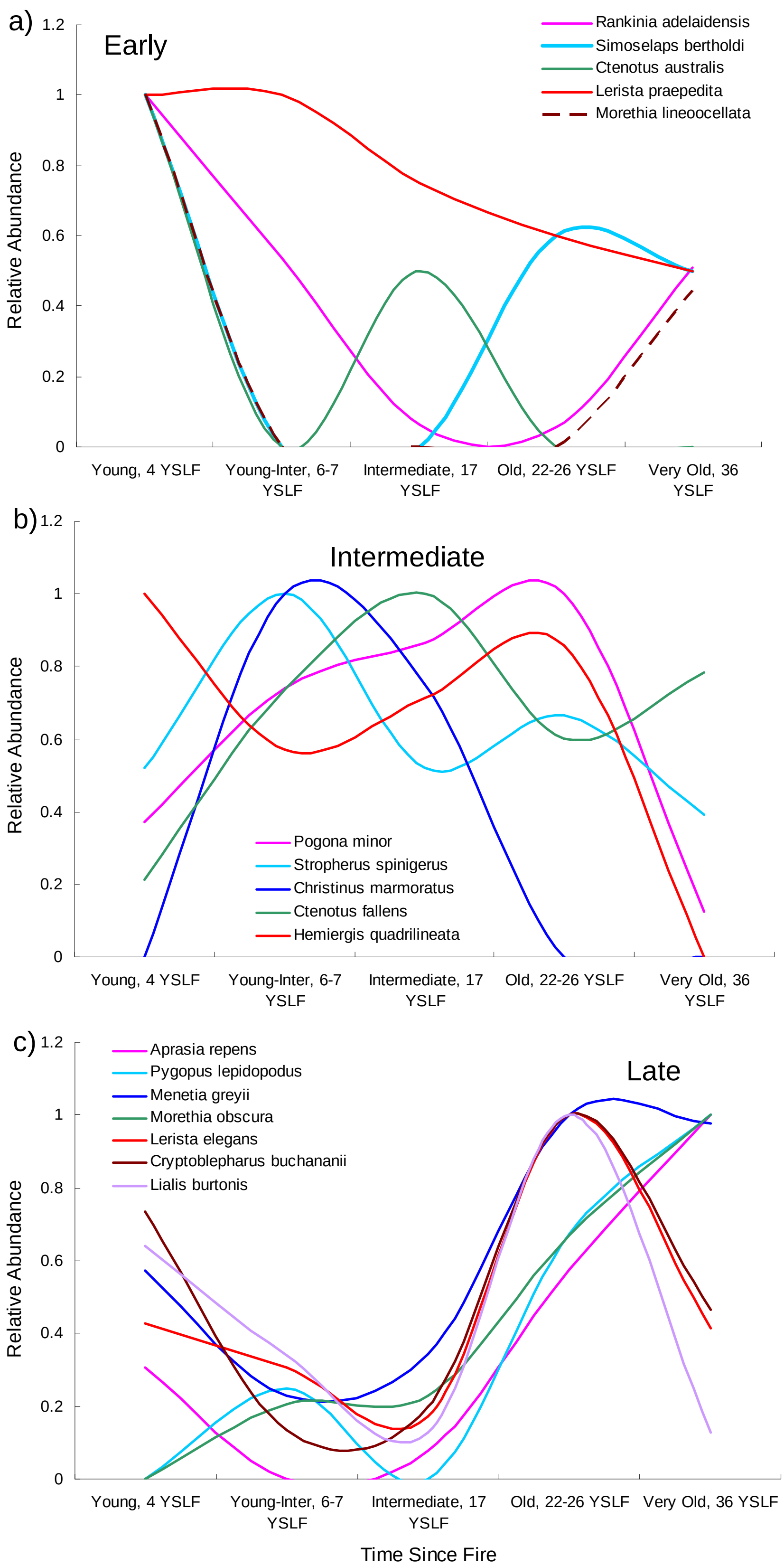


Figure 7.2: Significant differences in mean (± 95%CI) a) reptile species, b) species abundance, and c) selected individual species abundances between *Banksia* and *Melaleuca* sites, showing the interaction with fuel age where applicable.

The response of mammals to fire was fairly clear. The introduced house mouse (*Mus musculus*) preferred more recently burnt sites, and is often associated with disturbed habitat. In contrast, honey possums (*Tarsipes rostratus*) were more abundant in older sites, with peaks in relative abundance at sites 20 – 26 YSLF (Figure 7.3). Although honey possums (*Tarsipes rostratus*) are known to return to burnt areas within 2 – 4 years since fire (Bamford 1986; Everaardt 2003; Richardson and Wooller 1991), higher densities are typically recorded in older vegetation, with peaks in abundance in vegetation 20 – 30 years since last burnt (Bradshaw *et al.* 2007; Everaardt 2003). Our results were very similar, with low abundance in recently burnt sites (< 7 YSLF), followed by an increase in abundance as time since fire increased. However, in the *Banksia* woodland in the GSS study area we also noticed lower abundances in sites that have remained unburnt for a very long time (> 36 YSLB). Honey possums are dependent on nectar and pollen, particularly from plants of the Proteaceae, Myrtaceae and Epacridaceae families (Wooller *et al.* 1984). Capture rates of honey possums are closely linked to food sources (Bradshaw *et al.* 2007) and have been correlated with the densities of flowers and the flowering periods of *Banksia*s (Everaardt 2003). Hence, the impact of fire on honey possums will be related to the post-fire responses of target food species (Bradshaw *et al.* 2007).

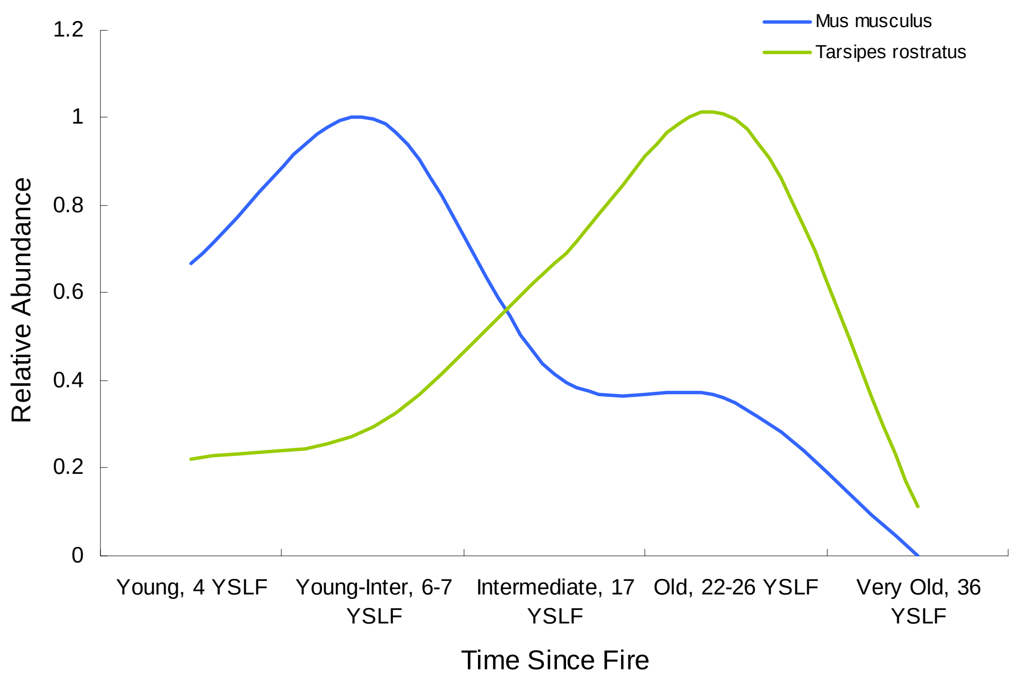


Figure 7.3: Successional responses of *Mus musculus* and *Tarsipes rostratus* (using relative abundance estimates) to time since fire.

# Project 2. Fire and the *Banksia* woodlands of the Swan Coastal Plain – fuel reduction burns and water recharge on the Gnangara groundwater system

Technical report reference: Reaveley, A., Bleby, K., Valentine, L., and Wilson, B. (2009). *Fire and the Banksia Woodlands of the Swan Coastal Plain – Fuel Reduction Burns and water recharge on the Gnangara mound: Preliminary Report*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

As part of the GSS, modifying the current burn regime by increasing the frequency of burning native vegetation on Crown land above the Gnangara groundwater system has been suggested as a cost effective technique that may increase groundwater recharge (Canci 2005; Yesertener 2007). CSIRO is undertaking an adaptive management project from 2008-2010 to examine the hypothesis that burning increases groundwater recharge and their work is not further discussed here. However, prior to the potential application of increased burn frequency as a management option, the biodiversity consequences of burning must be understood and the water yield and biodiversity balance quantified. DEC has begun a complementary adaptive management project to assess the impacts of burning to the biodiversity of *Banksia* woodland, in particular the rate of vegetation regeneration, and the effect of grazing, following a burn.

CSIRO's trial site to measure the groundwater recharge hypothesis was in Unallocated Crown Land (UCL) (Caraban UCL) in the northern end of the GSS study area, in an area proposed for a prescribed burn by DEC's Swan Coastal District. The total area of the trial site was 754 ha and 23 years since last burnt. The eastern side of the site was burnt on 6 June 2008, leaving the western side unburnt as a control (Figure 7.4). In conjunction with the groundwater recharge trial, the preliminary monitoring of vegetation regeneration following the burn and the compounding effect of grazing was undertaken by DEC. This was measured by establishing two adjacent pairs of 75 x 75 m plots, one pair on the burnt side, and one pair on the control side, with one plot in each pair fenced to exclude grazing following the burn.

Six vegetation attributes were recorded before the burn, repeated one month and six months after the burn: (1) floristic assessment; (2) assessment of juvenile period and response to fire; (3) vegetation structure; (4) canopy cover; (5) vegetation, soil and litter cover; and (6) litter depth. The impact of grazing was estimated by measuring the number and identification of scats and the presence or absence of evidence of grazing.

It needs to be noted that the following results are preliminary as they are based on the first seven months following the burn. This project is ongoing and no conclusions can be drawn at this stage.

A total of 72 species flowered in the 7 months following the burn. In the fenced burnt plot 40 species (29%) had reached juvenile period, and 26 species (19%) had reached juvenile period in the unfenced burnt plot. Of the species that had reached juvenile period, most (95%) were resprouters. Only 5% were annual re-seeders. No perennial re-seeders reached juvenile period.

In the burnt plots, vegetation cover 6 months post-burn was significantly lower than prior to the burn, as would be expected. In the unburnt plots, vegetation cover was greater in the grazed (unfenced) plot, possibly caused by grazing promoting some localised growth of affected vegetation. The removal of litter during the burn was still evident at 6 months post-burn, with litter cover significantly lower and soil cover significantly higher in the burnt plots than the unburnt plots. There was also a significant effect of the burn on vegetation structure (0-20 and 20-40 cm), due to the removal of low level vegetation in the burnt plots.

Grazing was more evident in the burnt unfenced plot (than the unburnt unfenced plot), indicating that kangaroos were more attracted to the succulent new growth on the burnt side of the trial. At 6 months post-burn the litter depth in the fenced burnt plot was significantly greater than in the unfenced burnt plot, suggesting that grazing may influence the build-up of litter (fuel accumulation rate). Future work should include determining fuel accumulation rates over time in the grazed and ungrazed burnt plots.

Although the data collected on juvenile period and post-fire regeneration strategies are preliminary, this is the first step in the process to assess the juvenile period of flora species for the purpose of determining the appropriate fire interval (burn regime) for *Banksia* woodland on the Swan Coastal Plain. Further study is required to fully understand the juvenile period of key fire species (e.g. species that have long juvenile periods as well as species that require frequent fire) as an ecologically appropriate burn regime for the *Banksia* woodland on the Swan Coastal Plain should take into account the juvenile period of the slowest growing species. A comparison between the juvenile period of species found in both the jarrah forest (Burrows *et al.* 2008) and *Banksia* woodlands would also be useful to gauge climatic variation within species.

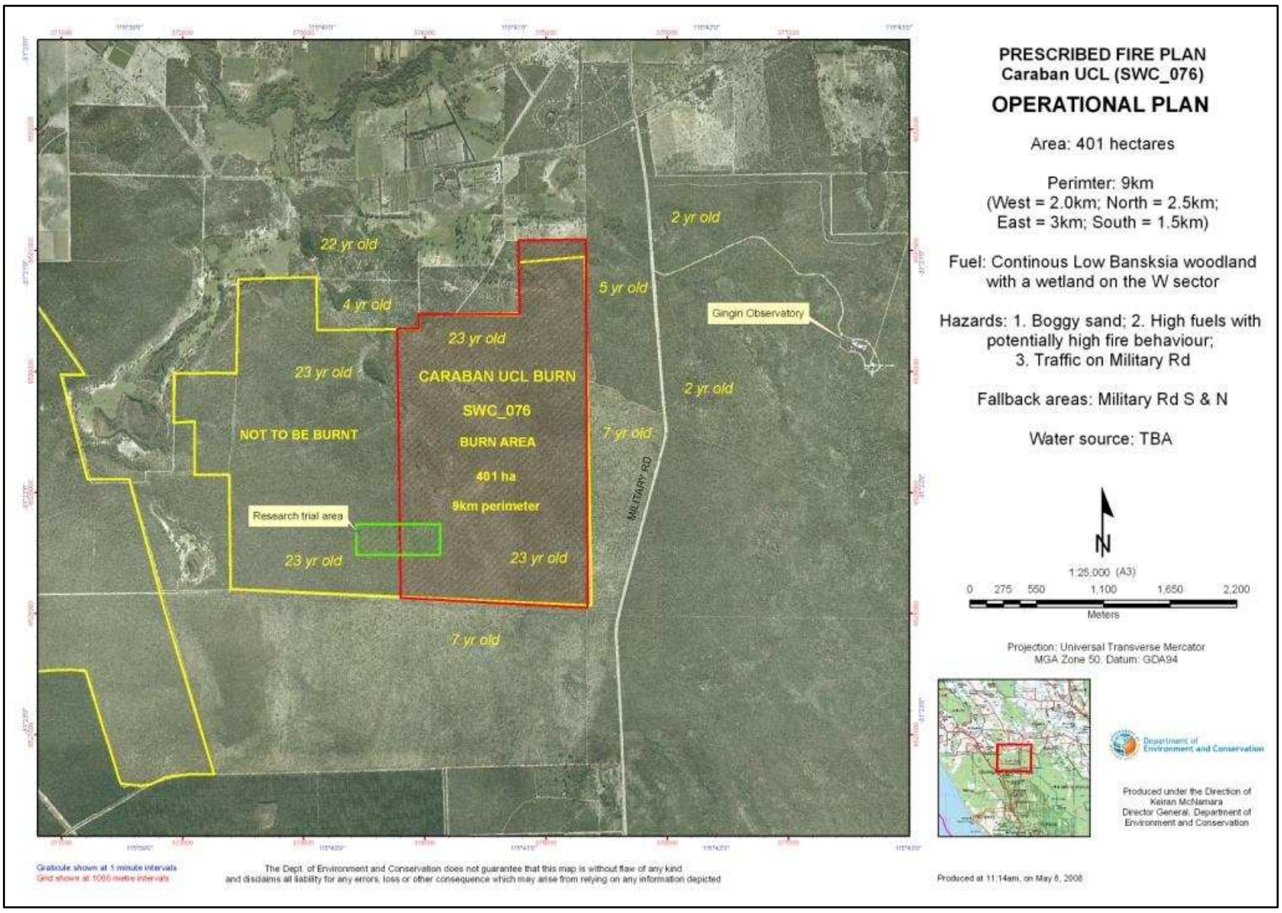


Figure 7.4: Outline of the burn trial area in Caraban UCL (yellow line), the outline of the area that was burnt (red line), and the outline of the treatment plots (green line).

# Appropriate fire regimes in the GSS study area

There has been little work on determining appropriate fire regimes for *Banksia* woodland on the Swan Coastal Plain, particularly compared to the work conducted in the forests of south-west Western Australia. Hopkins and Griffin (1989a) suggest that, to develop fire management guidelines for vegetation, data needs to be gathered on the rates of regeneration of vulnerable plant species and weed invasion due to fire. Fire interval may be an important factor for controlling weed invasion, indicating that a relatively long interval between fires would be appropriate (Hobbs and Atkins 1990).

Fire regime can have a significant impact on determining the population dynamics of *Banksia* spp. (Lamont *et al.* 2007). Non-sprouting species can be at risk of local extinction if an intense (non-patchy) fire occurs before young plants have produced many seeds. In addition, there is a greater likelihood of reaching the minimum numbers of seeds that are required for ‘population self-replacement’ with increasing fire interval (Lamont *et al.* 2007).

Burrows and McCaw (1989) suggest burning *Banksia* woodlands every 3-4 years to keep fuel-reduced buffer zones effective. However Hobbs and Atkins (1990) suggest that this fire regime would have a pronounced effect on the vegetation and would not be appropriate over wide areas. Hobbs and Atkins (1990) therefore recommend a balance between shorter fire intervals designed for fuel reduction and longer fire intervals that are more suitable for conservation. Data from their study indicates that a fire rotation of > 5 years, and preferably 5-10 years, would be more ecologically appropriate for the *Banksia* woodlands. Lamont *et al.* (2007) suggest that managed fire intervals of up to 10 years are too frequent for many *Banksia* woodland species to persist; however, fire intervals of greater than 25-50 years can lead to local extinction in *Banksia* species. Cowling and Lamont (1987) suggest that in order to maintain populations of *Banksia* species near Eneabba (approximately 240 km north of Perth), extensive tracts of scrub-heath should be burnt in autumn, prior to the onset of winter rain.

# Fire age distribution of *Banksia* woodland

An important component of developing appropriate fire regimes in the GSS is to understand the current distribution of habitat in different time since fire categories. There are a number of spatial data sets that can help achieve this objective. We used the Mattiske 2003 spatial mapping of site-vegetation types (see Chapter 2) and the DEC-corporate Year Since Last Fire (YSLF) spatial data sets to derive the distribution of different fire-aged *Banksia* woodland in the GSS (Figure 7.5) as part of preliminary analyses. We combined site-vegetation types that were predominantly described by Mattiske (2003) as *Banksia*, to represent the distribution of *Banksia* woodland on the GSS. Although the site-vegetation types mapping does not cover the entire GSS study area, the majority of unmapped habitat is on Quindalup dune systems. In addition, there were a number of habitat patches where the YSLF is unknown for *Banksia* woodland.

Within the GSS study area, the area (ha) of *Banksia* woodland vegetation varies between the different years since last fire (Figure 7.5). A relatively large amount of *Banksia* woodland vegetation exists that has been burnt recently (< 6 YSLF) compared to the area of *Banksia* woodland of between 6 and 16 YSLF. There is relatively very little *Banksia* woodland that is long unburnt (> 16 YSLF) (Figure 7.5). However, there is a large amount of *Banksia* woodland (~ 14 500 ha) for which the fire age is unknown. It is possible that some of this habitat may be in the older fire-age categories.

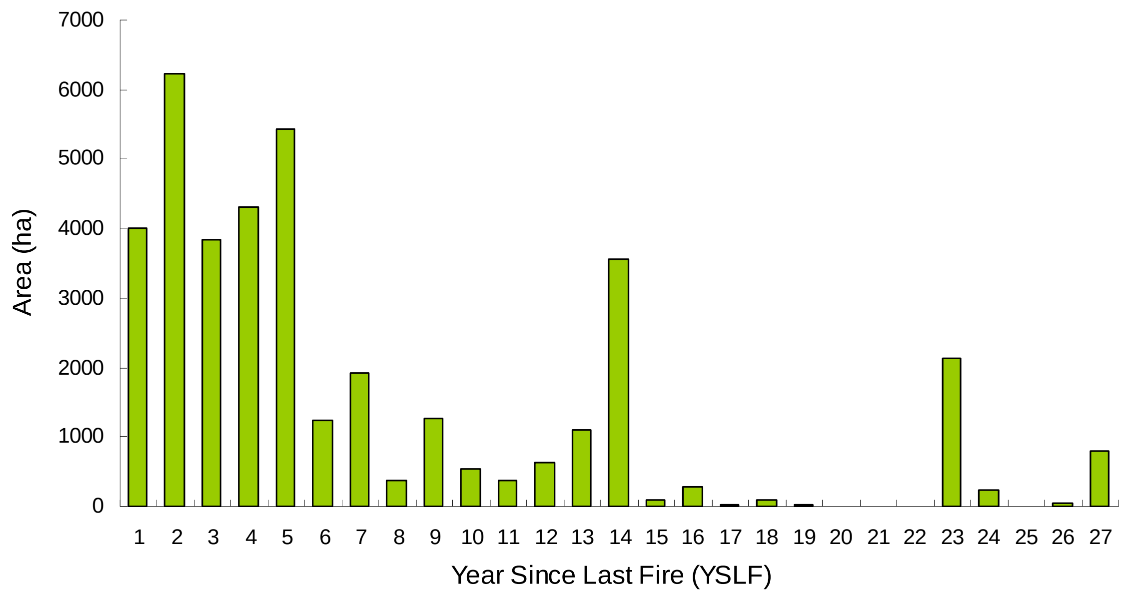


Figure 7.5: Distribution of *Banksia* woodland habitat in Year Since Last Fire categories.

As part of the continuing GSS fire project, the distribution of habitat in different time since last fire categories will be further explored.

## Future work on appropriate fire regimes

As part of the GSS, work on the impact of fire on biodiversity and groundwater recharge in the GSS study area will continue until June 2010. This chapter has presented only the work that has been completed to date. Future work is discussed below.

The Gnangara Sustainability Strategy (Government of Western Australia 2009a) states that there is 60 000 ha of native vegetation within the Crown estate and other significant areas of bushland throughout the system. To manage native vegetation for biodiversity conservation, the recommendation from the Gnangara Sustainability Strategy (Government of Western Australia 2009a) is to determine the optimum fire regimes of the *Banksia* woodland on crown land which will maximise groundwater recharge, yet maintain the biodiversity values of the GSS study area. This recommendation has arisen because previous groundwater bore monitoring conducted by the Department of Water has indicated that groundwater recharge may be enhanced by 0.5-2 m for 3-4 years following a fire (Canci 2005; Yesertener 2007). Further modelling and hydrograph analyses indicate an increase in groundwater recharge can be achieved by increasing the frequency of controlled burns in *Banksia* woodland (Vogwill *et al.* 2008).

The PRAMS (Perth Regional Aquifer Models) model, which models groundwater recharge on the Gnangara groundwater system, has incorporated aspects of this work. The PRAMS base case scenario incorporates a burning regime whereby 10% of total native vegetation in the GSS study area is burnt (each year) on a 10-year rotational burning system. For example, the model assumes that the 10% vegetation burnt in 2008 will be re-burned in 2018. However, the management of burning to promote recharge is still in a conceptual stage, and further work examining the practicalities and ecological impacts will be required during the GSS implementation phase.

As part of the ongoing DEC-GSS Fire Project, we are currently assessing fuel age distribution in native vegetation on DEC managed lands and UCL land; and wildfire occurrence according to distribution, cause, and season. This will include analyses of DEC fire maps and records and analyses of Landsat imagery (1972-2008) using trend analyses. Using this data, we can calculate the amount of area burnt and determine if the burn frequency has changed over time. Further, there is the opportunity to overlay Landsat imagery with vegetation complexes to calculate the proportion of each vegetation complex/type against fuel age. Calculation of the theoretical frequency distribution of different vegetation complexes can then be undertaken in order to assess the ecological aspect of the current fire regimes.

A number of projects have been initiated (2009-10) to review fire management operations on DEC managed Crown Land on the Gnangara groundwater system. They will address areas including the DEC fire planning framework and implementation on the Gnangara groundwater system; the weather conditions for fire management, and specific requirements for maintaining wetlands, threatened species and reference areas. They will also review the impacts of changes in land use (e.g. rural subdivisions, urbanisation) on fire management including changed fuel condition, ignition risk, values at risk, and cost of burning. A particular focus will be on the implications for fire management following the replacement of pine plantations with strategic ecological linkages and a parkland landscape, which is a major recommendation of the Gnangara Sustainability Strategy. The impact on fire management practices on threatening processes including climate change, *Phytophthora* dieback, drying wetlands, bushland fragmentation will also need to be assessed.

The aims of this future work will be to develop fire management and ecological burning regimes on the Gnangara groundwater system that incorporates diverse requirements including: (i) maintaining diverse fire regimes based on vital attributes and life histories of threatened or focal species; (ii) applying regimes of diverse frequency, season, intensity and scale, for optimal habitat diversity and fine-grained mosaics; and (iii) regimes that aim specifically to manage fuel accumulation (Burrows *et al.* 2008).

Our preliminary work has indicated that several species have a preference for habitat that is older than 10 YSLF. To ensure that appropriate habitat for these species is available, retention of habitat older than 10 YSLF, including habitat older than 15 YSLF, is strongly recommended.

# Discussion

Although the impact of inappropriate fire regimes has been identified as a major threat and barrier to biodiversity conservation (DEC 2009), information on the impacts on biodiversity on the Swan Coastal Plain are limited. Information on the fire regimes in the Gnangara groundwater system and current management practices that have, and currently are, impacting on biodiversity has also been limited. A consideration of the Gnangara Sustainability Strategy is to determine the optimum fire regime of the *Banksia* woodland to maximise groundwater recharge, yet maintain biodiversity values.

The aims of this chapter were thus to: review information on the role of fire in ecological communities and the responses of biodiversity to fire in Australia and in particular on the Gnangara groundwater system; examine the current fire management in protected estate, specifically for managed estate of the DEC Swan Region. In addition DEC GSS projects were undertaken to address identified knowledge gaps including: the effects of time since fire on ground-dwelling vertebrates (Project 1, 2007-08), and the assessment of the impact of fuel reduction burning on water recharge and biodiversity in *Banksia* woodland on the Gnangara groundwater system (Project 2).

Fire has been an integral part of the Australian environment for millions of years and although flora and fauna have traits that enable them to survive fire, they are adapted to specific fire regimes, determined by intensity, frequency, season and scale. Fauna species vary in their response to fire regimes and factors such as behaviour, food and shelter requirements affect their reaction to fire, the post-fire environment and habitat successional patterns. Fire regimes may have undesirable consequences including declines or local extinctions of flora and fauna taxa. If they occur too frequently for example, some plant species may be unable to produce seed, and animal species that require long unburnt habitat, may not have access to suitable habitat.

Although fire is a natural disturbance process in the GSS study area the impacts on biodiversity and communities are currently poorly understood and have largely been limited to information on impacts on flora (declared rare flora and threatened ecological communities) and on fauna in urban fragments. There have been few studies of the impacts of fire on frogs, reptiles and mammals on the Gnangara groundwater system or generally in south-west Western Australia (Bamford and Roberts 2003). While one study on the Gnangara groundwater system found that significantly higher lizard diversity occurred on the longest unburnt sites compared to recently burnt sites (Dell and How), another found no such relationship, although fire effects may have been obscured by landscape related abundance patterns (Bamford 1986; 1992; 1995). There has however been evidence of post-fire seral responses for a small number of reptile species and mammals where species such as house mice achieve maximal abundance in early succession (0-3 YSLF), ash-grey mice in mid succession (3-6 YSLF), and honey possums in late succession (11 YSLF) (Bamford 1986). There is evidence of too frequent or high intensity burning threatening mammal species on the Swan Coastal Plain (Kitchener *et al.* 1978) and causing local extinctions in small isolated vegetation remnants (Dell and How 1995) .

The DEC GSS project (2007-08) on the effects of year since fire on ground-dwelling vertebrates (Project 1) has advanced our understanding of the impacts of fire on vertebrate fauna in the Gnangara groundwater system. A significant finding was that overall reptile abundance, as well as the abundance of some specific species, was higher in long unburnt sites. Burrowing snake species, including the Priority listed *Neelaps calonotos* and lizards such as *Menetia greyii* were captured at sites of old fuel age. This is likely to reflect a difference in resource availability; vegetation and litter cover between recently burnt and long unburnt sites. In contrast several reptile species preferred recently burnt sites, whilst others were most abundant in intermediate fuel age sites. Few mammals were trapped however the response to post fire age of those that were captured was clear. While the introduced *Mus musculus* (house mouse) was more abundant in recently burnt sites *Tarsipes rostratus* (honey possums) had low abundance in recently burnt sites (< 7 YSLF), with a peak in relative abundance at sites 20 – 26 YSLF. These results were similar to previous studies in more southern populations where higher densities are recorded in older vegetation 20 – 30 YSLF (Bradshaw *et al.* 2007; Everaardt 2003). The evidence of post-fire seral responses for reptiles and mammals provides strong support for maintenance of a diverse range of post-fire aged habitat including long- unburnt habitat to provide for the species rich fauna in the GSS study area.

The collaborative project “Recharge and fire in native *Banksia* woodland on Gnangara groundwater system” is being undertaken by DEC/CSIRO for the GSS (2007-10). The

CSIRO component aimed to quantify the recharge under native *Banksia* woodland following fire, determine whether this was different from recharge under unburned bush, and estimate whether fire could be used as a realistic recharge management option. Preliminary data has been reported elsewhere (Silberstein *et al.* 2009). The DEC component was to assess the ecological consequences of fire on the site and preliminary data was obtained for the first 7 months post-fire. This incorporated changes in vegetation and litter cover, and the responses of plant species such as proportions of resprouters and seeders, flowering and juvenile periods. A limitation of the study is that it was conducted at one site, however this was related to the cost of the work undertaken to measure recharge by CSIRO (Silberstein *et al.* 2009) which precluded assessing more sites. Future work (2009-10) will gather information on regeneration strategies, flowering age of species and the juvenile period of key fire species (e.g. species that have very long juvenile periods and species that require frequent fire). This work will be expanded to more sites with a range of different ages since burn and the juvenile period of species will be compared to that found in jarrah forest (Burrows *et al.* 2008). Calculations of the amount of fuel and fuel accumulation rates over time will also be undertaken (Burrows and McCaw 1989).

The recommendation from the Gnangara Sustainability Strategy (Government of Western Australia 2009a) to determine the optimum fire regime to maximise groundwater recharge is based on the PRAMS model with the base scenario of 10% of total native vegetation within the GSS study area (each year) on a 10-year rotation. Future work on fire in the GSS study area (until June 2010) aims to develop an appropriate fire regime and fire management on the Gnangara groundwater system that will incorporate diverse requirements, including the protection of biodiversity and management of fuel accumulation.

## Recommendations

In order to compile information on the past burning history in the GSS study area, it is recommended that:

- assessments of fire frequency area and distribution based on DEC fire records and analyses of Landsat imagery are completed.
- Assessment of the distribution and area of habitat (e.g. vegetation complex) in different years since last fire categories is completed.

In order to examine the potential effects of the GSS recommended 10-year rotational burning on the biodiversity values in the GSS study area, it is recommended that:

- the effect of 10-year rotational burning on seed storage in the soil and seed production is investigated.
- the potential effect of 10-year rotational burning on fire response of seeders and sprouters and the time to flowering period is investigated.
- the potential effect of 10-year rotational burning on fauna is investigated.

In order to develop ecologically appropriate fire regimes in the GSS study area, it is recommended that:

- ecological burning regimes based on vital attributes - frequency distribution are developed.
- the impact of fire intensity of biodiversity values in the GSS study area is investigated.
- ecological burning regimes based on habitat requirements for fauna are developed that ensure that there will be different fire ages over time in the long term, and retention of long-unburnt *Banksia* woodland – important to species such as honey possum, some reptiles and some birds, due to habitat features including litter, and food sources.
- work on floristics to determine the species with the longest time to flowering is completed.
- reviews of DEC fire management operations are completed, including: the fire planning framework; conditions such as weather that impact fire management, and the implications for fire management for maintaining wetlands, threatened species, reference areas and following the replacement of pine plantations with strategic ecological linkages and a parkland.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter Eight: Impacts of Introduced Species on Biodiversity

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Our environment, our future

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# 8. IMPACTS OF INTRODUCED SPECIES ON BIODIVERSITY

## Key points

- Introduced species are serious threats to native biodiversity and can substantially modify ecosystem processes and functions.
- Common traits of introduced plant species that enable them to become invasive (pest plants) include an ability to rapidly reproduce, rapid growth from seedling to maturity, phenotypic and environmental plasticity, and an ability to spread in a new environment (high dispersal rate).
- Some of the direct effects of pest plants on ecosystems are changes in structure, increased productivity and increased litter, different litter breakdown rates, and altered nutrient regimes, hydrological cycles and fire regimes. Indirect effects include deleterious associations with micro-organisms such as bacteria and mycorrhiza and flow-on effects for larger invertebrate and vertebrate animals.
- Of the 834 introduced plant taxa recorded on the Swan Coastal Plain, 564 have records also within the GSS study area. Introduced plant taxa comprise nearly 30% of the plant taxa records within the GSS study area.
- Thirty species of introduced plants have been identified as high priority for management due to their significant ecological impacts, invasiveness and potential or actual distribution as pest plants.
- Predation, habitat destruction and competition are three threatening processes by which introduced fauna species affect biodiversity values.
- In the GSS study area, predation by feral foxes and cats is recognised as the main threat to the declining mammal fauna. Habitat destruction and competition by pigs, rabbits, rainbow lorikeets, feral bees and three species of introduced fish are also having an impact on biodiversity values in the area.
- Fox baiting can work, and it does help native fauna recovery, but without integrated multi-species management, the removal of foxes can allow other introduced species such as feral cats to increase and take their place. This effect is known as ‘meso-predator release’.

- Introduced fauna species are regularly observed in the urban landscape and the remnant vegetation of the GSS study area, and fox activity appears to be higher than in other areas of the state, both baited and unbaited. However, no coordinated management of introduced species occurs in the GSS study area, other than management of predators of the critically endangered *Pseudemydura umbrina* (western swamp tortoise).
- As has been successfully demonstrated in areas such as Twin Swamps Nature Reserve, the removal of introduced fauna species from a core conservation area provides the opportunity for persisting native species to recover.

## Introduction

Introduced species severely threaten some native flora and fauna communities, either directly or by modifying ecosystem process and functions (Gordon 1998; Hulme 2006; Vitousek *et al.* 1997). The introduction of non-native plants or animals has usually been deliberately facilitated by humans, and although most introduced species do not deleteriously affect ecosystems, a small proportion become invasive (Hulme 2006). Traits that enable introduced species to persist or spread successfully include high fecundity, good defence mechanisms, high survival rates, adaptability and a lack of natural enemies such as predators and diseases (Cheal *et al.* 2006). Introduced species that become invasive can have catastrophic effects on native biodiversity assets and ecological processes by altering nutrient levels, hydrological cycles, fire regimes and community composition, including the removal of keystone species (Brooks *et al.* 2004; D'Antonio and Vitousek 1992; Le Maitre *et al.* 1996; Vitousek and Walker 1989; Yurkonis *et al.* 2005). For conservation strategies to be successful, it is essential that the introduced species, both plant and animal, that pose major threats, be identified and the mechanism through which they threaten biodiversity assets be understood.

Invasion of native ecosystems by introduced plant species, from here on also referred to as pest plants, is a serious environmental issue in Australia, and causes damage to natural landscapes, agricultural lands, waterways and coastal environments (NRMMC 2006). Most pest plants have been introduced intentionally for agricultural, pastoral or ornamental purposes (Humphries *et al.* 1991; Lonsdale 1994; Martin *et al.* 2006). Approximately 2700 introduced plant species are now naturalised in Australia (Groves *et al.* 2003), meaning they reproduce unaided by humans, and over 800 of these occur on the Swan Coastal Plain

(Keighery and Longman 2004). The spread and proliferation of pest plants is closely linked to human activities and disturbance, such as clearing, fires, road and track construction, grazing and dumping of garden refuse. This is most pronounced in urban areas where human activities are the main cause of disturbance leading to weed invasion.

Invasive or pest animals include introduced and native animals that have or may become overabundant and that pose threats to agriculture, the environment or human health and safety. At least 80 species of non-native vertebrates have established wild populations in Australia and more than 30 of these species have become pests (Bomford and Hart 2005). The impact of introduced animals on native ecosystems can be either direct or indirect. Predation of native mammals, birds and reptiles by feral foxes and cats is an example of a direct impact, whereas an indirect impact may be competition from feral bees that have invaded nesting hollows used by the endangered *Calyptorhynchus latirostris* (Carnaby's black-cockatoo), or the destruction of riparian vegetation that supports a population of the endangered *Pseudemydura umbrina* (western swamp tortoise) (Cheal *et al.* 2006).

The impact of introduced fauna species in the GSS study area has not been quantified but is presumably deleterious. In particular, predation of native mammals by foxes is considered to be the main factor currently contributing to the decline and local extinction of mammal species on the Swan Coastal Plain (Johnson and Isaac 2009; Kitchener *et al.* 1978; Reaveley 2009).

The aims of this chapter are to collate, review and summarise available information about the impacts of introduced species on biodiversity and ecological processes, to give some examples of introduced species present in the GSS study area, and discuss the current management of some of these species. In particular, we addressed:

- impacts of pest plants on biodiversity
- pest plants of concern on the Swan Coastal Plain and in the GSS study area
- impacts of introduced fauna on biodiversity, specifically focusing on the ecological impacts (e.g. predation) by examining focal species of concern in the GSS study area (e.g. foxes)
- management options for introduced species and implementation strategies within the GSS study area.

In addition, two DEC–GSS projects were undertaken to address gaps in our knowledge of introduced species in the GSS study area. These projects assessed the presence of foxes and their impact on persisting native fauna. The projects are outlined below.

1) **Feral sand pad survey**

   The objective of this project was to formulate an index of fox activity in Yanchep National Park by establishing a series of sand pads along vehicle tracks in the park and recording abundance of fox tracks during a one week period. This index could then be compared with similar studies at other sites in Western Australia to assess the level of fox activity in the GSS study area.

2) **Targeted quenda survey**

   A targeted survey was undertaken to locate populations of *Isoodon obesulus* (quenda) persisting within the GSS study area and the types of habitat in which they occur. The survey sites included a fenced reserve to assess the population effects of predator exclusion.

The major findings and results of these projects have been summarised in this review while full reports are provided in technical reviews (see Appendix 2).

## Pest plants

Pest plants are those that are unwanted because of their deleterious effects and that have become naturalised in a particular ecosystem. They fit the *Australian Weeds Strategy 2006* definition of a ‘weed’ as a plant that requires some form of action to reduce its harmful effects on the economy, the environment, human health and amenity. While some were accidentally introduced and have become naturalised, many have been deliberately introduced to Australia for crops, pastures or as ornamentals (Adair and Groves 1998). Although the majority of pest plants are exotic species, a small but increasing number are Australian natives that have spread in response to changed environmental conditions (Hussey et al. 2007; Keighery 2002).

# Characteristics of pest plants

There are several traits that predispose a plant to become invasive (Baker 1974; Sakai et al. 2001). These include an ability to rapidly reproduce, rapid growth from seedling to maturity, and both phenotypic and environmental plasticity (Baker 1974; Leishman et al. 2007; Sakai et al. 2001). In addition, an ability to spread in a new environment (high dispersal) is also a common trait (Sakai et al. 2001). As an example, the Geraldton carnation weed (Euphorbia terracina) displays a number of these characteristics and represents a high risk to the biodiversity values of the Swan Coastal Plain (Keighery and Keighery 2007). It is capable of phenotypic and morphological plasticity in response to varying environmental conditions, a trait which enables non-native species to successfully invade a range of habitats (Riordan et al. 2008).

# Impacts of pest plants on biodiversity

The deleterious effects of pest plants on the ecosystems they invade include changes in species richness, abundance or ecosystem function (Grice 2004; Vitousek et al. 1997). These impacts can be direct or indirect. Direct impacts include changes in structure, increased productivity and litter, different litter breakdown rates and altered nutrient regimes, hydrological cycles and fire regimes (Brooks et al. 2004; D'Antonio and Vitousek 1992; Le Maitre et al. 1996; Vitousek and Walker 1989; Yurkonis et al. 2005). Indirect impacts include detrimental associations with micro-organisms such as bacteria and mycorrhizae and flow-on effects for larger invertebrate and vertebrate fauna (Zedler and Kercher 2004). In the following section, some of the impacts of pest plants on biodiversity and ecosystem functions, including changes in species richness, nutrient cycling, hydrology and altered fire regimes are briefly described.

## *Native species richness and community composition*

Pest plants have reduced the species richness, diversity and composition in a range of communities (Hejda *et al.* 2009) and have had a detrimental impact on community structure (D'Antonio and Meyerson 2002; Gordon 1998; Ogle *et al.* 2000; Parker *et al.* 1999; Yurkonis *et al.* 2005). However, few studies have quantified the impacts of invasive plants on biodiversity (Adair and Groves 1998), possibly because their community-level effects are often difficult to measure.

Introduced pest plants can reduce the diversity of native flora by displacement of native species and by inhibiting the establishment of new individuals (Yurkonis *et al.* 2005). Species displacement is the most widely accepted mechanism by which introduced pest species reduce native floristic diversity (Levine *et al.* 2003), and in regions where more than one pest species is abundant, native plant species richness is typically lower (Grice 2004). The subsequent changes in native species diversity are often associated with vegetative structural changes that may further affect native floristic diversity (Adair and Groves 1998).

There are numerous Australian examples of introduced pest plants having an impact on native floristic diversity (Grice 2004; Groves and Willis 1999; Keighery 2004). For example, introduced grasses, such as *Hyparrhenia hirta* (Tambookie grass), are of particular concern in Western Australia (Keighery 2006). Tambookie grass is one of many perennial grass weeds that invade native vegetation by forming dense tussocks capable of dominating and replacing (and reducing) native plant diversity. The flow-on effects of this species also contribute to altered fire regimes that subsequently further modify vegetation structure and composition (CRC Australian Weed Management 2007).

By altering plant diversity and habitat structure, the introduction of pest plants will have flow-on effects for faunal communities. The responses of fauna to pest plants are thought to occur as a consequence of habitat alteration and changes in trophic interactions (Sakai *et al.* 2001). Changes in habitat structure caused by introduced pest plant species can have cascading consequences that alter the resources available to fauna in the modified environment (Valentine *et al.* 2007). When this happens, the habitat created by the pest plant may not meet the requirements of native fauna, disadvantaging certain species. For example, the introduced *Cryptostegia grandiflora* (rubber vine) provides a worse environment for litter-dwelling lizards than the native species it has taken over because of the lower ambient temperatures, reduced availability of prey and a reduction in camouflage from predators (Valentine *et al.* 2007). As a result, fewer reptile species utilise habitat infested with it (Valentine 2006). In wetlands, fewer birds are associated with the exotic weed *Urochloa mutica* (para grass), originally introduced to increase pasture production (Ferdinands *et al.* 2005). The loss of biodiversity caused by invasive alien plants may have cascading trophic effects (Sakai *et al.* 2001) that alter fundamental ecosystem processes (Hulme 2006; Knops *et al.* 1999).

### *Nutrient cycling*

The impacts of plant invasions on soil nutrient cycling suggest that introduced plant species differ from native species in their biomass and productivity, tissue chemistry, plant morphology and phenology, and are able to alter soil nutrient dynamics (Ehrenfeld 2003). Such species have the potential to change many components of the carbon, nitrogen, water and other cycles of an ecosystem. Introduced pest plants often increase biomass and net primary production, increase nitrogen availability, alter nitrogen fixation rates, and produce litter with higher decomposition rates than co-occurring natives (Ehrenfeld 2003). This can have flow-on consequences for community structure and dynamics.

There are cases of introduced woody shrubs and grass species in Australia that are capable of nitrogen-fixing and altering nutrient regimes through different rates of leaf litter decomposition. *Cytisus scoparius* (scotch broom) in South Australia is able to change nutrient availability and is highly competitive in nutrient rich soils, both of which could be important mechanisms for this species' invasion in native woodlands (Fogarty and Facilli 1999). Other studies, in particular of grasses, have identified the pathways that non-nitrogen fixing species use to alter nutrient cycling (Mack and D'Antonio 2003; Rossiter *et al.* 2006). Despite considerable evidence that invasions can alter nutrient cycling, the effect of this upon nutrient availability for communities is not well demonstrated (Levine *et al.* 2003).

### *Hydrology and aquatic ecosystems*

Wetland habitats are vulnerable to introduced pest plant invasions, and invasive terrestrial and aquatic plants can affect the diversity of aquatic environments and hydrological cycles. Indeed, 24% of the world's most invasive plants are wetland species, even though < 6% of the Earth's land mass is wetland (Zedler and Kercher 2004). Pest plants that invade wetland habitats can change the rate or timing of evapotranspiration, run-off and access to water, as well as affecting the structure of the waterway, water chemistry and faunal communities (Levine *et al.* 2003). For example, in Australia, species of introduced willow (*Salix species*) can severely affect wetland systems. Willows spread their roots throughout the bed of a watercourse, resulting in reduced water flow and aeration. In addition, they form thickets that competitively displace native vegetation and divert water that causes flooding and erosion (NLWRA 2008). Furthermore, the leaves shed by willows create a flush of organic matter during autumn months, reducing water quality and available oxygen that subsequently affects native aquatic plants and animals (NLWRA 2008).

Aquatic weeds can block waterways and irrigation channels, forming a dense cover that restricts light and depletes oxygen, causing death of aquatic faun, including mass fish deaths (Waters and Rivers Commission 2000). Invasive wetland plants are generally assumed to reduce both plant and animal diversity, often forming monotypes or monocultures. These alter habitat structure, change nutrient cycling and productivity and modify food webs (Zedler and Kercher 2004). An example of a species in the GSS study area that is capable of altering the physical structure of a site, changing the hydrological conditions and subsequent affecting animal use is sharp rush (*Juncus acutus*). This rapidly spreading, highly invasive sedge is capable of restricting native animal access to waterways, harbouring pest animal species and restricting the flow of water, resulting in flooding (Department of Primary Industries 2008; Keighery and Keighery 2006; Parsons and Cuthbertson 2001).

### *Fire*

The interaction of pest plants with fire and their effect on fire regimes, particularly fire intensity and frequency, has been the subject of considerable study (Brooks *et al.* 2004; D'Antonio 2000; D'Antonio and Vitousek 1992). Pest plants affect fire regimes by invading an area and substantially modifying vegetation structure and composition, which can affect the intensity and/or frequency of a fire (Levine *et al.* 2003). For example, the grass–fire cycle occurs when an introduced grass species invades a shrubby habitat, alters the vegetation structure and creates a continuous fuel bed that can lead to an increase in fire frequency, and subsequently result in the conversion of shrublands to grasslands (D'Antonio and Vitousek 1992). In addition, introduced grass species may increase fuel loads and may contain more combustible elements than native species. These two factors subsequently alter fire intensity (Grice 2004; Levine *et al.* 2003).

Invasion of post-fire vegetation by herbaceous pest plants has been identified as a threat to the conservation of south-west Western Australian Proteaceae species (Lamont *et al.*

1995). Intense fire can open areas of vegetation and create a rich ash bed, allowing invasive pest plants with competitive advantages to rapidly establish with, or instead of, native vegetation. Fragmented and remnant areas of native vegetation are particularly susceptible to pest plant invasion following fire, often leading to a loss in native vegetation. Milburg and Lamont (1995) documented the invasion of remnant sclerophyll woodland vegetation by exotic species after fire and found that the number of pest plant species, as well as their frequency and cover, increased after fire, whilst the abundance of native species decreased. The most abundant pest plant species are perennial grasses *Eragrostis curvula* and *Ehrharta calycina*. It has been suggested that their abundance increases the susceptibility to fire of the vegetation community, since grasses are normally an insignificant component of sclerophyll vegetation (Milberg and Lamont 1995).

## Introduced pest plants in the GSS study area

Naturalised plants in Western Australia have been recorded in various checklists (Hussey *et al.* 2007; Keighery and Longman 2004; Keighery 1995). A review of pest plants by the Perth natural resource management region (NRM) management group in 2007 listed 834 naturalised plant taxa or environmental weeds on the Swan Coastal Plain (SWA2) IBRA Region, with 119 families represented (Keighery and Bettink 2008). Eighteen taxa are Western Australian native species that have an uncertain classification (native or pest species). There are 45 true aquatic pest taxa, 21 aquatic and dampland taxa, 560 purely terrestrial taxa, and 229 terrestrial and dampland taxa. There is a significant number of herbs (469), while grasses (Poaceae) represent one of the largest families of pest plant species present in Western Australia (130 taxa). Other introduced species include 93 geophytes, 130 shrubs or trees and 10 sedges (Keighery and Bettink 2008). Of the ~ 1900 plant taxa recorded in the GSS study area, nearly 30% are classified as introduced. Herbarium records show that 564 introduced plant taxa, comprising 316 genera, have been recorded in the GSS study area (Chapter 2).

Of the 834 taxa recorded, several appear on national environmental ‘weed’ lists and state declarations. There are three ‘national alert list’ species, *Dittrichia viscosa*, *Lachenalia reflexa* and *Retama raetam*, 12 species that are ‘weeds of national significance’ (WONS) and nine state declared species (Hussey *et al.* 2007; Keighery and Bettink 2008; Thorp and Lynch 2000).

## *Species of concern in the GSS study area*

Of the environmental weeds recorded by the Perth region NRM management group , 30 species have been identified as high priorities for management (Table 8.1; Keighery and Bettink 2008), all of which occur in the GSS study area. This group comprises 25 terrestrial and 5 aquatic species. These have been prioritised for their invasiveness, actual and potential distribution, trends, classification or rating and ecological impacts. All of the species satisfy one or more ecological impact attribute criteria, based on Platt *et al.* (2005). These criteria range from altered fire regimes, altered nutrient conditions and altered hydrological patterns, to loss of biodiversity and allelopathic effects.

Current national classification or listing, including current listing as a WONS species or WONS ranking (Thorp and Lynch 2000), species identified nationally as a potential or established environmental weed (Csurches and Edwards 1998) and state declared plants, listed under the *Agriculture and related resources protection Act*, *1976* (DAFWA 2007) were also taken into account.

Table 8.1: Weed species that have been identified as high priorities for management and that occur in the GSS study area (Keighery and Bettink 2008).

| **Scientific name** | **Common name** | **Appears on other lists**[1] |
|---|---|---|
| *Schinus terebinthifolius* | Brazilian pepper | WONS nominated |
| *Hydrocotyle ranunculoides* | robust pennywort | WONS nominated, P1,P2 (whole state) |
| *Colocasia esculenta* | taro | |
| *Zantedeschia aethiopica* | arum lily | WONS nominated, P1, P4 (whole state) |
| *Asparagus asparagoides* | bridal creeper | WONS, P1 (whole state) |
| *Carduus pycnocephalus* | slender thistle | |
| *Chrysanthemoides monilifera* subsp. *monilifera* | boneseed | WONS, P1 (whole state) |
| *Isolepis hystrix* | club rush | |
| *Euphorbia terracina* | Geraldton carnation weed | |
| *Myriophyllum aquaticum* | parrots feather | WONS, P1,P2 (whole state) |
| *Lachenalia reflexa* | yellow soldier | NAL |
| *Ferraria crispa* | black flag | |
| *Freesia alba* x *leichtlinii* | freesia | |
| *Gladiolus undulatus* | wavy gladiolus | |
| *Moraea flaccida* | one-leaf cape tulip | P1 (whole state), P4 (some shires) |
| *Sparaxis bulbifera* | sparaxis | |
| *Watsonia meriana* var. *bulbillifera* | bulbil Watsonia | WONS nominated |
| *Juncus acutus* subsp. *acutus* | sharp rush | |
| *Acacia* spp. (incl. *longifolia pycnantha, decurrens,* | | |

| | | |
|---|---|---|
| *dealbata*) | | |
| *Leptospermum laevigatum* | Victorian teatree | |
| *Retama raetam* | white weeping broom | NAL |
| *Cenchrus ciliaris* | buffel grass | |
| *Cortaderia selloana* | pampas grass | |
| *Cynodon dactylon* | couch | |
| *Ehrharta calycina* | perennial veldtgrass | |
| *Hyparrhenia hirta* | tambookie grass | |
| *Tribolium uniolae* | haas grass, tribolium | |
| *Rubus* spp. | blackberry | WONS, P1, P2 (whole state) |
| *Salvinia molesta* | salvinia | WONS, P1, P2 |
| *Typha orientalis* | typha, bulrush | |

[1] WONS – 'weed of national significance' list, NAL – national alert list

## High priority species – case studies

### *Hyparrhenia hirta* (tambookie grass)

Tambookie grass is among a suite of invasive perennial grasses that have major ecological impacts and pose significant threats to the biodiversity of natural ecosystems.
Deep rooted and drought resistant, it reproduces by seed and is commonly spread by soil movement, mowing and water (CRC Australian Weed Management 2007). Tambookie grass rapidly regrows after it is burnt, grazed or slashed, with the highest regrowth rates in warm to hot conditions. Australian studies have shown it has rapid growth rates, wide temperature ranges for germination and growth, and the ability to grow and become invasive on a range of soil types (Chejara *et al.* 2008; Chejara *et al.* 2006; McWilliam *et al.* 1970).

Capable of spreading rapidly, it now infests large areas of northern New South Wales, southern Queensland, Victoria and South Australia and is becoming more widespread in Western Australia. A large proportion of populations occur around Perth, where plants can be seen infesting road verges and disturbed sites as well as intact shrubland and woodlands (Western Australian Herbarium 2008). Of the environmental weeds recorded by the Perth region NRM management group, it is classed as a high priority environmental weed because it is capable of having an impact on remnant bushland, including threatened ecological communities on the eastern side of the Swan Coastal Plain (Keighery and Bettink 2008).

Tambookie grass forms densely tufted tussocks that can dominate and reduce native plant diversity, and affect native fauna. In a study conducted in New South Wales on endangered Eucalypt woodland, tambookie grass greatly reduced the cover and species richness of native flora, in some cases by over 50% (Chejara *et al.* 2006).

Similar to *Ehrharta* spp. (veldt) and *Eragrostis curvula* (African love grass), tambookie grass can change either fire intensity and/or frequency, which ultimately alters vegetation structure and favours reinfestation. It has also been suggested that *Hyparrhenia* species may produce allelochemicals that affect the growth of surrounding species. However, this is not well understood. The greater production of plant litter and closer inter-tussock spacing of tambookie grass appears to favour the formation of monocultures compared with some native grasses, which form more open clumps (CRC Australian Weed Management 2007).

### ***Lachenalia reflexa*** **(yellow soldier)**

Yellow soldier is an invasive bulbous species currently listed on the national alert list because of the impact it has on biodiversity and its potential for rapid spread. Introduced for its ornamental value, it is now considered the most problematic and invasive of the *Lachenalia* species known in Australia (CRC Australian Weed Management 2003). Western Australia currently has the only known occurrences of the weed in Australia and all of these populations are restricted to the Perth metropolitan area, which is included in the GSS study area. A total of 31 populations have been located. All are found on the Swan Coastal Plain. The populations extend to 35 km north of the centre of Perth (DEC 2008).

*Lachenalia reflexa* has the potential to spread throughout the south-west corner of the state, and increase in abundance within its current range. This species invades both intact and disturbed bushland, and is having a severe impact on biodiversity. Once established in medium to high densities, up to 2000 plants/$m^2$, it displaces native herb and annual plant species, causing loss of plant diversity. Most commonly, yellow soldier is found on the Quindalup and Spearwood dune associations, characterised by grey, white or yellow calcareous sands. Almost all known populations occur in bushland reserves, such as Craigie Open Space, Shepherds Bush Reserve and Woodvale Nature Reserve in the GSS study area, many of which have high conservation value.

The seed is not naturally easily spread over long distances, and human activity appears to be the main cause of medium to long distance dispersal. Strategic management of this species by DEC has focused on understanding the distribution and threats and then eradicating outlying populations at high conservation value sites (DEC 2008).

## DEC management of introduced plant species

DEC is responsible for administering several pieces of legislation and has among its objectives to ‘protect, conserve and, where necessary and possible, restore Western Australian biodiversity values. Weed invasion is one of the most significant threatening processes to biodiversity values and DEC’s primary focus is to minimise the impacts of weeds on these values. It is responsible for managing weeds in a large number of sites in the GSS study area, including two regional parks, Yanchep National Park, nature reserves and other reserves in the Swan Coastal District. DEC may also undertake weed management works as part of the recovery of threatened species and communities.

Although weeds are a major threat to biodiversity, there is only limited recurrent funding available for weed management on the DEC-managed estate. A significant portion of funding is sought from external grants or through components of other projects, including threatened species recovery. Community involvement often plays a central role in applying for these grants. The department’s Saving our Species Biodiversity Conservation Initiative of 2006–09 provided scope for addressing threatening processes, including weeds, throughout the state. The eradication, control or substantial reduction of priority environmental weed populations was a major commitment of the initiative. There were minor components of several projects undertaken as part of the initiative, in the GSS study area, including work on *Lachenalia reflexa* (yellow soldier).

The *Environmental Weed Strategy for Western Australia* (1999) largely guides the department’s weed management. Weeds were classified according to their invasiveness, distribution and environmental impacts. Generally, high priority species, some of which are also declared plants, are targeted for priority control operations. The majority of weed control on DEC-managed lands is prioritised within each DEC region, in line with the *Environmental Weeds Strategy for Western Australia* (1999). In accordance with this strategy, expenditure and resources are prioritised to those weeds and infestations that can return the best value for biodiversity conservation (DEC 2007).

DEC has annual programs aimed at controlling priority weeds in the region, including typha, bridal creeper, pampas grass, arum lily and cotton bush. Each of DEC's regions has been encouraged to develop a strategic weed plan, including a systematic review of the weeds occurring on DEC-managed land in each region. Few regions within DEC have undertaken this process. However, an assessment and prioritisation of environmental weeds in the Perth NRM region (formerly Swan NRM Region), completed in 2008, (Keighery and Bettink 2008) can provide a tool in the planning process for DEC managers in the GSS study area. This could provide a species management approach and could be combined with an asset based approach to prioritise sites to become the focus of resources for weed management. Some of the existing area based management plans and recovery plans for threatened species and communities may also contain more detailed and specific guidance for prioritising control efforts.

## Weed management in the GSS study area

DEC's Swan Coastal District is responsible for managing reserves in the Swan Region as either the sole management agency or in joint management with other government agencies. Priorities for weed management in particular reserves are outlined in annual works programs. These programs are developed with only limited budgets, but are augmented with funding from other projects and internal and external grants. Although weed priorities vary across reserves, declared plants are a prime focus. Other high impact species such as bridal creeper (*A. asparagoides*) and Watsonia (*Watsonia* spp.) are targeted at specific sites.

Yanchep National Park, of 2799 ha, contains significant natural areas including wetlands, but also has significant weed legacies from past land use and management practices. A management plan (CALM 1989) outlines strategies to protect these natural areas from weed infestation. This plan outlines prioritised actions in control programs and targets declared species, annual grasses in recreation areas (to reduce fire hazards) and control, as resources permit, of non-declared species which have been determined to have adverse ecological impacts. Important environmental weeds present include *Moraea flaccida* (Cape tulip) and *Solanum sodomeum* (apple of Sodom), *Ferraria crispa* (black flag) and a number of introduced grasses. *Pennisetum clandestinum* (kikyu) has spread from developed areas into wetland vegetation. The continued spread of *Typha orientalis* (typha) is identified as a priority as it was deemed to be changing the range of habitats in at least one of the wetlands (CALM 1989).

### *Regional parks*

The two regional parks in the GSS study area, Herdsman Lake and Yellagonga, represent the majority of land reserved for parks and recreation in the Perth metropolitan area. There are a number of different agencies involved in the management of the parks. DEC manages Herdsman Lake Regional Park in collaboration with the City of Stirling and the Water Corporation, and it  manages the Yellagonga Regional Park in collaboration with the City of Joondalup and the City of Wanneroo. ‘Friends groups’ also contribute to the implementation of on-ground works. Weeds have been identified in management plans as a major problem requiring action by DEC. DEC’s objective is to minimise the impact of environmental weeds on local plant species and communities, with performance measures based on changes in abundance and distribution of priority species (CALM 1989; 2003).

DEC is responsible for meeting this objective by implementing weed control and revegetation plans, which outline priorities for weed control and prescribe that local species be used for plantings, and if non-local species are required, that they be non-invasive. Other related measures include the management of fire and restrictions on the importation of soils that may be a vector of weed spread. The continued invasion of *Typha orientalis* (typha) is seen as a major threat to the wetland ecological systems of both parks, as well as constituting a major fire threat in late summer and early autumn. *Cortaderia selloana* (pampas grass) is also problematic in each park, while *Zantedeschia aethiopica* (arum lily), *Asparagus asparagoides* (bridal creeper) and *Arundo donax* (giant reed) are significant problems in Yellagonga Regional Park.

## Introduced fauna

A pest animal species is typically defined as one which has a deleterious impact on a valued resource, whether that resource is economic or environmental (Cheal *et al.* 2006). Most animal pests in Australia were introduced after European settlement and were introduced as food, as companions, for sport, or to control other pest species. However, as is the case with pest plants, pest animals can include Australian native species that have extended their biological range. Common traits of invasive fauna species include high fecundity, good defence mechanisms, high survival rates, phenotypic plasticity that enables them to adapt to a range of environmental conditions, and a lack of natural enemies such as predators and diseases (Cheal *et al.* 2006; Saunders 2008).

Despite historical efforts to combat invasive pest animals, often with the goal of eradication, many continue to survive and cause significant, ongoing damage to conservation values (Saunders 2008). Introduced species have been a major contributor to global mammal extinctions in the past 200 years, nearly half of which have occurred in Australia (Short and Smith 1994), and introduced species have been implicated in the drastic decline of Australia's native freshwater fish populations and species (Sommer *et al.* 2008). In addition, introduced species have been implicated in the degradation of aquatic ecosystems (Sommer *et al.* 2008), and are considered to threaten the value of 14 of Australia's 15 natural world heritage areas, and 13 of the 15 'biodiversity hotspots' in Australia (Saunders 2008).

Within the GSS study area, there are at least 33 species of introduced animal taxa (Table 8.2), most of which are birds, including several Australian native species that have expanded their range, and mammals. It should be noted that introduced species of Insecta are probably the most prolific group of species but their impact is more likely to be an agricultural rather than a biodiversity threat. It is also difficult to obtain a complete list of introduced Insecta. Of the species listed in Table 8.2, at least 26 are thought to have impacts on native biodiversity, either directly or indirectly. In addition, at least 16 introduced species have been identified at risk of expanding their distribution into the GSS study area. These are listed in Table 8.3, along with an indication of whether they are thought to be threats to biodiversity.

Table 8.2: Introduced animals present in the GSS study area and whether they are perceived as a biodiversity threat (DEC 2009; Government of Western Australia 2007; ISSG 2009)

| Taxonomic class | Scientific name | Common name | Perceived biodiversity threat? |
|---|---|---|---|
| **Aves** | *Cacatua galerita subsp. galerita* | sulfur-crested cockatoo | Y |
| | *Cacatua sanguinea* | little corella | Y |
| | *Cacatua tenuirostris* | eastern long-billed corella | Y |
| | *Cairina moschata* | muscovy duck | ? |
| | *Carduelis carduelis* | European goldfinch | ? |
| | *Columba livia* | domestic pigeon | Y |
| | *Dacelo novaeguineae* | kookaburra | Y |
| | *Glossopsitta concinna* | musk lorikeet | ? |
| | *Passer domesticus* | house sparrow | Y |
| | *Passer montanus* | Eurasian tree sparrow | ? |
| | *Streptopelia chinensis* | spotted turtle dove | ? |
| | *Streptopelia senegalensis* | laughing turtle dove | ? |
| | *Trichoglossus haematodus* | rainbow lorikeets | Y |
| **Mammalia** | *Bos taurus* | European cattle | Y |
| | *Capra hircus* | goat | Y |
| | *Equus caballus* | horse | Y |
| | *Felis catus* | feral cat | Y |
| | *Mus musculus* | house mouse | Y |
| | *Mustela putorius* | European polecat (ferret) | Y |
| | *Oryctolagus cuniculus* | European wild rabbit | Y |
| | *Rattus norvegicus* | brown rat | Y |
| | *Rattus rattus* | black rat | Y |
| | *Sus scrofa* | feral pig | Y |
| | *Vulpes vulpes* | red fox | Y |
| **Osteichthyes** | *Carassius auratus* | goldfish | Y |
| | *Cyprinus carpio* | carp | Y |
| | *Gambusia holbrooki* | gambusia (eastern mosquito fish) | Y |
| | *Geophagus* sp. (*Geophagus brasiliensis*?) | pearl cichlid | Y |
| **Insecta** | *Apis mellifera* | feral honey bee | Y |
| | *Iridomyrmex humilis* | Argentine ant | Y |
| | *Pheidole megacephala* | coastal brown ant (big-headed ant) | Y |
| | *Vespula germanica* | European wasp | Y |
| | *Hylotrupes bajulus* | European house borer | N |

Table 8.3: Introduced animals that have a potential distribution or future distribution in the GSS study area and whether they are perceived as a biodiversity threat (DEC 2009; Government of Western Australia 2007; ISSG 2009)

| Taxonomic class | Scientific name | Common name | Perceived biodiversity threat? |
|---|---|---|---|
| **Amphibia** | *Bufo marinus* | cane toad | Y |
| **Aves** | *Acridotheres tristis* | common myna | Y |
| | *Anas platyrhynchus* | mallard duck | Y |
| | *Psittacula krameri* | Indian ringneck parrot | Y |
| | *Pycnonotus jocosus subsp. jocosus* | red-whiskered bulbul | Y |
| **Mammalia** | *Funambulus pennanti* | Indian palm squirrel | N |
| **Osteichthyes** | *Oncorhynchus mykiss* | rainbow trout | Y |
| | *Perca fluviatilis* | redfin perch | Y |
| | *Phalloceros caudimaculatus* | one spot livebearer fish | Y |
| | *Salmo trutta* | brown trout | Y |
| **Reptilia** | *Hemidactylus frenatus* | Asian house gecko | Y |
| | *Ramphotyphlops braminus* | common blind snake | ? |
| | *Trachemys scripta elegans* | red eared slider turtle | Y |
| **Insecta** | *Ommatoiulus moreleti* | Portugese millipede | Y |
| | *Solenopsis invicta* | red imported fire ants | Y |
| **Crustacea** | *Cherax destructor* | yabbie | Y |
| **Gastropoda** | | snail | Y |

## Impacts of introduced fauna on biodiversity

Although the impacts of introduced fauna are have not been quantified in the GSS study area, we use examples of species that occur within the GSS study area region to illustrate potential impacts in native biodiversity and ecosystem processes.

Three mechanisms by which introduced species affect biodiversity values are predation, habitat destruction and competition. These are described below, using introduced species common to the GSS study area to illustrate how they operate. Predation by foxes and cats is particularly emphasised as this is considered to be the most significant impact introduced species are having in the GSS study area (Reaveley 2009).

## *Predation*

Introduced predators affect species through direct predation, which can keep prey in a 'predator pit' of low abundance (Pech *et al.* 1992), in which either the predation alone may cause extinction (over-harvesting), or other causes and interactions exacerbate the predation effect. Direct predation may also lead to changes in the habitat use of prey species, so that species become confined to refugia where the availability of dense vegetation and food provide some degree of protection and resilience (Kinnear *et al.* 1988). These areas are not necessarily typical of a species' habitat requirements but provide protection from predators. For example, in the GSS study area *Isoodon obesulus* (quenda) is restricted to dense wetland-associated vegetation, although it occupies upland habitat in areas where predators have been suppressed (Bamford and Bamford 1994; Valentine *et al.* 2009).

Predation upon native wildlife species by *Vulpes vulpes* (red fox) and *Felis catus* (feral cat) pose significant threats for Australian vertebrate taxa and communities (Cheal *et al.* 2006; Dickman 1996; Kinnear *et al.* 1988; Kinnear *et al.* 1998; Paton 1993). These two top order predators have been recognised by both the state and federal governments, which have listed their impacts as potentially threatening processes (e.g. the Victorian *Flora and Fauna Guarantee Act 1988* and the federal *Environment Protection and Biodiversity Conservation Act 1999*.

### *Vulpes vulpes* (European red fox)

The red fox was successfully introduced into Australia for sport in the mid 1870s and within 60 years was common throughout most of the country, except the tropical north and some offshore islands (Reddiex and Forsyth 2004). Foxes occupy nearly all vestiges, including urban, alpine and arid areas, but are most common in woodland and semi-open habitats (Saunders *et al.* 1995). Foxes prey upon a wide variety of animals including small to medium-sized mammals, as well as birds and insects, and are regarded as a major factor in the decline of many ground nesting birds, small to medium-sized mammals, and reptiles. As native fauna did not co-evolve with the fox, susceptible prey species have few natural adaptations and strategies to avoid predation.

Foxes are a known or perceived threat for 51 species listed under the EPBC Act: 31 mammals, 8 birds, 3 amphibians, 7 reptiles, 1 invertebrate and 1 plant species (Reddiex and Forsyth 2004). Marsupials between 100 g and 5 kg (critical weight range according to Johnson & Isaac 2009) are considered most at risk from fox predation, particularly if they are ground dwelling, non-arboreal species in low rainfall areas (Burbidge and McKenzie 1989; Johnson and Isaac 2009; McKenzie *et al.* 2007). Not only do arid zone species have increased exposure to predation due to the open habitat, but they are also under increased pressure from land use change, introduced herbivores and fire impacts (Johnson and Isaac 2009; McKenzie *et al.* 2007). Species below the critical weight range have persisted more successfully, as their higher population growth rates allow them to recover more quickly from declines due to predation (Johnson and Isaac 2009). Species larger than 5 kg are a less preferred prey size for foxes (and feral cats), and their herbivorous diet has enable them to benefit from some of the land use changes (e.g. agriculture) in Australia (Johnson and Isaac 2009).

During the Western Australian Museum's comprehensive inventory of fauna that inhabited the northern Swan Coastal Plain in 1977–78, Kitchener *et al.* (1978) recorded the presence of foxes on 14 occasions, widely distributed across their survey sites on the northern plain. Similarly, the 2007–08 GSS fauna survey often recorded foxes on cleared tracks (Valentine *et al.* 2009). Foxes are regularly observed by DEC Swan Coastal District staff during daily field operations, and following a large fire in Yanchep National Park in January 2009, there were frequent and widespread sightings of foxes in the burnt landscape during the day. Jackson (2003) also reported fox populations at Whiteman Park, Kings Park and Bold Park. Therefore, it is likely that foxes have a similar impact on native wildlife in the GSS study area to other areas in Western Australia and in Australia in general.

### ***Felis catus* (feral cat)**

Cats became established in Australia soon after European settlement, arriving in Western Australia in 1829 as pets and to provide early settlers with a means of rodent control. Feral cats were widespread throughout the entire south-west region by 1890 (Abbott 2002). Feral cats now occupy most of the country other than dense rainforest. They are in Tasmania and many offshore islands (Reddiex and Forsyth 2004). Rabbits have aided the spread of feral cats, as they are a prey item and as they provide burrows for shelter. Feral cats are carnivores, solitary and nocturnal, with males occupying a home range of 10 $km^2$ or more, depending on the availability of food (DEH 2004b; DEWR 2007b). They can survive on only the moisture they get from their prey, which has facilitated their penetration of the arid zone.

Feral cats are opportunistic predators, relying on small to medium-sized prey including mammals, reptiles and birds (Dickman 1996; Paton 1993). On the mainland, the species most at risk from feral cat predation are mammals weighing less than 220 g (Moseby *et al.* 2009) and birds weighing less than 200 g. On islands, however, mammals and birds up to 3500 g can be taken (Dickman 1996). According to Dickman (1996), the susceptibility to feral cat predation is increased if protective cover is not readily available, and if foraging, burrowing or nesting activities of native species take place on the ground surface or other exposed sites. Another threat posed by feral cats is disease. Feral cats carry infectious diseases such as toxoplasmosis and sarcosporidiosis, which can be transmitted to native animals, domestic livestock and humans (DEH 2004b; Dickman 1996).

While the overall level of impact from cat predation remains uncertain (DEWR 2007b), there is clear evidence of significant impact on island fauna. On the mainland, cats have probably contributed to extinctions and have probably compromised recovery programs for threatened species (DEH 2004b), such as the reintroduction of *Macrotis lagotis* (bilbies) to Lorna Glen (Brent Johnson pers com. 2008). However, despite numerous records of feral cats preying upon native birds and mammals, Abbott (2008) is dubious of the extent to which cat predation has responsible for mammal decline.

## *Habitat destruction*

One of the factors leading to the decline and/or extinction of mammals in Australian habitat is degradation caused by introduced herbivores. Feral herbivores can degrade vast tracts of habitat, promote invasion by serious weeds, and pose an ongoing threat to rare plants and animals. Despite sustained effort to manage feral herbivores, at least 16 species are expanding their distribution within the rangelands of Australia and another six are newly established (Norris and Low 2007). Within the GSS study area, four introduced species including wild rabbits, feral pigs, European carp and the pearl cichlid, cause habitat degradation and have an impact on native fauna populations.

### ***Oryctolagus cuniculus*** **(European wild rabbit)**

Wild rabbits were introduced to Victoria and South Australia in the 1850s and are now widespread across Australia, other than in the far north (Williams *et al.* 1995). Rabbits flourish in many different habitats, ranging from deserts to coastal plains, and although in arid areas they need access to water, elsewhere rabbits obtain enough moisture from their food. Their fecundity is the key to their success. Rabbits are one of Australia's most destructive feral pests (Norris and Low 2007), and are responsible for large losses of native vegetation and subsequent erosion. Rabbits destroy threatened plant species, consume protective vegetation, and compete with native mammals for food and burrows. They are also a prey item for both native predators such as wedge-tailed eagles, and introduced predators such as the fox and feral cat (DEH 2004b; DEWR 2007a). Rabbits ringbark trees and shrubs, and prevent regeneration of plants by eating seeds and seedlings. Over substantial areas of Australia, rabbits have transformed the vegetation by consistently removing mulga seedlings (Norris and Low 2007). The impact of rabbits often increases during drought when food is scarce and they consume whatever they can (Williams *et al.* 1995). The destructive habits of feral rabbits have indirectly contributed to the decline of many native plants and animals (DEH 2004b; DEWR 2007a).

### ***Sus scrofa*** **(feral pig)**

Domestic pigs were brought to Western Australia by European settlers as a food source. Some escaped and became feral in Australia, founding the population of tens of millions that are now spread across the continent (DEH 2004b). Habitat degradation by pigs can be locally extensive and pig diggings are often associated with sites modified by people, or close to roads, tracks and watercourses (Choquenot *et al.* 1996). Extensive disturbance in moist gullies or watercourses by pigs may cause erosion and affect soil nutrient cycling and litter composition – directly from the mixing and aeration effects and indirectly from predation on earthworms (Choquenot *et al.* 1996). The impact of pigs on watercourses has destroyed breeding sites and degraded important habitats of frogs (DEH 2005).

Feral pigs may also spread environmental weeds and the plant pathogen *Phytophthora cinnamomi* through either ingestion and defecation of plant material or by transporting introduced plant seeds or dieback on their hooves or other body parts (Choquenot *et al.* 1996).

Feral pigs are widespread in south-west Western Australia, including large populations in the jarrah forest in the Darling Range (Reaveley A. pers. com. 2007). Within the GSS study area, feral pigs have been observed in Yeal Nature Reserve and Moore River National Park and signs of their diggings are widespread (Valentine *et al.* 2009).

### *Cyprinus carpio* (common carp/European carp)

The following information about introduced fish species is adapted from Sommer et al. (2008). Further information regarding the impact of introduced fish species on wetlands can also be found in Chapters 4 and 5.

A study conducted by Sommer *et al.* (2008) on wetlands in the GSS study area identified a number of introduced fish species, including *Cyprinus carpio* (common carp), that threaten the integrity of wetland habitats. *Cyprinus carpio* is a subtropical freshwater fish species tolerant of a range of environmental conditions, and an opportunistic omnivore (Sommer *et al.* 2008). Endemic to Europe, Russia, China, India and south-east Asia, *C. carpio* is now distributed across the globe (Kottelat 1997), and in the GSS study area has been found at Loch McNess, Lake Goolellal, Herdsman Lake (industrial) and Floreat Waters, Emu Lake and Bennett Brook (Sommer *et al.* 2008). The feeding behaviour of this species, which involves grubbing for food in benthic sediments (Allen *et al.* 2002) increases turbidity and releases nutrients into the water column (Kottelat 1997). Consequences of this activity include unfavourable changes in vegetation cover and food availability for native species, and increased algal blooms resulting in significant environmental damage to wetland systems (Kottelat 1997). For more information see Sommer *et al*. (2008).

### *Geophagus brasiliensis* (pearl cichlid, pearl earth-eater)

Another species identified by Sommer *et al.* (2008) as threatening GSS wetlands is the *Geophagus brasiliensis* (pearl cichlid). This species is an aggressive, territorial freshwater cichlid that can tolerate brackish conditions and is a generalist omnivore (Lowe-McConnell 1969; Mazzoni and Iglesias-Rios 2002). Endemic to South America, *G. brasiliensis* was introduced to Australia as an aquarium and food species in 1989 (McKay 1989). This cichlid is expected to cause significant damage to wetlands habitat, potentially resulting in the loss of native fish species (Department of Fisheries 2006).

In the GSS study area *G. brasiliensis* were first detected in Bennett Brook in April 2006. Despite a dramatic, multi-agency response aimed at controlling the spread of this species (Department of Fisheries 2006), a recent survey (Bray & Astbury 2008 as cited in Sommer *et al.* 2008) suggests it is unlikely to prevent the downstream spread of this species. Populations of native fish, including *G. occidentalis*, *B. porosa* and *E. vittata* at Bennett Brook, may be under considerable threat from *G. brasiliensis*, particularly as winter rains facilitate dispersal of this species (Sommer *et al.* 2008).

## *Competition*

Species in competition use one or more resources in common, and can suffer mutually depressed population sizes while seeking or using those resources (Dickman 2006). For species that are mobile, there is often competition for food, shelter or for habitats where these and other scarce resources can be found. Introduced species may out compete native fauna by either better exploiting resources required by native species for survival or by aggressively excluding native species from space and/or resources they would otherwise use (Stokes *et al.* 2009). Complete displacement of native species from preferred habitats may occur if there is high niche overlap (Stokes *et al.* 2009). Rainbow lorikeets, feral honeybees, mosquito fish, black rats and house mice are some of the introduced species recognised as competitors for resources with local native fauna in the GSS study area.

### ***Trichoglossus haematodus*** **(rainbow lorikeet)**

The rainbow lorikeet, a species native to eastern and northern Australia, was introduced to Perth during the 1960s. From fewer than 10 birds when it was first recorded, its numbers will have grown to an expected population of over 20 000 birds by 2010. The rainbow lorikeet is now a declared pest (Chapman and Massam 2007). The feeding and nesting requirements of rainbow lorikeets closely overlap with species such as the purple-crowned lorikeet, regent parrot, red-capped parrot, western rosella and Australian ringneck, which are native birds in Western Australia. As rainbow lorikeets are dominant and aggressive birds, they exclude native species from food and nest resources. In urban and suburban areas, such as those within the GSS study area, rainbow lorikeets frequently nest in bushland reserves where nesting hollows are scarce, causing competition for hollows. In addition, rainbow lorikeets displace *Eolophus roseicapilla* (galah), *Barnardius zonarius* (Australian ringnecks) and the endangered *Calyptorhynchus latirostris* (Carnaby's black-cockatoo) from potential nest sites and are known to kill and expel the nestlings of Australian ringnecks prior to taking over the nest (Lamont 1996). Furthermore rainbow lorikeets pose a disease risk to wild lorikeet and parrot populations as they are carriers of Psittacine beak and feather disease (DEH 2004a).

### *Apis mellifera* (feral honeybees)

Colonies of feral honeybees established soon after the introduction of honeybee hives in Western Australia in the 1830s. Their distribution and abundance has increased dramatically over the last 60 years and feral honeybees are now widespread except in arid inland areas, away from water (Paton 1996). Feral beehives are commonly observed in native trees in south-west Western Australia and may usurp nesting hollows used by possums, bats, some dasyurids, birds, and reptiles (Abbott I. pers. comm. 2009). However, according to Paton (1996) studies have shown that there are generally low rates of hollow occupancy by feral bees (typically < 1% of hollows and < 1% of trees). In addition, the natural hollows in most areas have appeared underutilised by native species (29% to 53% of hollows at two sites occupied by eight species of birds in spring over a number of years). This suggests that hollows were not in short supply within these breeding areas and that occupation of some hollows by feral bees would not have affected the birds (Paton 1996).

In contrast, Johnstone and Kirkby (2007) consider that feral bees are a major ecological problem as their invasion of hollows threatens the future conservation of many obligate hollow nesters. During their study of the breeding biology of four Western Australian black cockatoos, approximately 20% of breeding hollows studied were appropriated by feral honeybees. At one site, 50% of breeding hollows were invaded by the bees. Feral honeybees have also occupied hollows of the Australian ringneck, red-capped parrot, boobook owl and sacred kingfisher, and in some cases when birds were incubating eggs or brooding chicks (Johnstone and Kirkby 2007).

### *Gambusia holbrooki* (mosquito fish)

*Gambusia holbrooki* is a relatively small exotic freshwater species introduced from northern and central America to Australia in the 1920s and to Western Australia in 1934 (Allen *et al.* 2002). This species is a popular aquarium fish and was originally considered as a control agent for mosquito populations. Consequently, this species is now considered the most widely distributed freshwater fish species in the world (Service 1996). It can tolerate a wide range of conditions and is a generalist predator (Morgan *et al.* 1996; Pen and Potter 1991). *Gambusia holbrooki* displays aggressive behaviour, often causing caudal fin damage, which displaces native species (Morgan *et al.* 1996), particularly in areas with limited fringing vegetation. *G. holbrooki* is the dominant freshwater fish species on the Swan Coastal Plain (Pen and Potter 1991) and is a major threat to the wetlands of the GSS study area (Sommer *et al.* 2008).

### ***Rattus rattus* and *Mus domesticus/musculus***

Competition between introduced and native rodents may be dictated by dispersal advantage and therefore the first species to establish an area has the competitive edge (Stokes *et al.* 2009). The introduced *Rattus rattus* and the native *Rattus fuscipes* are both persisting in the GSS study area (Valentine *et al.* 2009). Stokes *et al*. (2009) found that maintenance of territorial space rather than resource competition was driving the competition between *Rattus rattus* and *Rattus fuscipes*. The superior colonisation ability of the invasive introduced rodent, their potential for rapid population growth and their broad ecological tolerances enabled them to be the first to colonise an area that had been subjected to wildfire. Their establishment subsequently inhibited recolonisation by *Rattus fuscipes*. However, in other areas that had not been subjected to a stochastic extinction event such as wildfire, neither *R. fuscipes* nor *R. rattus* were competitively dominant and instead partitioned macrohabitats so that both species could co-exist at high densities in their particular patch.

In the Arid Recovery Reserve Study (Moseby *et al.* 2009), *Mus domesticus* remained at similar levels both inside the fenced predator-free reserve and outside the fenced reserve. In contrast, there were significant increases in small native mammals within the fenced reserve over the six-year study, indicating that native rodents may be able to outcompete *M. domesticus.* This study strengthened previous suggestions that *M. domesticus* may not have a significant negative impact on local native mammal species (Moseby *et al.* 2009).

# Management of introduced fauna

On a national scale, the impacts of introduced species such as foxes, rabbits, pigs, cats, cane toads, and black rats are listed as key threatening processes under the Commonwealth's *Environment Protection and Biodiversity Conservation Act 1999*. A threat abatement plan for each of these species has been developed in consultation with the states and territories, for example the *Threat abatement plan for competition and land degradation by feral rabbits* (DEWHA 2008). This legislation has led to substantial control effort by organisations and agencies, with federal, state and local governments committing significant resources to managing these species.

However, there is little evidence that this management has led to a reduction in these threats and a reversal in the decline of native species (Reddiex and Forsyth 2004). There are few control programs that include monitoring their effect on the targeted introduced species. Non-treatment areas are also rarely used as a comparison, nor are the native species and habitats of interest assessed prior to the control program (Reddiex and Forsyth 2004). Factors affecting the success of a control program for introduced species include the intensity and frequency of the control program, abundance of the target following control, the size of the area being managed and the ability of the native species habitat to recover (Reddiex and Forsyth 2004).

Control of a particular feral species is most effective if it is coordinated with other threat abatement and recovery plans; for example, rabbit control should be undertaken in conjunction with control of foxes and feral cats and the implementation of recovery plans for threatened plants.

## DEC management of introduced fauna

### *Predator control measures*

#### Foxes and the Western Shield Program

In 1996 DEC commenced the broad scale fox control program 'called Western shield'. This program is now applied to nearly 3.5 million ha of land, extending from Esperance in the south-east to Karratha in the north, and occurring mostly on DEC-managed estate such as national parks, state forests and nature reserves (Orell 2004). The Western Shield

Program was initiated following the results of smaller fox control trials in the late 1980s that demonstrated a dramatic recovery of *Petrogale lateralis* (black-footed rock wallaby) populations in the presence of fox baiting (Kinnear *et al.* 1988; Kinnear *et al.* 1998). Subsequent fox baiting trials elsewhere displayed similar population increases of *Myrmecobius fasciatus* (numbats), *Bettongia penicillata* (woylies) and *Dasyurus geoffroii* (chuditch) (Thomson and Algar 2000). A particular focus of the Western Shield Program is to target the recovery of critical weight range mammals 35 g to 5.5 kg (CWR according to (CWR = 3.5g-5.5kg, Burbidge and McKenzie 1989), as this suite of species has exhibited the largest decline in range and abundance in the last 200 years, and this is partly attributed to fox predation (Possingham *et al.* 2004). The objective of the Western Shield Program in relation to foxes is to reduce their density, by 1080 baiting, to allow for the recovery of existing native animals, as well as enable the re-introduction, through translocation, of other species that have become locally extinct.

Success of the fox baiting program in Western Australia has been based on the tolerance of Western Australian mammals to monofluoroacetate (Mead *et al.* 1985), an active ingredient in the 1080 poison used in baits. Western Australian native animals have co-evolved with native peas (*Gastrolobium* spp.), which contain monofluoroacetate as a secondary compound to deter herbivory.

Of 40 fauna monitoring sites established throughout the baited areas of Western Australia, at least one of the mammal species present at 22 of the sites before broad scale baiting commenced has increased in abundance in the presence of fox baiting (Orell 2004). Similarly, Dexter and Murray (2009) demonstrated that fox control by poisoned baits in the East Gippsland region of Victoria resulted in higher abundance of three of the five medium-sized mammal species that were trapped in baited treatment compared to the numbers prior to fox control measures.

Foxes rapidly recolonise areas where control measures have produced substantial localised reductions in fox population density. This was demonstrated by Dexter and Murray's (2009) study in East Gippsland where fox abundance declined in both treatment blocks that were baited and in non-treatment unbaited blocks. This landscape scale decline suggested that as foxes were poisoned and died in the treatment area, the vacated territory was recolonised by foxes from the non-treatment areas, which would in turn become poisoned, subsequently drawing foxes in from even further away (Dexter and Murray 2009). The treatment sites acted as a sink, with the surrounding unbaited habitat acting as a source. From a management perspective, an unfenced core baiting area would not only have an effect in the baited area but would additionally exert a control effect beyond the core baiting area, potentially reducing fox abundance over a large area.

Baiting a buffer zone around the core area may provide additional reduction of fox populations (Thomson *et al.* 2000). This may be especially effective during the dispersal phase of foxes, with the buffer zone acting as a 'dispersal sink' and minimising fox migration into the core area (Thomson *et al.* 2000). This would be important if the core baiting area was specifically established to protect a particular species or group of species from predation. Due to economic and practical limitations regarding buffer size–core area ratio, the use of a buffer zone is probably most applicable in situations where the aim is to minimise reinvasion by foxes into very large areas (e.g. > 2000 $m^2$).

In addition to baiting, an effective management tool may be the use of islands and barriers, such as predator proof fencing, to exclude predators from certain areas. Other strategies to control foxes include trapping, shooting and den destruction, but all these methods are more labour intensive and have not had a significant or lasting impact on fox numbers (DEH 2004b).

### Control of feral cats

The management of feral cats is difficult on the mainland due to the lack of effective and humane broad scale control techniques, as well as the presence of domestic cats (DEH 2004b). Feral cats are not easily trapped, do not readily take baits, are difficult to locate and any control technique must safeguard against accidentally affecting domestic cats. Control of feral cats on islands is more feasible than on the mainland, with complete eradication of cats having been achieved in some instances. An example is Faure Island in Shark Bay, where complete eradication has enabled the reintroduction of a suite of threatened species without the risk of feral cat predation (Algar and Burrows 2004). For feral cat control to be successful on the mainland, replicating a pseudo-island approach is the most effective method. This approach identifies areas or species of high conservation value and protects it by surrounding it with exclusion fencing. Feral cats are eradicated within the enclosed area and a buffer zone surrounding the exclusion area is also maintained by trapping, shooting and tracking of individual cats; for example, cats on Peron Peninsula in Shark Bay have been effectively controlled in this way (DEH 2004b; DEWR 2007b). However, effective population control over large areas will only be possible with the development of a toxin or biological control (DEH 2004b).

One of the objectives of the Western Shield Program has been to develop cost-effective control techniques for feral cats (Possingham *et al.* 2004), as historical baiting programs for feral cats have been largely ineffective. 'Eradicat' bait has been developed as an effective alternative bait that reduces cat numbers across broad scale areas (Richards and Algar 2008), although this bait is not yet approved for release. In trial baiting programs at three former pastoral stations in the southern rangelands, the bait was successfully taken, with a subsequent decline in population of feral cats. Baiting in July 2006 reduced numbers of foxes and feral cats to zero immediately following the first baiting and for six months afterwards (Richards and Algar 2008).

The control of feral cats has also been examined experimentally at Heirisson Prong in Shark Bay. Populations of *Pseudomys albocinereus* (ash grey mice) and *Pseudomys hermannsburgensis* (sandy inland mice) responded strongly to feral cat control (Risbey *et al.* 2000). Capture success doubled where both foxes and cats were controlled. It was unchanged in the absence of any predator control, and declined by 80% over five years where only foxes were controlled. This particular result demonstrates a clear link between small mammal abundance and feral cat predation. In addition, it suggests that it is not only the critical weight range mammals that are vulnerable to predation, but also small mammals that are < 35 g.

## *Predator control measures in the GSS study area*

As yet, there is no coordinated fox or cat baiting program within the GSS study area (which is part of DEC's Swan Coastal District). Some reasons for the lack of fox baiting are given below.

- The potential risk associated with using 1080 baits in close proximity to an urban environment (i.e. private property boundaries and subsequent potential for dog deaths), and the probability of this risk increasing as urbanisation expands further north of Perth is considered high. This prevents the risk assessment process that must be undertaken in conjunction with a baiting application from being approved (N Powell pers. comm. 2009).

- The Western Shield Program targets threatened and priority species that are the most vulnerable to predation by introduced predators (primarily mammals within the critical weight range, but also some reptiles and ground nesting birds). Although the GSS study area provides suitable habitat for a diverse range of fauna (Valentine et al. 2009) most of the species that would be targeted by the Western Shield Program are well represented elsewhere in the state and in areas already being baited under the program. The western swamp tortoise is the only exception as it is restricted to the GSS study area and is critically endangered. The management of this species is discussed further below. *Hydromys chrysogaster* (rakali, Priority 4) is the only other species that is not well represented on the Swan Coastal Plain or other parts of the south-west of Western Australia that would potentially be targeted under the Western Shield Program. However, there are other threatening processes affecting this species' persistence – for example lowering ground water levels and declining rainfall. Unless these other threats are addressed in conjunction with fox control, a fox baiting program is not considered to provide any long-term benefit (N Powell pers. comm. 2009).
- The Western Shield Program does not currently have the resources to take on any additional areas without sacrificing some currently baited areas from the program. Any justification to do this must be based upon the need to protect existing high priority fauna populations or the needs of proposed or existing translocation programs outlined in threatened species recovery plans.

Whiteman Park is one area within the GSS study area where a fox control program has been carried out. This program has been operating since 1990. The fox control is undertaken by shooting, egg baits that are glued to PVC pipe in the ground and meat baits tethered to a stake. Both bait types contain the 1080 poison. Monitoring by Jackson (2003) has demonstrated that non-toxic bait uptake by foxes at Kings Park and Bold Park was higher than at Whiteman Park, suggesting some level of effectiveness of fox control at Whiteman Park. Due to continual re-invasion into Whiteman Park from surrounding unbaited areas and restrictions upon the locations within Whiteman Park where baits can be laid, Whiteman Park can only control foxes to a certain density. However, the populations of *Isoodon obesulus* (quenda) and *Macropus irma* (brush wallaby) within Whiteman Park have both increased in number since fox baiting commenced (C Rafferty pers.com. 2009).

### ***Pseudemydura umbrina*** **(western swamp tortoise) – a special case for fox control**

The provision of resources for the management of introduced species in DEC's Swan Coastal District, in particular within the GSS study area, is almost exclusively focused on protection of the western swamp tortoise. The critically endangered status of this reptile, for which the only remaining habitat occurs on the northern Swan Coastal Plain, generates funding and resources directed towards protecting both the species and its habitat.

Foxes are known to predate upon the western swamp tortoise, mainly during aestivation (Burbidge and Kuchling 2004). Twin Swamps and Ellenbrook Nature Reserves, both of which contain habitat critical for the tortoise, are enclosed by a 2.4 m high electrified fence, to prevent access by foxes, although they do occasionally break through this. Both reserves are also baited once every three to four months. If there is a fox incursion, immediate baiting is implemented to kill the fox as soon as possible. The regular baiting regime entails tethering between 20 and 60 baits on small stakes at 250 m intervals, mostly around the perimeter firebreak just inside the fence, and also some on internal firebreak tracks. As Ellenbrook and Twin Swamps Nature Reserves are located in urban, semi-rural areas, fox baiting with 1080 has been rated as low risk by the risk assessment process. Mogumber Nature Reserve and Moore River National Park, two areas of suitable habitat into which the western swamp tortoise has been translocated, are also baited monthly on perimeter and internal tracks. However, there is no fencing around the tortoise habitat at these two sites (R Martyn pers. com. 2009).

## *Herbivore and competitor control measures*

### Rabbits and pigs

Historical methods used to limit the distribution and abundance of rabbits include the erection of barrier fences, fencing of farm boundaries and water supplies, and poisoning of water (Abbott 2008). In more recent times rabbit control methods have included poisoning with baits, trapping, exclusion fencing around threatened plant populations, fumigation or destruction of warrens, and introduction of disease (Abbott 2008). Biological control such as the myxoma virus was effective initially before some resistance developed within the feral rabbit populations. However, this virus still maintains rabbits at 5% of the former population in wetter areas and 25% in arid areas (Williams *et al.* 1995). Rabbit calicivirus disease has also been more effective in wetter areas.

Feral pigs are controlled by mustering and shooting from helicopters, poisoning or erecting electric fences to protect small areas of high conservation priority. The most common method used by DEC for pig control is to erect a pen-style trap that can hold multiple pigs near where they are active, such as a waterhole or along a water course. The trap is baited with various food items such as grain, apples and molasses, and any pigs that are trapped are shot.

In the GSS study area, despite the widespread occurrence of rabbits (Valentine *et al.* 2009) they are not controlled by DEC's Swan Coastal District, partly due to the risk of secondary poisoning of non-target species. There is similarly no coordinated pig control program within the GSS study area. However, targeted pig control, aimed at protecting western swamp tortoise habitat, has been undertaken in wetland areas in Moore River National Park, Mogumber Nature Reserve and adjacent private property.

## Responses of wildlife to management of introduced species

### *Predator–prey interactions*

Controlling an introduced predator is not always beneficial to the native prey. For example, controlling foxes may lead to an increase in feral cats (meso-predator release), with consequent changes in predation patterns, and altered but equally severe impacts (Algar and Smith 1998). This was demonstrated at Heirisson Prong where feral cat counts increased three-fold in areas where management targeted foxes (Risbey *et al.* 2000). Meso-predator release can also affect ecosystem processes as well as native prey populations. Studies of feral cat eradication on sub-Antarctic Marion Island showed that prior to eradication, the cats preyed on introduced house mice, which in turn preyed on an endemic moth, which plays an important role on the island in nutrient cycling. Removal of the cats enabled the mice to build up, moth consumption increased and that led to subsequent changes in soil nutrient availability (Zavaleta *et al.* 2001).

Removal of a prey item with the intention of controlling a predator can also lead to prey switching, and the new prey may be a native species. This process was demonstrated in

New Zealand when deliberate removal of the introduced *Rattus rattus*, a prey item of the introduced predatory stoat caused the stoat to switch to native birds and bird eggs (Zavaleta *et al.* 2001). In addition, the introduction of myxomatosis in the 1950s, which significantly decreased the abundance of rabbits, may have temporarily intensified fox predation on native marsupials (Abbott 2008).

Conversely, high numbers of an introduced prey species facilitates a population increase of its predator (Risbey *et al.* 2000), which in turn can lead to greater consumption of native species. This hyperpredation has been demonstrated in Australia. (Zavaleta *et al.* 2001) showed that native mammals have only declined in areas invaded by feral cats if the densities of rabbits and house mice are also high. More specifically, native rodent decline has been attributed to direct predation by highly abundant foxes and feral cats in areas associated with the presence and high fecundity of rabbits (Risbey *et al.* 2000; Smith and Quin 1996).

The integrated control of both introduced predators and herbivorous species can significantly improve the status of mammal and reptile species. Following the removal of foxes, cats and rabbits from a fenced reserve at Roxby Downs in South Australia, native mammal abundance and richness significantly increased, with an unexpectedly huge leap in the abundance of native rodents (Moseby *et al.* 2009), attributed primarily to the absence of cat predation. Mammals that are lighter than the critical weight range may be more important in a feral cat's diet, as they continued to decline in the presence of fox control only. In addition, it was proposed that the increased grass cover and vegetation structure resulting from rabbit removal in the Roxby Downs reserve had played a key role in facilitating the increases in rodent numbers (Moseby *et al.* 2009).

In places where introduced herbivores and introduced plants co-occur, the removal of the herbivore can cause a proliferation of the introduced plant. Removal of the Asian water buffalo from Kakadu National Park facilitated large-scale regeneration of the wetlands. However, the introduced species para grass also proliferated in the absence of the water buffalo, and now covers 10% of the floodplain habitats (Zavaleta *et al.* 2001). Similarly, the removal of introduced herbivores can enable a different introduced fauna species to broaden its niche. For example, rabbit removal on Macquarie Island enabled a native tussock grass to expand its cover, subsequently providing habitat for the introduced *Rattus*

*rattus* (black rat), which then preyed upon ground-nesting bird species (Zavaleta *et al.* 2001). In these situations, management of introduced herbivores requires concurrent management of introduced plant species as well. The removal of introduced herbivores is likely to work the best when native vegetation still dominates in the habitat (Zavaleta *et al.* 2001).

# Summary of DEC–GSS projects (2007–09)

## Project 1. Feral sandpad survey

Technical report reference:
Reaveley A. (2009) *Preliminary findings from sandpad survey in Yanchep National Park*. Unpublished report prepared for the Department of Environment and Conservation, and the Gnangara Sustainability Strategy, Perth.

Fox activity in Yanchep National Park was quantified in August 2008 by using a discontinuous 40 km transect of vehicle tracks on both the west and east sides of Wanneroo Road, along which 47 sand pads were established. For four consecutive days following the establishment of the pads, the transect was driven early each morning, and at each pad the number of different prints and the species they belonged to were recorded.

The identifiable species recorded on the sand pads included *Macropus fuliginosus* (kangaroos), *Oryctolagus cuniculus* (rabbits), *Dromaius novaehollandiae* (emus), *Varanus gouldii* (Gould's monitor), *Vulpes vulpes* (foxes), and *Felis catus* (cats). There were also unidentifiable bird tracks. Fox tracks were the most prevalent but rabbit and kangaroo tracks were also frequent.

The overall results from the sand pad survey provide an index of fox activity, although not of fox abundance. An activity index for foxes can be ascertained by determining the average number of tracks on sand pads per day and then averaging the daily index (Morris 2009). Based on this methodology, the mean activity index for foxes in Yanchep National Park during this survey was 0.54. When compared to other sites in Western Australia that have been surveyed using the same method (DEC, unpublished data), the index for foxes at Yanchep National Park was clearly higher (Figure 8.1). Some sites were baited, some unbaited, and Yanchep National Park, which is an unbaited reserve, had the highest index of fox activity.

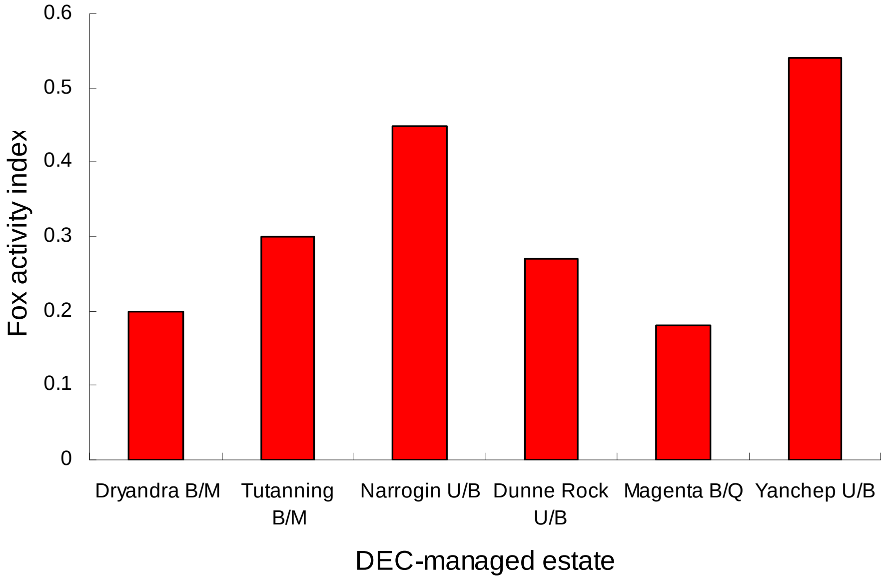


Figure 8.1: Comparison of fox activity recorded by the sand pad method in different areas of DEC-managed land. B/M = baited monthly, UB = unbaited, B/Q = baited quarterly.

The high levels of fox activity at Yanchep National Park indicate the potential benefit of a fox baiting program in the GSS study area. An integrated approach in which other threatening processes are managed (e.g. feral cat and rabbit control) would also need to be considered in conjunction with fox control.

## Project 2. Targeted quenda survey

Technical report reference: Valentine *et al*. (2009) *Patterns of ground-dwelling vertebrate biodiversity in the Gnangara Sustainability Strategy study area*. Unpublished report prepared for the Department of Environment and Conservation, and the Gnangara Sustainability Strategy, Perth.

In May 2008 a targeted survey was undertaken to locate populations of *Isoodon obesulus* (quenda) within the GSS study area. They had not been trapped during the general GSS fauna survey (Valentine *et al.* 2009), yet are one of the few medium-sized native mammals known to still be persisting in the area.

The targeted field survey focused on conservation reserves managed by either DEC or the City of Wanneroo that had been recorded as containing quenda in the 1990s (Friend 1996) or comprised habitat visually assessed as suitable for quenda. Nine sites were chosen, all of which comprised dense wetland-associated vegetation, as the known populations in Whiteman Park and Ellenbrook Nature Reserve were persisting only in this type of habitat (Bamford and Bamford 1994).

Five of the nine sites recorded captures of quenda, with the highest population being recorded at Twin Swamps Nature Reserve (Figure 8.2). This site is both fenced to prevent access by predators and internally baited to eradicate any fox invaders. Given the similarity in habitat between all five sites at which quenda were recorded (dense wetland and dampland associated vegetation), the high population of quenda in this nature reserve implies a correlation with the absence of foxes, and corresponding population suppression at the other sites due to predatory pressure.

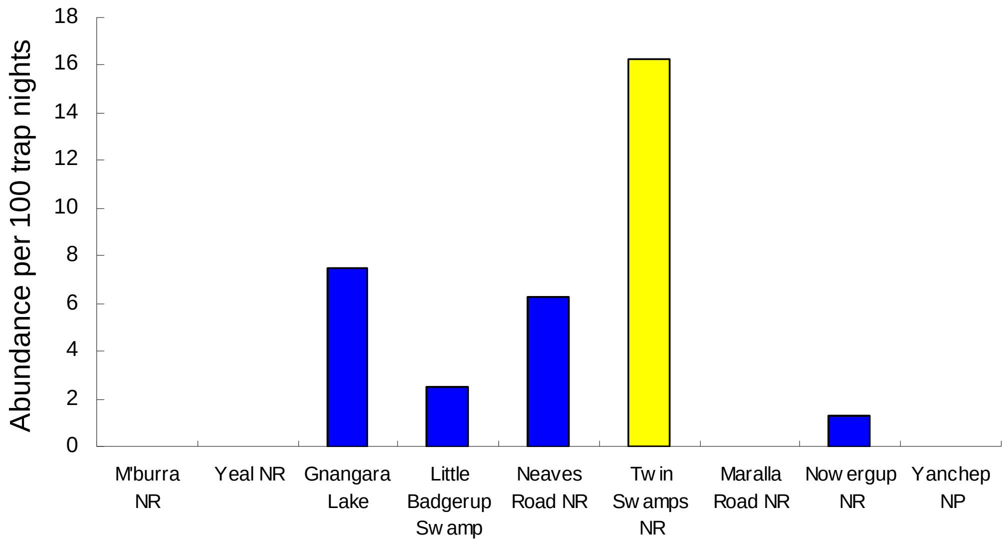


Figure 8.2: Sites at which quenda were recorded and their abundance per 100 trap nights at each site. Yellow indicates the only site (Twin Swamps NR) that was fenced and baited for introduced predators. Mburra = Muckenburra Nature Reserve.

The results of this survey also confirm the hypothesis that quenda in the GSS study area persist in habitat comprising dense 1 to 1.5 m high mid-storey vegetation, commonly associated with wetlands and damplands.

# Discussion

Introduced flora species have an ecological impact on native biodiversity by changing vegetation structure, increasing litter and litter breakdown rates, and altering fire regimes, nutrient conditions and hydrological patterns. These impacts can lead to loss of biodiversity and allelopathic effects.

Of the ~ 1900 plant taxa recorded within the GSS study area, 30% are classified as introduced, comprising 564 species. Thirty of these weeds have been identified as high priorities for management, including 25 terrestrial and 5 aquatic species. Their high priority for management is based on invasiveness, distribution and level of ecological impact. However, despite their threat to biodiversity, there are only limited resources directed towards weed management. The lack of sufficient recurrent funding for weed management and the high numbers of invasive weeds means that only a small set of the highest priority weeds are managed. If DEC acquires more estate (eg unallocated Crown land, ex-pastoral leases), the extent of weeds will become even more difficult to manage.

It is essential that weed management on the DEC estate is strategic. Ideally a systematic review and prioritisation of all local weed species in conjunction with biodiversity asset prioritisation is required to develop site-based and species-based management priorities in the GSS study area. However, a constraint of this review was the lack of time and resources to undertake a full assessment of the distribution and abundance of priority weeds in the GSS study area. In addition the impacts of many invasive environmental weed species are poorly understood and not well documented.

Three mechanisms by which introduced fauna species impact native biodiversity values are predation, competition, and habitat destruction. Predation has a direct impact by removing animals from a population, competition can exclude or displace native species from limited resources or habitat, and habitat destruction reduces the quality and/or extent of suitable habitat available for use by native species. In the GSS study area, predation by foxes and feral cats is a particularly significant threat, specifically contributing to the decline of native mammal species. Overall there are at least 33 introduced species currently recorded within the GSS study area and another ~ 16 species that could expand their distribution into the GSS study area. Most of the introduced species are birds, including Australian native species that have expanded their range. A limitation of this review was the inability to quantify the extent of the impact that introduced fauna species have in the GSS study area. Fox and feral cat densities in particular, have never been studied.

Despite the widespread occurrence of introduced species such as foxes and feral cats, rabbits and black rats, as well as aggressive, competitive species such as rainbow lorikeets and feral honeybees, there is negligible management directed towards the control of these species, other than management to protect the critically endangered *Pseudemydura umbrina* (western swamp tortoise). Predator control has been clearly demonstrated to benefit native fauna species such as quenda. Conflicts with baiting in an urbanised environment could be avoided by selecting high conservation sites (high mammal species richness) for baiting, with a surrounding buffer zone also baited, and fencing around the core area of protection considered as a supplementary option to strengthen the effectiveness of baiting. Whiteman Park provides a successful example. Fox baiting is likely to benefit species such as *Hydromys chrysogaster* (rakali) and *Macropus irma* (brush wallabies).

## Recommendations

In order to manage the effects of pest plants and introduced fauna, it is recommended that:

- systematic and targeted management programs be carried out to control populations of introduced fauna, using an integrated multi-species approach
- areas (e.g. Yeal Lake and wetland EPP 173 in Melaleuca Park) be selected for control of introduced fauna species, with a focus on fox and cat control
- further research be carried out in the GSS study area to identify the threat from introduced flora and fauna
- weed species of concern in the GSS study area be prioritised for management using an asset-based approach, with targeted management of priority wetlands such as those identified in Chapters 5 and 11.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter Nine: Distribution and Impacts of *Phytophthora cinnamomi*

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Department of Environment and Conservation
Our environment, our future

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GOVERNMENT OF WESTERN AUSTRALIA

Department of Water
Department of Agriculture and Food WA
Department for Planning and Infrastructure
Department of Environment and Conservation

WATER CORPORATION

CSIRO

Forest Products Commission WESTERN AUSTRALIA

# 9. DISTRIBUTION AND IMPACTS OF *PHYTOPHTHORA CINNAMOMI*

## Key points

- Disease in the Australian environment caused by *P. cinnamomi* is listed as a key threatening process under the EPBC Act.
- The disease has a severe impact on vegetation and fauna communities. These effects include declines in species richness and species' populations, decreases in vegetation cover and structure and degradation of fauna habitats.
- Approximately 10% (20 747 ha) of the GSS study area is infested with *P. cinnamomi*. The disease is widely distributed, but occurs predominantly within *Banksia* woodland.
- Eight threatened ecological communities located in the GSS study area were identified as having species susceptible to *P. cinnamomi*. Four of these communities were at high risk due to the high proportion of susceptible taxa in them and due to the significant effects the loss of key overstorey species may have on the community.
- A combination of on-ground interpretation, orthophotos and Landsat information was successfully employed to identify and map *P. cinnamomi* impacts.
- Estimated historical rates of spread of the pathogen in *Banksia* woodlands were at a peak from 1953 to 1963 and were at their lowest from 1997 to 2003.
- The impact of the pathogen on flora and birds in the *Banksia* woodlands were assessed and it was found that both plant and bird species richness were significantly lower in infested sites.
- Development of spatial models to predict distribution of the pathogen more accurately will improve the formulation of management options for areas deemed to be at risk of becoming infested.
- Dieback management plans need to be implemented for any activities within the DEC-managed estate that are likely to increase the risk of spread of *P. cinnamomi*.

# Introduction

The impacts of invasive species on native biodiversity values are a serious global problem (Vitousek *et al.* 1997). *Phytophthora cinnamomi* is a soil-borne water mould (Class Oomycetes) that is listed as one of the world's 100 most devastating invading species by the IUCN Species Survival Commission. Originally from the south-east Asian tropics, *P. cinnamomi* is an aggressive pathogen of numerous plant species around the world (Cahill *et al.* 2008). The plant pathogen has been shown to alter plant species abundance and richness, as well as the structure of vegetation in sclerophyllous vegetation throughout Australia (McDougall *et al.* 2002; Podger and Brown 1989; Shearer *et al.* 2007a; Weste 1974; Weste *et al.* 2002). The lethal epidemic of *Phytophthora* 'dieback' that occurs when there is a combination of plant species susceptibility, presence of the fungal pathogen and vulnerability due to favourable environments has been identified as a 'key threatening process' in the Australian environment (Environment Australia 2009; O'Gara *et al.* 2005).

Zoospores are considered to be the major infective propagule of *P. cinnamomi* (Cahill 1999; Phillips and Weste 1984; Weste and Cahill 1982). They are produced under warm and moist conditions from the vegetative state, or mycelia, that proliferate within or on host tissue. Zoospores may be carried in flowing water across the landscape, resulting in the rapid downslope spread of disease (Weste and Marks 1987). Zoospores encyst on plant roots and form a germ tube that penetrates the roots. The hyphae then colonise the plants and destroy the root and stem tissue of susceptible species (Shea *et al.* 1982; Tippett *et al.* 1987). The vascular system of the host plant is damaged, thus impeding water and nutrient supply. Plant to plant spread of the disease can also occur when mycelia grow from the infected host to the roots of adjacent plants.

Factors essential for *Phytophthora* dieback to occur include the presence of the pathogen (*P. cinnamomi*) itself, susceptible plant hosts and favourable environmental conditions (e.g. warm, moist, infertile soils, poor drainage) (Cahill *et al.* 2008; Shearer *et al.* 2007a). Natural dispersion of *P. cinnamomi* occurs by growth through contacting root systems or by propagules in water flowing through surface and near-surface drainage systems (Shearer *et al.* 2007a). *P. cinnamomi* can be carried long distances in infested soil that is moved by human activities (Shearer *et al.* 2007a).

The susceptibility of Australian plant taxa to *P. cinnamomi* varies among plant families and species, with more than 1000 native plant taxa known to be susceptible (McDougall 2005). Prominent Australian taxa that are susceptible include most of the Proteaceae family (e.g. *Banksia*), some of the Papilionaceae family, and a few of the Myrtaceae (e.g. *Eucalyptus marginata* (jarrah)) (Shearer *et al.* 2007a).

The presence of *Phytophthora* dieback in a community is often highly visible, with old diseased areas typically displaying reduced biomass and structural complexity as a result of the removal of susceptible taxa. Areas where *P. cinnamomi* is active (the infection front) are identified by progressive stages of dead and dying vegetation (Shearer *et al.* 2007a). The impacts of *Phytophthora* dieback on native vegetation can be severe, leading to major changes of plant community composition, structure and function (Cahill *et al.* 2008; Shearer *et al.* 2007a). Consequences of *Phytophthora* dieback may include loss of susceptible flora species, reduction in primary productivity and biomass, major disruption to plant community structure and degradation to remaining habitat for flora and fauna (Cahill *et al.* 2008; Environment Australia 2001; Hill *et al.* 1994; Shearer *et al.* 2007a; Wilson *et al.* 1994). Modification of vegetation structure and floristics associated with *P. cinnamomi* infestation has been linked to changes in small mammal communities, including reduced species richness and abundance of fauna in diseased vegetation, and decreased capture of individual species (Annett 2008; Laidlaw and Wilson 2006; Newell and Wilson 1993; Wilson 1990).

In general, the consequences of *Phytophthora* dieback on the diversity and functioning of communities are inadequately understood, and management of *P. cinnamomi* infested areas is a major problem (Cahill *et al.* 2008; Shearer *et al.* 2007a). Current management strategies include hygiene measures, *ex situ* seed conservation, and fungicide control (Shearer *et al.* 2007a). The systemic application of the fungicide potassium phosphite to plants in infested areas can significantly reduce the mortality rate, and increase the length of time required to achieve 50% mortality in an area (Shearer and Fairman 2007).

The Gnangara Sustainability Strategy study area is located within south-west Western Australia, one of the world's 32 biodiversity hotspots (Mittermeier et al. 2004). The area is renowned for its floristic diversity and endemism (Beard *et al.* 2000; Hopper and Gioia 2004). The study area is also known for the richness of its terrestrial vertebrate fauna (DEC

2009), particularly the reptilian fauna, and was historically a rich zone for mammal species, both in diversity and abundance (Abbott 2006). The impact of *P. cinnamomi* on flora, fauna and ecosystems has been identified as a major threatening process in the Department of Environment and Conservation's Swan Region (DEC 2009). Although there is evidence of the occurrence of *P. cinnamomi* on the Gnangara groundwater system since the 1940s, information on its distribution and impacts is limited. In their audit of biodiversity of the Swan Coastal Plain (SWA2) IBRA subregion, Mitchell *et al.* (2003) identified that, in general, plant communities with susceptible plant species can be considered ecosystems at risk and that there is little quantitative data on the effect of the pathogen.

The aims of this chapter are to:

- collate and review information on the impacts of *P. cinnamomi* infestation on biodiversity
- address gaps in our knowledge of the occurrence and distribution of the pathogen in the GSS study area
- assess the impacts of the pathogen on the biodiversity values of the area
- assess current management regimes.

These aims were addressed by reviewing and assessing literature and data on:

- the impacts of *P. cinnamomi* infestation on biodiversity
- the historical distribution and impacts within the GSS study area
- the susceptibility of flora and fauna in the GSS study area
- risk analyses undertaken to assess *P. cinnamomi* threats to biodiversity.

In addition, four DEC–GSS projects were undertaken to address knowledge gaps for the GSS study area. These were:

**1) Identification of threats to floristic diversity and threatened ecological communities**

The objectives of this project were to assess the susceptibility of plant species to the pathogen and to determine the likely impacts on threatened ecological communities.

**2) Current extent of *P. cinnamomi* across the GSS study area**

As part of the regional-scale Project Dieback study the Forest Management Branch of DEC compiled spatial information on the extent of *P. cinnamomi* across the Swan and Northern Agricultural regions. The objective of this study was to utilise this data to examine the current distribution and location of *P. cinnamomi* infestation across the GSS study area.

**3) Mapping and spread of *P. cinnamomi* in *Banksia* woodland (DEC–Leeuwin Centre, CSIRO)**

This project aimed to assess if historical aerial photos and Landsat data could be used to identify and map changes in vegetation cover over time due to the impact of *P. cinnamomi*. This involved using an index derived from Landsat imagery to identify areas where vegetation has been affected by *P. cinnamomi*, mapping historical changes in vegetation, and rates of spread over a 50-year period at a long-term study site.

**4) Assessment of impacts of *P. cinnamomi* on flora and fauna (2008–09)**

The objectives of this project were to assess the impacts of *P. cinnamomi* on fauna and habitats in *Banksia* woodland. This involved field studies to compare the avifauna, terrestrial vertebrates (mammals, reptiles) and invertebrates inhabiting infected and uninfected sites, and to compare the vegetation floristics and habitat structure at these sites.

These projects address a number of the approaches to landscape pathology (Figure 1) proposed by Holdenrieder *et al.* (2004). The major findings and results of these projects have been summarised in this report. Technical papers detailing the work are also available and are listed in Appendix 2.

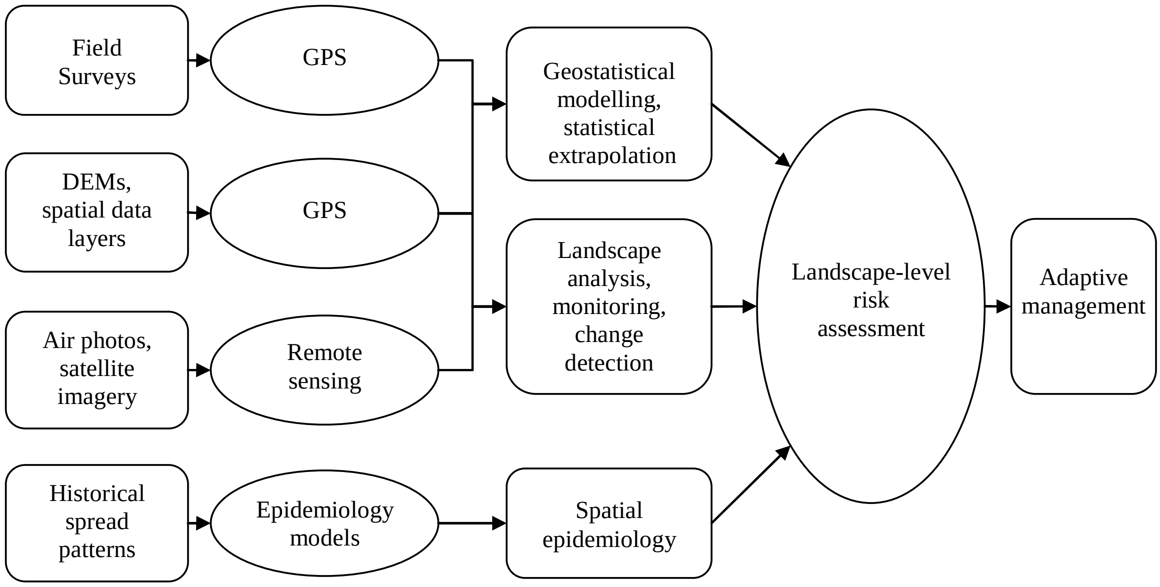



Figure 1: Data sources, quantitative tools and analytical approaches central to landscape pathology (DEM – digital elevation model) (Holdenrieder *et al.* 2004).

## Impacts on flora

The effect of *P. cinnamomi* on flora in Australia was first reviewed in the 1980s (Podger and Brown 1989; Weste and Marks 1987). Knowledge of the impacts of the pathogen has increased significantly since this time, but problems associated with managing the disease remain complex. Species have been designated as susceptible or resistant from field observations, where isolation of the pathogen from soil and roots has been obtained, or from field or glasshouse experiments or a combination of these (Cahill *et al.* 2008). Nevertheless the impact of the pathogen is known only for ~ 5% of Australia's vascular flora.

*P. cinnamomi* is known to mainly affect woody perennial species, the most susceptible families being Proteaceae, Fabaceae, Dillenaceae, Epacridaceae and Xanthorrhoeacae (Podger and Brown 1989; Shearer and Dillon 1995; Weste 1994; Weste and Marks 1987; Wills 1993), whereas many herbaceous perennials, annuals and geophytes (e.g. Asteraceae and Poaceae) survive or are resistant to infection. Susceptibility to the pathogen also varies considerably within families, genera and species. Although monocotyledons are generally less susceptible to *P. cinnamomi,* they can act as hosts and are thus an inoculum source of the pathogen (Cahill *et al.* 1989; Kennedy and Weste 1986; Podger 1972; Podger and Brown 1989).

Typical symptoms of the disease observed in the field include yellowing of foliage from lack of chlorophyll production (chlorosis), retarded growth, and plant death. The pathogen infects the roots of host plants, causing root and collar rot that leads to restricted transpiration (Dawson and Weste 1984; Marks and Smith 1991). In susceptible species superficial roots, major roots and the lower stem may be infected (Dawson and Weste 1984; Marks *et al.* 1981; Shea *et al.* 1982), and the pathogen may also invade the phloem and cambium (Tippett *et al.* 1987). In resistant species such as rushes, sedges and grasses, inoculation results in penetration, infection and root lesions, but the lesions are confined, and new healthy roots develop (Phillips and Weste 1984; Tippett *et al.* 1985; Weste 1994).

Approximately 10% of plant species listed as nationally threatened under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* are considered to be threatened by *P. cinnamomi* (Environment Australia 1999; McDougall 2005). In Western Australia 96% of the species of Proteaceae rated as priority taxa are susceptible to *P. cinnamomi* (Wills and Keighery 1994). Eight endangered species from the south coast of Western Australia are highly susceptible to the disease (Shearer *et al.* 2007a; Wills and Keighery 1994).

Studies in Western Australia show that approximately 2000 of the 6000 to 8000 taxa in the south-west botanical province are directly susceptible to infection by *P. cinnamomi* (Colquhoun and Hardy 2000; Komorek *et al.* 1994; Shearer *et al.* 2004; Wills 1993). Shearer *et al.* (2004) estimated that of these, approximately 800 species are considered to be highly susceptible. The potential impact of *P. cinnamomi* on native plants of south-west Western Australia is considered to be extreme given the extraordinarily high plant diversity (Cahill *et al.* 2008).

## Impacts on vegetation communities

The impact of *P. cinnamomi* on vegetation communities has been examined in temporal studies where control and impact sites are assessed over time, and in chronosequence studies where sites distributed spatially across the landscape are considered to correspond to different infestation ages and impacts (Cahill *et al.* 2008). *P. cinnamomi* has been shown to alter plant species abundance and richness, as well as the structure of vegetation, in sclerophyllous vegetation throughout Australia (McDougall *et al.* 2002; Podger and Brown 1989; Shearer and Dillon 1996a; Weste 1974; Weste *et al.* 2002; Wills 1993). Studies of infected vegetation have shown that *P. cinnamomi* has a major impact on the understorey, with up to 60% of the plant species eliminated after infection (Dawson *et al.* 1985; Duncan and Keane 1996; Kennedy and Weste 1986; Laidlaw and Wilson 2003; Weste 1974; Wills 1993). Infection results in decreased species diversity, seedling regeneration, and declines in the populations of susceptible species (Dawson *et al.* 1985; Kennedy and Weste 1986; Weste 1974). In contrast, there is often an increase in the frequency and cover of resistant monocotyledons (Annett 2008; Dawson *et al.* 1985; Duncan and Keane 1996; Kennedy and Weste 1986; Laidlaw and Wilson 2003; Weste 1986).

*Phytophthora cinnamomi* has been recorded in all states and territories of Australia (O'Gara *et al.* 2005). It has been found to cause disease in a range of vegetation types, including stringybark (*Eucalyptus obliqua*) and silvertop ash (*E. sieberi*) forests (Victoria), jarrah (*E. marginata*) forests (Western Australia), species-rich woodland and heathland communities (New South Wales, Victoria, Western Australia, Tasmania) and has been recorded in heathlands and tropical rainforest of Queensland (Gadek 1998; Newhook and Podger 1972; Weste 1994).

The long-term impact of *P. cinnamomi* on native vegetation has been investigated at very few localities. However, changes in floristic composition have been recorded at several sites in Victoria (Weste 2003; Weste *et al.* 2002). Studies which began in the 1970s in forests of the Brisbane Ranges (Victoria) showed that plant deaths occurred for half the species present within six months of infestation, and that on severely diseased sites more than 40% of mature trees (*E. macrorhyncha, E. baxteri)* died and understorey shrubs were replaced with graminoids (Weste *et al.* 1973; Weste and Taylor 1971). In the 1980s highly susceptible species such as *Xanthorrhoea australis* and *Isopogon ceratophyllus* became locally extinct, tree density declined, bare ground increased greatly and graminoids increased in abundance (Weste 1986). In the 1990s resprouting of moderately susceptible species such as *B. marginata*, and *Grevillea steiglitziana* was recorded and the cover of bare ground declined to 10% (Weste and Ashton 1994). Between 1971 and 1998 the percentage of *P. cinnamomi* isolations substantially declined and the pathogen could no longer be found at two of three sites in 1998 (Weste 2003). It is unclear if the regeneration observed will continue should the conditions of the original disease epidemic return. *Phytophthora* dieback may then recur as a cycle of disease outbreaks (Walchhuetter 2001; Weste *et al.* 2002).

In southern Victoria, in the Great Otway National Park, symptoms of *P. cinnamomi* disease were first identified in 1972 and have been continually observed throughout the area (Aberton *et al.* 2001; Annett 2008; Laidlaw and Wilson 2003; 2006). Significant differences have been found in both floristics and structure between diseased and non-diseased heathland vegetation (Laidlaw and Wilson 2003). Species from the families Proteaceae and Ericaceae, and the keystone *Xanthorrhoea australis*, became extinct or were lost locally after infection and post-disease areas are dominated by species such as *Leptospermum continentale*, *L. myrsinoides*, and *Gahnia radula*. There was a significant decline in cover in diseased vegetation and significant changes in floristic composition in post-disease areas. Long-term mapping of the disease showed a progressive change in distribution over time (1988–2006) (Annett 2008). Further, the mean number of plant species decreased by approximately half, the abundance of species declined, and the percentage cover of species decreased.  The long-term consequence of loss of species and structure in the area implies that the vegetation is unlikely to return to its former status, especially if the pathogen continues to reinfect the persisting vegetation.

In Western Australia the death of patches of jarrah (*Eucalyptus marginata*) forests to the east and south-east of Perth was first reported in the 1920s, but the association between plant dieback and *P. cinnamomi* was not made until the 1960s (Podger 1968). The impact of *P. cinnamomi* on jarrah forest has been studied intensively (McDougall 1997). Many of the understorey species are susceptible to infection, including *Banksia grandis.* There is a rapid decline in most susceptible species and although *E. marginata* trees may die they can survive with reduced canopy. Although many grasses and sedges are unaffected there is a significant increase in the area of bare ground in the years following infection. Some susceptible species recolonise from soil-stored seed, but do not survive. After 20 years some resistant shrubs and perennial herbs invaded or increased in cover but there was still a significant amount of bare ground on infested sites (McDougall *et al.* 2002). Although there was evidence of recolonisation and survival in infested sites after 50 years, recovery of canopy and understorey cover was limited (McDougall *et al.* 2002).

## Effects of the disease on fauna and faunal habitats

The severe vegetation degradation and significant alterations in plant communities associated with *P. cinnamomi* infection should also substantially affect fauna through changes to major resources such as food and availability of nesting sites, together with habitat and protective cover (Garkaklis *et al.* 2004; Wilson *et al.* 1994). Effects on mammal species are likely to be variable (Garkaklis *et al.* 2004; Wilson *et al.* 1994). Pseudomyine rodents such as *Pseudomys shortridgei* (heath mouse) and *P. novaehollandiae* (New Holland mouse) are dependent on floristically diverse understorey and are threatened by the loss of plant species diversity as a result of *P. cinnamomi* infestation of their habitat. Specialised species such as *Tarsipes rostratus* (honey possum) may be very susceptible as *Phytophthora* dieback reduces the availability of its proteaceous food plants (Garavanta *et al.* 2000; Wooller *et al.* 2000). In comparison, rodent species such as *Rattus lutreolus* (swamp rat), and *R. fuscipes* (bush rat), that require dense, low vegetation for shelter may prefer habitat that has been affected by *P. cinnamomi*, if there has been a consequent increase in the cover of resistant monocotyledons.

A study in woodlands of southern Victoria found that the percentage of vegetation modified by *P. cinnamomi* was a significant variable in explaining variations in small mammal diversity and density (Wilson 1990; Wilson *et al.* 1990). Further studies in the area to examine the effects of the pathogen on the microhabitat and populations within species rich mammal communities found that fewer small mammal species were captured in post-diseased vegetation than in non-diseased vegetation (Laidlaw 1997; Laidlaw and Wilson 1989; 2006). The mean capture rate of small mammals overall was also significantly lower in the post-disease areas compared to the active disease and non-diseased areas. Three species, *R. lutreolus* , *R. fuscipes*, and *Antechinus agilis* (agile antechinus), were also less abundant in diseased heathland than in healthy areas, and this was found to be directly related to both floristics and structure of the vegetation (Laidlaw and Wilson 2006).

A study on the effects of *P. cinnamomi* on the bush rat (*Rattus fuscipes*) in south-west Western Australia also found a marked reduction in bush rat numbers in a disease-affected area and most trap captures were in dense vegetation (Whelan 2003). A study of the impact of *Phytophthora* dieback on *Antechinus flavipes* (mardo) in jarrah (*Eucalyptus marginata*) forests found a significant difference in trap success of mardos, with the highest in sites with no impact of *Phytophthora* and the lowest trap success recorded at the high impact sites (Armistead *et al.* 2004).

While there have been some studies investigating the impacts on mammalian taxa and communities, research on the impacts of *P. cinnamomi* on birds, reptiles, frogs and invertebrates is limited (Garkaklis *et al.* 2004; Newell 1997; Nichols and Bamford 1985; Nichols and Watkins 1984; Wilson *et al.* 1994). There is evidence that *Phytophthora* dieback affected sites in the jarrah forests of Western Australia have fewer bird species, and lower abundance, but the patterns of declines between sites are not consistent (Nichols and Watkins 1984). *P. cinnamomi* affected jarrah forests appeared to support lower numbers of reptiles and frogs species than healthy forests, although some species were more abundant in diseased forest (Nichols and Bamford 1985). This benefit could be attributable to increased sunlight available for these species which require elevated surfaces (e.g. logs) for basking and foraging (Nichols and Bamford 1985). While the effects on reptiles remain unclear because of the scant empirical data, it is likely that significant changes to vegetation would have an effect on reptile communities and populations.

Many nectar producing plant species such as those of the Proteaceae are susceptible to *P. cinnamomi* infection (Wills and Keighery 1994). Reduction in nectar producing plants is likely to affect vertebrates that use this resource, and also plant species that require vertebrates for pollination. The magnitude of this impact is not known as effects of the disease on this function have not been adequately assessed. One study found that 59% of plant species with vertebrate pollinated flowers were susceptible to *P. cinnamomi*, indicating that vertebrate pollinators would be vulnerable (Wills 1992).

## Mapping and remote sensing of *P. cinnamomi* impacts

The distribution of *P. cinnamomi* in Australia has been assessed by mapping of records derived from isolation, on a national scale (O'Gara *et al.* 2005). However, development of on-ground management actions to address infestations at a local level depends on knowledge of the location and extent of the disease in the landscape. A number of methods have been employed to map the boundaries of infestations, including on-ground surveys of symptoms, soil sampling for presence of the pathogen, and interpretation of disease symptoms using aerial photographs (Bluett *et al.* 2003; Cahill *et al.* 2002; Daniel *et al.* 2006; Hogg and Weste 1975; O'Gara *et al.* 2005; Wilson *et al.* 1997).

Some difficulties involved with mapping are evident. For example, the pathogen may be detected, but may not cause disease, or symptoms may not be visible due to the absence or low numbers of susceptible species. The useful life of maps of infestation boundaries is limited because of the natural rate of spread of the pathogen as well as the accelerated rate of spread of the pathogen under suitable conditions. Aerial photography is also considered ineffective for mapping in areas where the disease is restricted to understorey vegetation that is covered by a dense, *P. cinnamomi* resistant emergent layer.

In Western Australia the disease status of vegetation is routinely mapped before logging and mining operations, using indicator plant species (CALM 2003a). The disease interpretation process of DEC in Western Australia involves a systematic process for detecting, diagnosing, demarcating and mapping of *P. cinnamomi* (O'Gara *et al.* 2005). Initial interpretation is undertaken where possible from aerial photographs, followed by on-ground surveys based on the identification of disease symptoms in indicator species (CALM 2001). Disease interpretation has also been employed in the conservation estate to identify areas considered to be ‘protectable’ in the medium to long term and to be given priority management.

Methods for mapping the extent of disease caused by *P. cinnamomi* do have limitations. For example, they are expensive and rely on the availability of trained interpretative personnel. Remote sensing and image analysis methods have been considered as suitable by organisations with remote sensing capacity, for addressing some of these concerns, and are being increasingly adopted as tools for broad scale disease mapping, assessment and management (Metternicht 2007; Nilsson 1995; Olsson *et al.* 2008; Ristaino and Gumpertz 2000). Examples include assessment of forest health (Chaerle and Van der Straeten 2000; Liu *et al.* 2006; Stone *et al.* 2001) and extent of crop disease (Apan *et al.* 2004; Lenthe *et al.* 2007; Mirik *et al.* 2006; Zhang *et al.* 2003). Satellite remote sensing technologies are appropriate for monitoring vegetation dynamics over time. They offer quantitative, repeatable methods using multi-date imagery that can be applied over large areas, and can be extended both retrospectively and into the future as data is captured on an ongoing basis (Furby *et al.* 2004; Pickup *et al.* 1993). Landsat satellite imagery, for example, has the spatial resolution and historical records that make it suitable for providing information for assessing change at scales ranging from small remnants to whole regions (Wallace *et al.* 2006).

An alternative to satellite imagery is remote sensing via airborne videography in combination with digital multi-spectral imagery (DMSI; Hill *et al.* 2009; Lamb 2000; Wallace *et al.* 2006; Wallace *et al.* 2007). Studies in *Banksia* woodlands on the Gnangara groundwater system have demonstrated that this method can detect changes at the level of individual trees (Wallace *et al.* 2006; Wallace *et al.* 2007). A system for assessing *P. cinnamomi* related disease in the Bald Hills Heath in southern Victoria has also been successfully developed using this technique (Hill *et al.* 2009). Fine detail maps provided rapid availability of data that could be used for accurate measurement of the extent of disease and its changing status.

## Predicting the distribution of *P. cinnamomi*

While mapping and remote sensing provide information on the historical and current extent of disease together with patterns of change, there is a need to determine where the pathogen is likely to spread in the future in order to provide management options. Spatial models, employing multivariate statistical analyses and a geographic information system, have been used to predict the distributions of organisms and their habitats (Gibson *et al.* 2004a; Gibson *et al.* 2004b; Guisan and Thuiller 2005; Guisan and Zimmermann 2000). These models have also been employed recently for predicting pathogen (e.g. *P. cinnamomi)* distribution (Holdenrieder *et al.* 2004; Wilson *et al.* 2000; Wilson *et al.* 2003; Wilson *et al.* 1997). The categorisation of habitat likely to be affected by *P. cinnamomi* displayed in the spatial model can be used to identify areas requiring protection, such as those which are presently disease free and contain rare or threatened species with a high probability of becoming infected.

A model of *P. cinnamomi* distribution at a local scale was developed for heathlands in the Anglesea district (Victoria), where GIS was employed to record site data, provide accurate estimation of spatial variables such as elevation, slope and contributing catchment area, and to develop a predictive spatial model for the distribution (Wilson *et al.* 2000; Wilson *et al.* 2003; Wilson *et al.* 1997). Logistic regression analysis of 17 variables (measured at sites and from the GIS attributes) identified two variables (elevation and sun-index) as statistically significant in determining the probability of *P. cinnamomi* infestation. The presence of infection was negatively associated with elevation (i.e. the lower the elevation, the more likely the presence of the pathogen) and positively associated with the sun-index (i.e. places that have a steep northerly aspect). These findings are consistent with the downhill spread of zoospores with free water movement and suggest that within the mid elevations of the catchment, the warmth associated with northerly aspects is conducive to *P. cinnamomi* activity. The main limitation of this model is that it contains no dynamic elements (Austin 2002). Extrapolation from this type of static spatial model could be improved by considering the dynamics of the driving variables and feedback processes of the pathogen and the environmental and landscape factors at the sites.

## The rate of natural spread

The rate of natural spread of the pathogen is strongly influenced by topography, vegetation and climate. Annual rates of spread are highly variable, ranging from several metres to hundreds of metres downslope in gullies or watercourses. In Western Australia, upslope disease extension on the Darling Plateau is 0.37 m/year, compared to 2.15 m/year for the Blackwood Sedimentary Plateau where a perched watertable provides long periods of favourable conditions for proliferation of the pathogen (Strelein *et al.* 2005). In the jarrah forest, upslope and across slope spread seldom exceeds an average of 1 m per year (Podger et al. 1996 cited in O'Gara *et al.* 2005).

A spatial model was developed to assess correlations between *P. cinnamomi* disease and site characteristics in the Wet Tropics World Heritage Area (S. Worboys pers. comm; Gadek *et al.* 2001). The project was undertaken to determine if outbreaks of disease are associated with particular site characteristics, as although the pathogen is uniformly distributed in the area, disease expression does not always occur. Areas of disease were found to be correlated with acid–igneous geology, flat areas where drainage is impeded, notophyll dominant vegetation and elevations of 750 m and higher (Gadek *et al.* 2001).

## *P. cinnamomi* risk assessment and priority setting

The disease caused by *P. cinnamomi* in natural systems is listed as a key threatening process under the EPBC Act. The National Threat Abatement Plan for *Phytophthora* was developed with its major objectives being to promote the recovery of threatened species and ecological communities under threat, and to limit spread of the pathogen (CPSM 2006; Environment Australia 2001). Projects to address these objectives have developed processes and criteria to assess the risk to biodiversity (Wilson *et al.* 2005), and provide benchmarks for management (O'Gara *et al.* 2005). The essential requirements for risk assessment and management of disease are accurate knowledge of where it occurs, which species and communities are threatened, and where risks and consequences of infestation are likely. A review of the implementation and effectiveness of the National Threat Abatement Plan was undertaken in 2006 (CPSM 2006), and provided revised goals, objectives and actions. A risk assessment process has been developed for *P. cinnamomi* on the Gnangara groundwater system (see Chapter 11).

## Management of *P. cinnamomi*

Prior to the release of the National Threat Abatement Plan 2001, a range of legislation and management plans were developed to address the problem at a state level. In Victoria, for example, a strategic management plan for *P. cinnamomi* in national, state and metropolitan parks was developed (Anon 2005; DSE 2008; Parks Victoria 2000). Recovery plans for a number of species considered to be threatened by *P. cinnamomi* have also been prepared (English 1999; Evans *et al.* 1999; Keith 1997). A number of guidelines for *P. cinnamomi* management have been developed in Australia to satisfy a variety of applications. Examples include the *Assessment of guidelines for best practice management of Phytophthora cinnamomi in parks and reserves across Victoria* (Cahill *et al.* 2002) and the Tasmanian *Interim Phytophthora cinnamomi management guidelines* (Rudman 2004). Those developed for Western Australia are described below.

The major management strategies used to protect flora species and plant communities threatened by *P. cinnamomi* in the south-west botanical province of Western Australia have been hygiene and quarantine measures (Shearer and Tippett 1989). Currently, there are no proven methods to eradicate *P. cinnamomi* from a site or to prevent autonomous spread of the pathogen. As a result, two major management objectives are:

- to minimise the spread to uninfested sites by restricting access to them and enforcing hygiene procedures
- to mitigate the impact at infested sites, including control trials with chemicals and monitoring.

Long-term *ex situ* seed conservation (Cochrane and Coates 2004) and translocations (Monks and Coates 2002) have also been employed to recover critically endangered taxa.

Another method is the aerial application of the systemic fungicide phosphite (Barrett 2003). The chemical phosphite (phosphonate), the anionic form of phosphonic acid, has been deployed to slow the spread and reduce the impact in susceptible vegetation with some success (Cahill *et al.* 2002; Hardy *et al.* 2001). Its use is now recognised as a major strategy for disease mitigation in native vegetation (CALM 1999; Podger *et al.* 1996). Phosphite is applied by direct spraying of, or injection into, plants, or by aerial application. Although the mechanisms of phosphite action are complex and not fully understood, it evidently acts both directly on the pathogen and indirectly by stimulating plant defence responses (Daniel and Guest 2006; Guest and Grant 1991). Phosphite does not eradicate the pathogen, which remains in the soil–host plant environment, even though symptoms are suppressed.

In Western Australia guidelines for the management of *P. cinnamomi* have been developed (CALM 2003b) to provide staff with clear and concise descriptions of the methods and standards. Management guidelines include:

- *Phytophthora cinnamomi and disease caused by it. Volume 1 – management guidelines* (CALM 2003b)
- *Phytophthora cinnamomi and disease caused by it. Volume 2 – interpreter's guidelines for detection, diagnosis and mapping* (CALM 2001)
- *Phytophthora cinnamomi and disease caused by it. Volume 3 – phosphite operations guidelines* (CALM 1999).

A number of management guidelines have also been developed for particular industries as well as for community organisations:

- *Management of Phytophthora dieback in extractive industries* (DWG 2005)
- *Managing Phytophthora dieback: guidelines for local government* (DWG 2000)
- *Managing Phytophthora dieback in bushland: a guide for landholders and community conservation groups* (Dunne 2005).

## Distribution and impacts of *P. cinnamomi* on the GSS study area

*P. cinnamomi* is widely distributed in *Banksia* woodlands of the Swan Coastal Plain (Podger 1968; Shearer 1994). *Banksia* woodland, already extensively cleared and fragmented for urban development and agriculture, is further threatened by the presence of *P. cinnamomi*. However, the occurrence, population dynamics and impact of the pathogen in this community is poorly understood (Shearer and Dillon 1996b). A survey of the occurrence and impact of *P. cinnamomi* disease in *Banksia* woodland was conducted in national parks and reserves of the Swan Coastal Plain south of Perth (Shearer and Dillon 1996b). *P. cinnamomi* was isolated from dead plants or soil at 46 diseased areas. Diseased areas were estimated to be from 0.01 to 30 ha in size and the total area infested was 71.5 ha. Infestation was associated with decreased species number, with on average seven fewer species in infested areas compared to non-infested areas. Occurrence of *P. cinnamomi* was closely related to soil type, with 60% of the disease centres occurring on Bassendean or Southern River associations of the Bassendean Dune system. No disease centres were identified on the Spearwood or Quindalup Dune systems despite a representative number of national parks and reserves being surveyed. A large proportion of disease areas were associated with anthropogenic disturbance (such as firebreaks – 72%) and groundwater within 3 m of the soil surface (48%). These results strongly suggest that human activities are responsible for the introduction of the pathogen and that there is a relationship with groundwater depth.

Those plant communities of the Swan Coastal Plain occurring on infertile soils of poor drainage, including leached sands, are considered to provide very favourable conditions for epidemic development of *P. cinnamomi* (Shearer 1994). The population dynamics of *P. cinnamomi* have not been determined for these soils and better understanding of population dynamics is required in order to determine the impacts of the disease over time and between locations, and to determine the efficacy of control strategies (Shearer *et al.* 2007a).

Shearer *et al.* (2007b) examined the direct and indirect impacts of *P. cinnamomi* in the south-west botanical province of Western Australia, including the Swan Coastal Plain. They assessed impacts on biodiversity and ecosystem dynamics, the proportion of threatened ecological communities infested, and the declared rare flora threatened, either directly or indirectly, by *P. cinnamomi* infestation. A comparison of mortality curves for common species and declared rare flora species in *P. cinnamomi* disease centres determined that common species on the coastal plain such as *Banksia attenuata* and *B. grandis* reached 50% mortality in 7 to 12 years, whereas mortality rates for declared rare flora were much more rapid, with local extinction of most of the assessed declared rare flora occurring in < 3 years (Shearer *et al.* 2007b).

*P. cinnamomi* infestation also caused significant changes in ground and canopy cover (Shearer *et al.* 2007b). In woodlands the ground cover (40%) in old infested areas was reduced compared with adjoining healthy vegetation (68%). Canopy cover was reduced from 48% in healthy to 25% in old diseased areas.

Of the current 340 declared rare flora in the south-west botanical province of Western Australia, 86 are threatened either directly or indirectly by *P. cinnamomi*. Of these, 26% still occur on the Swan Coastal Plain (Shearer *et al.* 2007b). Some 56% of taxa are threatened indirectly compared to 44% that are directly threatened. Shearer *et al.* (2007b) reported that ten threatened ecological communities on the coastal plain are infested with *P. cinnamomi*. This includes 25 sites of the threatened ecological community *Banksia attenuata* woodland over species rich dense shrublands (community type 20a as described Gibson *et al.*1994), which occurs in the GSS study area.

There has been no long-term monitoring of changes in understorey composition following *P. cinnamomi* infestation of plant communities of the south-west botanical province of Western Australia, comparable to that undertaken in Victoria by Weste (2003). Some responses of vegetation to *P. cinnamomi* infestation are available for *Banksia* woodland on Bassendean dunes (Shearer and Dillon 1996b). The number of perennial plant species declined from 14 in healthy vegetation to 9 in old infested areas, and the cover of susceptible species was lower in the old infested area than in the adjoining healthy woodland. Decrease in species numbers following infestation of *Banksia* communities by *P. cinnamomi* have previously been reported (2004; 1996a; 1989).

Approximately 2300 of the 5710 described plant species in the south-west botanical province have been recorded as being susceptible to *P. cinnamomi*, with 800 being highly susceptible to the pathogen (Shearer *et al.* 2004). Because many of the highly susceptible species occur frequently and are structurally dominant, their death following infestation leads to a prominent reduction in biomass, with associated lower floristic diversity and capacity of infested sites to support dependent biota (Shearer and Dillon 1995; Shearer and Dillon 1996a; b; Wills 1993). *Banksia* woodlands infested with *P. cinnamomi* within the GSS study area have been observed to show reduced canopy cover, declines in understorey species richness, and less ground cover and biomass (Hill *et al.* 1994; Shearer *et al.* 2007a).

The first evidence of *P. cinnamomi* infestation in the GSS study area was observed from aerial photographs taken in the 1940s. Treeless patches 20 to 30 m wide in *Banksia* woodland could be seen spanning tracks radiating from nearby horticultural farms (Hill *et al.* 1994). Further review of aerial photographs by Hill *et al.* (1994) showed steadily expanding disease centres from which the spread of disease could be measured over a period of 35 years. More than 50% of the *Banksia* woodland visible in the early photographs had been destroyed by 1988. An estimated mortality rate of 50% can be reached within 6.5 years for *Banksia attenuata*, and 10 years for *B. grandis* (Shearer *et al.* 2007a).

## Management of *P. cinnamomi* in the GSS study area

### *Dieback management plans and strategic planning*

Dieback management plans are prepared on a local area basis for a specific reserve or area of DEC estate. The objective of these plans is to manage access to and hygiene in *P. cinnamomi* free 'protectable' areas to ensure that human activities are an inconsequential vector for establishment of new centres in them.

Actions to meet this objective include mapping and identification of protectable areas and implementing standard hygiene practices and access controls. Some of these are:

- a restriction of activities to only when soil conditions are dry
- all vehicles and machines must be ‘clean on entry’
- no soil movement is permitted
- restrictions on machinery activities near swamps.

All development proposals and operations involving the movement of soil on DEC estate and within leases on DEC estate require the inclusion of a dieback management plan within the construction environmental management plan of the proposal. A condition for approval is that mapping and identification of protectable areas must be carried out by an accredited interpreter to standards outlined by Forest Management Branch (Volume 1 Management Guidelines CALM 2003b). Within the GSS study area this has been applied to proposals for clearing for utilities, infrastructure and mining (e.g. power line and gas pipeline construction and maintenance, road upgrades, and sand extraction).

All DEC operations involving the movement of soil, such as firebreak upgrades for prescribed burning, track upgrades and forestry activities also require dieback plans to be approved by district managers and/or regional fire coordinators. Within the GSS study area, this mainly relates to fire and other necessary operations. Plantation forestry activities generally occur in areas that are now uninterpretable and therefore cannot be mapped, so dieback management plans are not required.

Strategic planning for *P. cinnamomi* in the Perth natural resource management area is currently being developed as an initiative of Project Dieback (Strelein *et al*. 2008). A plan, *Managing Phytophthora cinnamomi for biodiversity conservation in the Perth and Avon NRM regions*, is aimed at providing a strategic context for investment in *Phytophthora* dieback response within the region, and at engaging stakeholders. The plan is based on a 25-year vision statement and aims to provide a strategic approach to investment in management of the disease over a seven-year period (2010–17).

## *Phosphite*

Phosphite has not been commonly used by DEC as a management treatment within the GSS study area, although it is widely used by the department on the south coast to protect declared rare flora populations (M Pez pers. comm.). Treatment has been carried out by private contractors for local government authorities or by private landholders and volunteers, for example Chittering housing estates, Whiteman Park (targeted treatment), Lightning Swamp (Noranda) (C Dunne pers. comm.). The Dieback Working Group has also applied treatments for local government using volunteers or private contractors, and volunteer groups also apply phosphate treatment in nature reserves (C Dunne pers. comm.). Threatened ecological communities that have been treated include '*Banksia attenuata* woodland over species rich dense shrublands' (community type 20a); '*Banksia attenuata* and/or *Eucalyptus marginata* woodlands of the eastern side of the Swan Coastal Plain' (community type 20b) and 'Shrublands and woodlands of the eastern side of the Swan Coastal Plain' (community type 20c as described by Gibson *et al.* 1994). Declared rare flora that may have been treated include *Conospermum undulatum* and *Caladenia huegelii* (C Dunne pers. comm.).

## *Monitoring*

In south-west Western Australia and within the GSS study area, infestations on DEC estate are not monitored past the life of the current dieback occurrence map (three years). However, disease boundaries should be rechecked every 12 months within those three years to keep the map current. A full interpretation is recommended after three years if there are continuing or new activities within the mapped area. Monitoring of the rate of spread may be carried out periodically and there are infestations that are frequently monitored as part of DEC Science Division research plots (mostly on the south coast). In an area with high value assets (e.g. declared rare flora), monitoring may be done in conjunction with a phosphite program.

# Summary of DEC–GSS projects (2007–09)

## Project 1. Threats from *P. cinnamomi* to floristic diversity and threatened ecological communities in the GSS study area

Technical report reference:
Swinburn, M (2009) *Phytophthora* susceptibility of GSS flora and threatened ecological communities. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

Of the ten threatened ecological communities occurring within the study area, six have been identified previously as being at risk of infestation by *P. cinnamomi* because of the presence of susceptible plant species (Mitchell *et al.* 2003). The objectives of this project were to determine the susceptibility of native plant taxa in the study area and reassess the susceptibility of threatened ecological communities and the likely impacts of the pathogen on them.

Information on the responses of native plant taxa recorded in the study area to *P. cinnamomi* was collated, based on a review of recent publications including O'Gara *et al.* (2005). Susceptibility ratings ranged from 'field resistant' to 'highly susceptible'. Species that are typical or common to each threatened ecological community were identified (CALM 2006; DEC 2008; English and Blyth 2000; English *et al.* 2002; Gibson *et al.* 1994; Meissner and English 2005) and the susceptibility of taxa was then assigned based on information from the review. A risk potential rating was then assigned to each of the communities based on the proportion of species susceptible, the proportions that are typical or define the community type and the number or proportion of overstorey species that are susceptible (Swinburn 2009). Removal of overstorey species was presumed to have greater potential to affect the community by altering the structure, and thus habitat, for taxa including fauna and understorey flora species.

Information on the potential responses to *P. cinnamomi* was available for 240 native vascular plant taxa which represents ~ 18% of known native plant taxa in the area. Of the 128 taxa, 53% display some degree of susceptibility to the pathogen or its indirect effects on plant communities. The predominant families identified with susceptible taxa are the Proteaceae (32 spp.), Epacridaceae (20 spp.), Papilionaceae (16 spp.) and Myrtaceae (14 spp.). Taxa from 33 families, mainly the Myrtaceae (n=16) and Cyperaceae (n=13), have been identified as being field resistant, with no reports of susceptibility being found during this assessment.

Eight threatened ecological communities were identified as having species susceptible to *P. cinnamomi*, ranging from 1 to 42 species per community. Although the number of taxa susceptible to *P. cinnamomi* in each of these communities is low, the proportion of the taxa is as high as 53% (Table 9.1). Four threatened ecological communities were identified as at 'high risk' (Table 9.1) including the '*Banksia* woodland community' (community type 20a; Gibson *et al.* 1994) which had the highest number of susceptible species (n = 42) or 36% of the taxa for which responses are available. The susceptible species include dominant overstorey taxa such as *Allocasuarina fraseriana*, *Eucalyptus marginata*, and *Banksia* species. In addition to the direct impacts of species declines and loss of structural characteristics, understorey species are also likely to be indirectly affected by the pathogen as a result of loss of the overstorey and subsequent changes to growing conditions. Of the 20 known occurrences of this threatened ecological community in the study area, one is currently highly affected and another is known to be infested (DEC 2008).

Shrublands and woodlands on Muchea limestone were also assessed as high risk due to the large proportion of species (53%) potentially susceptible to the pathogen. Although dieback has not been recorded in this community it may be present (English and Blyth 2000). However, *P. cinnamomi* is not commonly isolated from limestone soils (C. Dunne pers.comm.). The '*Melaleuca* shrublands on limestone ridges' (community type 26a; Gibson *et al.* 1994) was assessed as at moderate risk but there is no evidence of *P. cinnamomi* disease in this community, which occurs on shallow soils over limestone (V. English pers. comm.). The other threatened ecological communities were considered to be at low risk (Table 9.1).

Susceptibility ratings in Table 9.1 are as follows:

- Low – few or no susceptible species
- Moderate – between 20% and 30% of species are susceptible but few are typical or define the community, or there are few overstorey species

- High – more than 30% of species are susceptible or a large proportion of susceptible species are typical or define the community, or belong to the overstorey.

Table 9.1: Threat potential for threatened ecological communities at risk from *P. cinnamomi* based on proportion of susceptible taxa.

| **Threatened ecological community** | **Number of susceptible taxa** | **Number of known taxa** | **% susceptible taxa** | **Risk** |
|---|---|---|---|---|
| Shrublands and woodlands on Muchea limestone | 10 | 19 | 53 | High |
| Woodlands over sedgelands in Holocene dune swales of the southern Swan Coastal Plain (community type 19b; Gibson *et al.* 1994) | 4 | 11 | 36 | High |
| *Banksia attenuata* woodland over species rich dense shrublands (community type 20a; Gibson *et al.* 1994) | 42 | 118 | 36 | High |
| Communities of tumulus springs (Organic mound springs, Swan Coastal Plain) | 7 | 29 | 24 | High |
| *Melaleuca* shrublands on limestone ridges (community type 26a; Gibson *et al.* 1994) | 13 | 56 | 23 | Mod |
| *Callitris preissii* forests and woodlands, Swan Coastal Plain (community type 30a; Gibson *et al.* 1994) | 5 | 30 | 17 | Low |
| Perth to Gingin ironstone association (Northern ironstones) | 5 | 45 | 11 | Low |
| Herb rich saline shrublands in clay pans (community type 7; Gibson *et al.* 1994) | 1 | 14 | 7 | Low |
| Forests and woodlands of deep seasonal wetlands of Swan Coastal Plain (community type 15; Gibson *et al.* 1994) | 0 | 14 | 0 | Low |
| Aquatic root mat community of caves of Swan Coastal Plain (Yanchep caves) | 0 | 1 | 0 | Low |

Limitations inherent in the assessment methods used in this project include incomplete information on the susceptibility or field resistance of species and difficulties in determining whether a species is affected directly through infection or indirectly through habitat changes (O'Gara *et al.* 2005). In addition, arbitrary limits were used to classify threatened ecological communities into risk categories. It may be more appropriate to assess the relative cover of susceptible species, rather than the proportion of susceptible species in the community, in order to assign a risk classification.

# Project 2. Current extent of *P. cinnamomi* across the GSS study area

Technical report reference: Kinloch, J (2009) *Mapping the current extent of P. cinnamomi in Banksia woodlands in the GSS study area.* Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

As part of the regional scale Project Dieback study, the Forest Management Branch (DEC) has compiled spatial information about the extent of *P. cinnamomi* across the Swan and Northern Agricultural regions. The objective of this project was to examine the current occurrence and extent of *P. cinnamomi* across the GSS study area, based on the Project Dieback mapping. This included operational interpretation mapping, aerial photographic interpretation and targeted field observations (Strelein *et al.* 2007). The status of areas were classified as infested, uninfested, unmappable (disturbances present that mask *P. cinnamomi* impact), uninterpretable (lack of susceptible vegetation) or not interpreted (< 50 ha, or cleared). Areas were also assigned a confidence level dependent on factors such as disease expression and presence of vectors. The proportion of each landform infested in the GSS study area was calculated. A total of 20 747 ha (10%) of the study area is classified as infested and is distributed across all land uses ranging from small urban remnants to large areas in the conservation estate (Figure 9.2). Most of the infested areas are classified as low or medium confidence, reflecting the limited amount of operational interpretation mapping in the north. Areas classified as uninterpretable included the Quindalup dunes and wetlands, and those as unmappable are predominantly pine plantations.

Ninety-four per cent of infested areas were located on the Bassendean Dune system, supporting previous studies of disease impact on the Swan Coastal Plain (Shearer 1994; Shearer and Dillon 1996a). Infestation on the Spearwood Dune System was low (3%) reflecting low impact on coastal limestone, and most of the Quindalup Dune system was classified as uninterpretable. Only 2% of infested areas were located on the soils of the Pinjarra Plain where the vegetation is considered susceptible to the disease. The low levels of impact probably reflect the high levels of clearing rather than resistance to the disease.

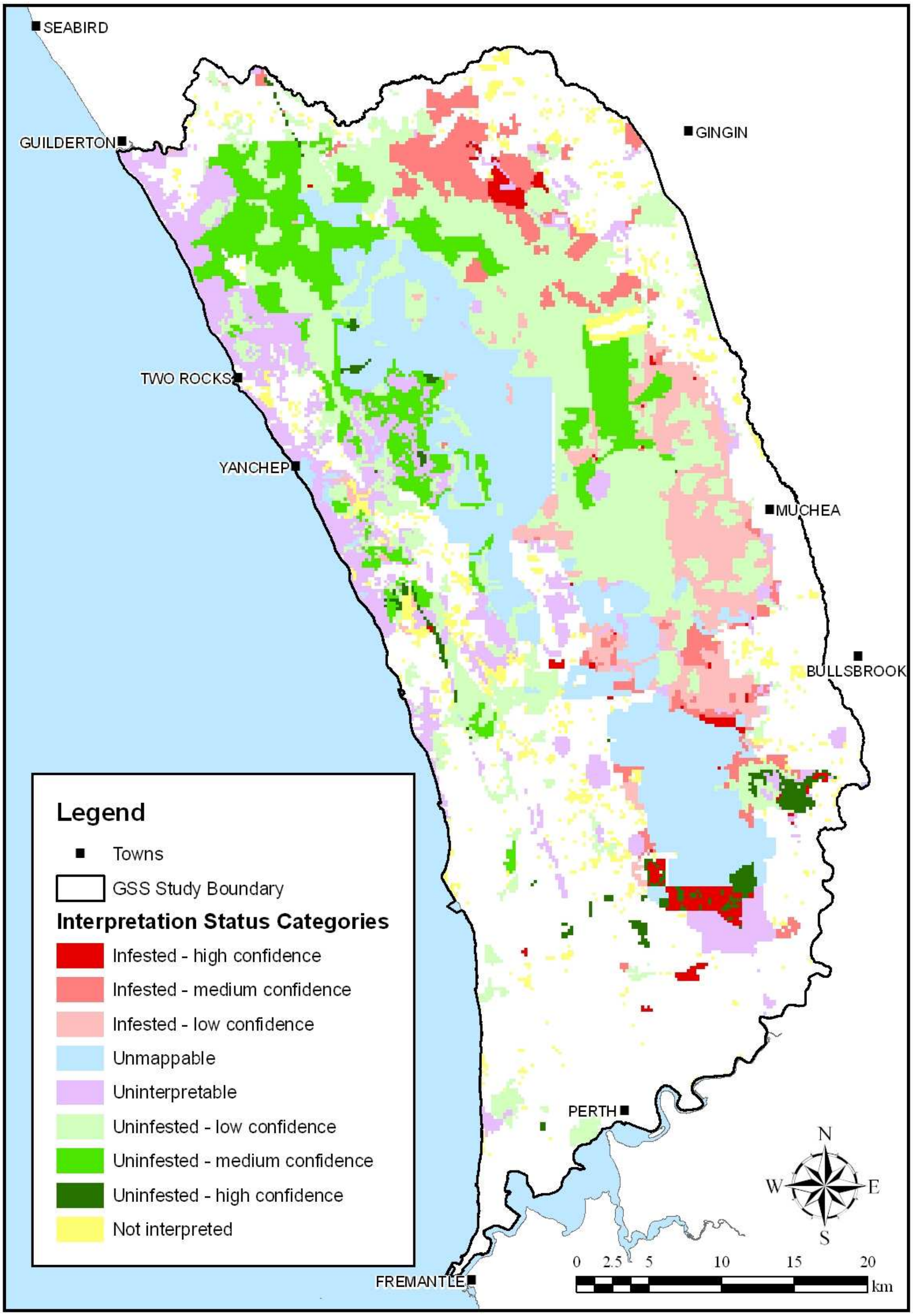



Figure 9.2: *P. cinnamomi* interpretation mapping for the GSS study area.

# Project 3. Mapping and spread of *P. cinnamomi* in *Banksia* woodland (DEC–CSIRO)

Technical report references:
1) Zdunic, K, Behn, G and Kinloch, J (2009) *Mapping the extent of P. cinnamomi in Banksia woodlands in the GSS study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

2) Behn, G and Zdunic, K (2009) *Development of remotely sensed vegetation cover index for the Gnangara Sustainability Strategy and vegetation cover trends analysis.* Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

Information on disease patterns and rates of spread on the Gnangara groundwater system was provided by Hill *et al.* (1994) who used historical orthophotos to map the location of disease fronts retrospectively at a site within the study area (Warbrook Road). While the use of aerial photos to map the impact of *P. cinnamomi* is an established method, satellite remote sensing has seldom been employed. The objectives of this project were to:

- determine if trends in vegetation cover, using an index derived from Landsat imagery, can be used to identify loss of vegetation cover in areas affected by *P. cinnamomi*
- map historical changes in vegetation, related to the impact
- determine rates of spread over a 55-year period .

Orthophotos (geometrically corrected aerial photographs) were obtained for January 2008 at four *Banksia* woodland areas: Warbrook Road, Neaves, Gnangara and Pinjar. A combination of orthophotos and Landsat trend information were employed to identify locations affected by disease. The current extents of the affected areas were mapped using 2008 orthophotos, and ground validation was subsequently undertaken at Gnangara, Neaves and Warbrook. *P. cinnamomi* affected areas were successfully identified (Warbrook Road, Pinjar, Gnangara) but no affected areas were identified at the Neaves study area, which was consistent with ground surveys. Comparison of the on-ground mapped boundaries with remote techniques, found that boundaries were not always entirely aligned, indicating the limitations of remote techniques in identifying disease areas where there has been little impact on overstorey species.

Historical orthophotos from between 1953 and 2003 were obtained for the Warbrook Road area. Where digital orthophotos were not available hardcopies were scanned and geometrically corrected. An analysis of trends in vegetation cover using an index derived from Landsat imagery between 1988 and 2007 was undertaken (see Furby *et al.* 2007; Wallace and Thomas 1999). The historical extent of the *P. cinnamomi* affected areas was then mapped, at a scale of 1:5000. The mapped boundaries for each year were combined to show the disease front contours during the last 55 years and the cumulative area affected was calculated.

It was determined that trends in vegetation cover derived from Landsat imagery were useful in aiding the identification of *P. cinnamomi* affected areas. The linear trends provided information on those areas of the landscape where cover had declined during the last 20 years. Some areas that had been infected for a number of years did, however, exhibit fluctuating trends, indicating vegetation cover had increased sometime during the period, possibly from colonisation by tolerant species (Figure 9.3). The low resolution of Landsat data (25 m) means that Landsat trends alone cannot be used to map the extent of disease affected areas. Rather, when historical aerial photos are not available, Landsat trends can provide useful data on vegetation cover over time to help determine if changes in vegetation cover are likely to be associated with *P. cinnamomi* impacts or other processes such as fire, clearing or grazing. Figure 9.3 shows an aerial photograph of the extent of the *P. cinnamomi* infestation in an area just north of the Gnangara pine plantation and the Landsat linear trends between 1988 and 2007 for the same area. The pink lines indicate the extent of the disease in 2008. For the Landsat trend analysis, red indicates a major negative trend, orange–yellow a negative trend, green fluctuating trends, black no major change, and blue a positive trend.

(a) 

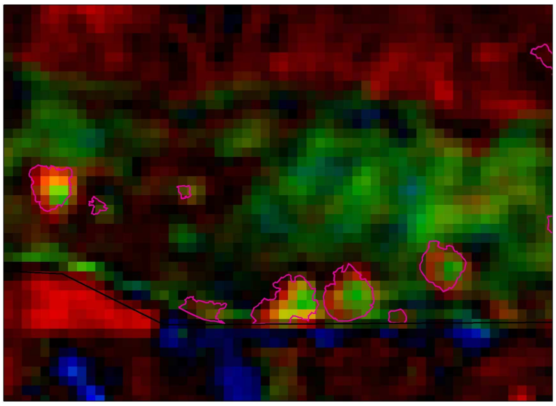
(b) 

Figure 9.3: (a) 2008 Aerial photograph showing the extent of the disease caused by *P. cinnamomi* in an area just north of the Gnangara Pine Plantation and (b) the Landsat linear trends between 1988 and 2007 for the same area.

## *Historical changes in vegetation at Warbrook Road*

Hill *et al.* (1994) determined that the initial infection of the Warbrook Road area is likely to have occurred before 1942 and that during subsequent decades the swamp complex at the site became contaminated, initiating disease fronts several kilometres long. They estimated that up to 50% of the woodland was severely affected by 1988. Our study has revealed that the disease has continued to expand to cover 68% of the study area (47.8 ha) in 55 years (Table 9.2, Figure 9.4). The period 1974 to 1988 appeared to be a particularly active period for the disease, with a number of new infections recorded in the 1988 mapping. Several discrete disease fronts coalesced to form two long fronts in the south-west and north-east of the study area. The areas that remain uninfested are of the same vegetation type and therefore are susceptible. The presence of tracks in these areas is likely to result in infestation of these areas also.

Table 9.2: Total cumulative area, and the proportion of the study area, infected with *P. cinnamomi* for a number of years.

| **Orthophoto date** | **Cumulative area infected (ha)** | **Proportion of study area infected (%)** |
|---|---|---|
| 1953 | 15.3 | 32.0 |
| 1963 | 18.0 | 37.7 |
| 1974 | 21.7 | 45.5 |
| 1988 | 27.1 | 56.7 |
| 1992 | 29.2 | 61.1 |
| 1997 | 30.8 | 64.5 |
| 2003 | 31.8 | 66.6 |
| 2008 | 32.5 | 68.1 |

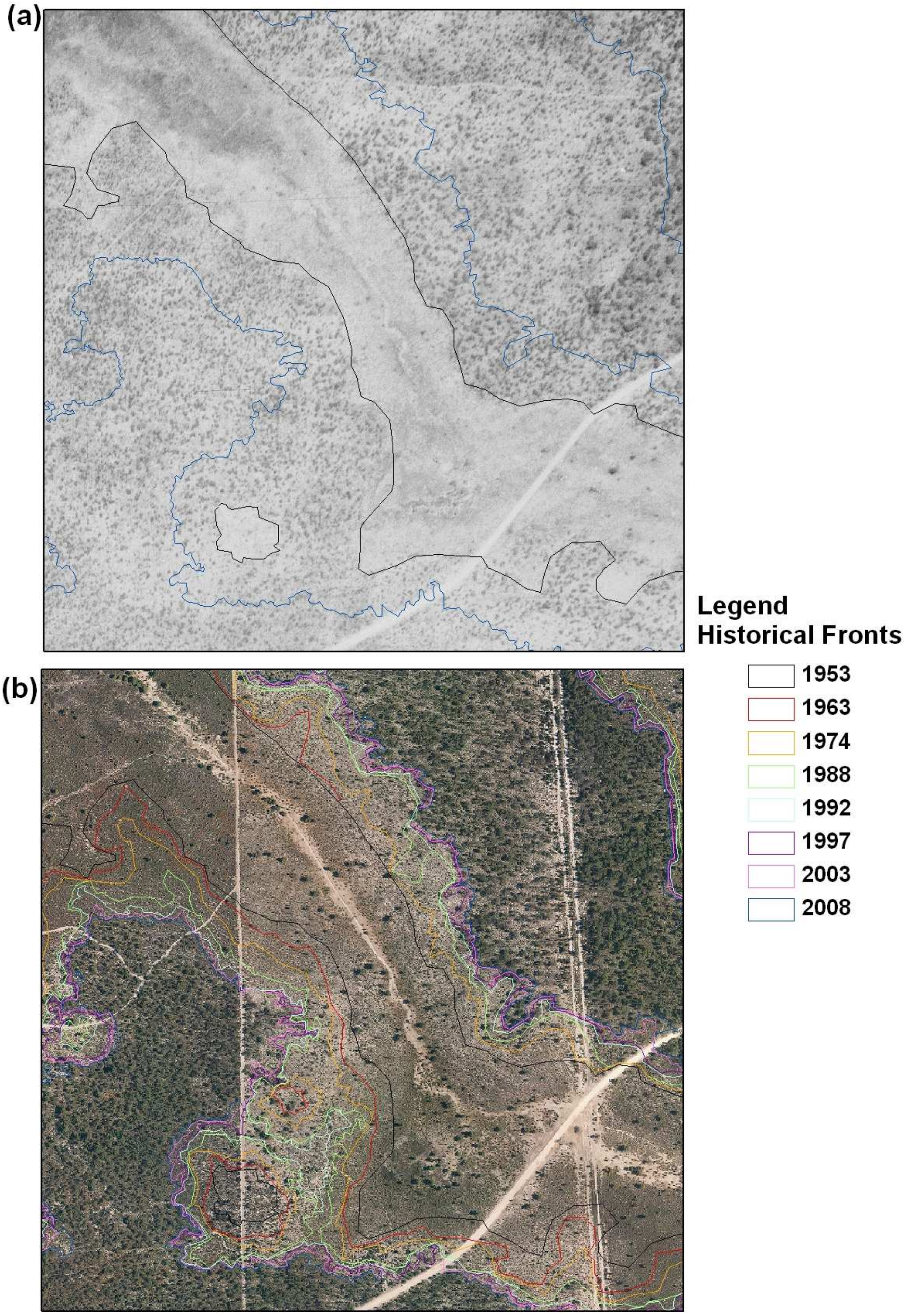



Figure 9.4: Aerial photographs of the study area in (a) 1953 and (b) in 2008. In (a) the lines show the extent of the disease in 1953 and 2008 and in (b) all time periods. Note the coalescence of discrete patches to form continuous fronts.

Mapped fronts for each year were combined to determine the spread for each time interval and the rate of spread was then calculated as the mean distance advanced for each time interval divided by the number of years in the time interval (Table 9.3).

Table 9.3: The time intervals that the rate of spread of *P. cinnamomi* was assessed at Warbrook Road including their length in years.

| **Time interval** | **Length of the time interval years** |
|---|---|
| 1953–1963 | 9.3 |
| 1963–1974 | 9.8 |
| 1974–1988 | 12.8 |
| 1988–1992 | 5.8 |
| 1992–1997 | 4.0 |
| 1997–2003 | 6.0 |
| 2003–2008 | 5.0 |

The rate of *P. cinnamomi* spread was greatest in the period 1953 to 1963, steadily declining to the lowest rate between 1997 and 2003 (Table 9.4). The rates of spread between 1953 and 1988 calculated in this study cannot be directly compared to those determined by Hill *et al.* (1994), as different methods were used to calculate spread. However, they are relatively close and thus support our methods. From 1992 onwards the rate of spread declines significantly, possibly in response to the decline in rainfall. The declining rate of spread may also be related to disease fronts moving predominantly through higher slope areas in the later years. Future work should investigate the relationship between the rate of spread and slope and also other landscape and topographical variables such as aspect and soil type.

Table 9.4: Rate of spread of *P. cinnamomi* at Warbrook Road over several time intervals as calculated by Hill *et al.* (1994) over 35 years and this study (GSS) over 55 years.

| | **Rate of spread (m year$^{-1}$)** | |
|---|---|---|
| **Time interval** | **Hill *et al.* (1994)** | **GSS** |
| 1953–1963 | 1.31 | 1.4 |
| 1963–1974 | 1.36 | 1.1 |
| 1974–1988 | 0.89 | 1.1 |
| 1988–1992 | N/A | 0.8 |
| 1992–1997 | N/A | 0.6 |
| 1997–2003 | N/A | 0.3 |
| 2003–2008 | N/A | 0.4 |

# Project 4. Assessment of impacts of *P. cinnamomi* on flora and vertebrate fauna (2008–09)

Technical report references:

1) Swinburn, M, Sonneman, T, Mickle, D and Valentine, L (in prep) *Impacts of Phytophthora dieback on flora and fauna in the GSS study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

2) Davis, R (2009) *Impact of fire and dieback on birds in the Gnangara Sustainability Strategy*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

Although *P. cinnamomi* infested *Banksia* woodlands in the GSS study area have depleted canopy cover, understorey species richness, ground cover, and biomass (Hill *et al.* 1994; Shearer *et al.* 2007a), there have been no studies of the effects on fauna. The objectives of this project were thus to assess the effects of *P. cinnamomi* on fauna and habitats in *Banksia* woodland. This involved an assessment of those native fauna taxa in the GSS study area likely to be highly susceptibility to *P. cinnamomi*, and field studies to compare species richness, abundance and composition of birds and terrestrial vertebrates between uninfested and infested sites. In addition plot species richness, canopy cover and other vegetative habitat attributes were compared between sites.

## *Dieback interpretation and vegetation floristics*

In 2008 seven *Banksia* woodland sites that contained large dieback fronts were chosen with the aid of aerial photography and existing dieback mapping. Dieback infection fronts were interpreted and demarcated by Forest Management Branch (DEC) at each of seven fauna survey sites before floristic quadrats were established (Pez and Swinburn 2009). Twenty-one 10 m x 10 m floristic quadrats were established at seven avifauna survey sites within paired 1 ha dieback infested and uninfested sites (see avifauna section below). Three floristic quadrats were located at each site, one quadrat in an uninfested area, one within a buffer of the infection front (transition), and one within infested vegetation (Figure 9.5). These quadrats were surveyed during two months in spring 2008. Species identification has now been completed by a botanist with assistance from the Western

Australian Herbarium, and information will be used for future analyses. Percentage live canopy cover was estimated within quadrats.

**a)** **b)** **c)**

Figure 9.5: Floristic site Pc7 in: a) uninfested woodland with healthy *Eucalyptus todtiana* and *Banksia* spp.; b) transition zone of the disease front with a fresh *Banksia* death at left of photo and older death at the right; and c) infested woodland with disease front and *Banksia* spp. deaths in background (Photos: David Mickle).

Comparisons of plant species richness and live canopy cover were compared between infested, transition and uninfested quadrats. There was a significant difference in mean plant species richness ($P < 0.05$) and percentage live canopy cover ($P < 0.01$) between uninfested, transition and infested sites. Post hoc analysis revealed a significant difference in mean plant species richness ($P < 0.01$) and percentage live canopy cover ($P < 0.01$) between infested and uninfested sites (Figure 9.6).

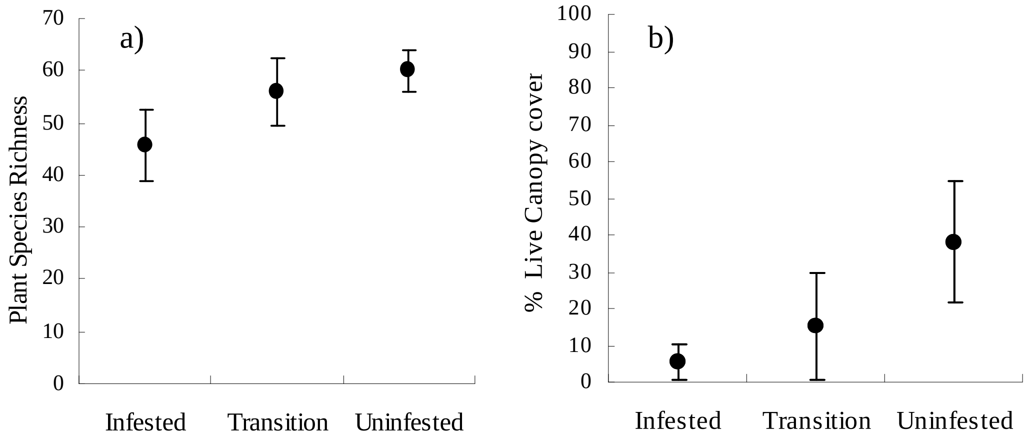



Figure 9.6: a) Mean (± 95% CI) plant species richness and b) percentage live canopy cover in *P. cinnamomi* infested, transition and uninfested floristic quadrats.

## *Susceptible fauna species in the GSS study area*

Information on the likely susceptibility of native fauna to *P. cinnamomi* in the GSS study area was collated. The information and susceptibility ratings were based on previous studies and information on the ecology and distribution of the taxa that would make the species susceptible (Garavanta *et al.* 2000; Garkaklis *et al.* 2004; Laidlaw and Wilson 1989; 2006; Wilson *et al.* 1994; Wooller *et al.* 2000). Eight birds, five mammals and three reptiles have been identified as being potentially susceptible to the direct or indirect effects of *P. cinnamomi* infestation (Table 9.5). The 'conservation threat' for species has been identified – for example threats to food resources or to habitat structure due to dieback.

Table 9.5: Fauna likely to be affected by *P. cinnamomi* infestation of *Banksia* woodland in the GSS study area.

| **Scientific name** | **Common name** | **Factors likely to be affected by *P. cinnamomi*** |
|---|---|---|
| | **Reptilia (reptiles)** | |
| *Menetia greyii* | common dwarf skink | Habitat structure |
| *Morethia obscura* | shrubland pale-flecked *Morethia* | Habitat structure |
| *Cryptoblepharus buchananii* | Buchanan's snake-eyed skink | Habitat structure |
| | **Aves (birds)** | |
| *Acanthiza apicalis* | inland thornbill | Habitat structure |
| *Acanthiza inornata* | western thornbill | Habitat structure, food resources? |
| *Acanthorhynchus supercilliousus* | western spinebill | Food resources, habitat utilisation |
| *Calyptorynchus latirostris* | Carnaby's black-cockatoo | Conservation threat, food resources |
| *Lichmera indistincta* | brown honeyeater | Habitat utilisation |
| *Malurus splendens* | splendid fairy wrens | Habitat structure, food resources? |

| Scientific name | Common name | Factors likely to be affected by *P. cinnamomi* |
|---|---|---|
| *Sericornis frontalis* | white-browed scrubwren | Habitat structure |
| *Zosterops lateralis* | silvereye | Habitat utilisation |
| | **Mammalia (mammals)** | |
| *Isoodon obesulus fusciventer* | quenda | Conservation threat |
| *Macropus irma* | western brush wallaby | Conservation threat |
| *Rattus fuscipes* | bush rat | Habitat utilisation, habitat structure |
| *Tarsipes rostratus* | honey possum | Food resources, conservation threat |
| *Trichosurus vulpecula hypoleucus* | western brushtail possum (SW mainland) | Conservation threat |

## *Distribution of Avifauna in uninfested and infested habitat*

In 2008 seven *Banksia* woodland sites that contained large dieback fronts were chosen with the aid of aerial photography and existing dieback mapping. At each site two survey plots (100 m x 100 m) were established, one extending from the disease front in dieback infested vegetation, the other in uninfested vegetation. Sites were surveyed for 10 minutes each month for eight months. Bird species observed or heard were recorded, together with information on the vegetation location or flight location. Five of the sites were also surveyed for ground-dwelling vertebrates using pit-fall traps, aluminium traps and funnel traps in late March 2009 (Swinburn and Sonneman 2009).

Preliminary results showed that mean species richness was lower ($P = 0.005$) in dieback infested sites than uninfested. However, the evidence for differences in bird density was inconclusive and warrants further analyses. Some species exhibited clear trends in their use of habitats. Tawny-crowned honeyeaters preferred more open habitats presented by dieback affected sites (Figure 9.7a), whereas the western spinebill, brown honeyeater and silvereye were more prevalent in uninfested sites (Figure 9.7b–d).

In dieback affected sites the percentage of birds observed was greater in ground and lower-storey vegetation. This indicates that birds may either utilise habitat strata differently and/or different species utilise this altered habitat.

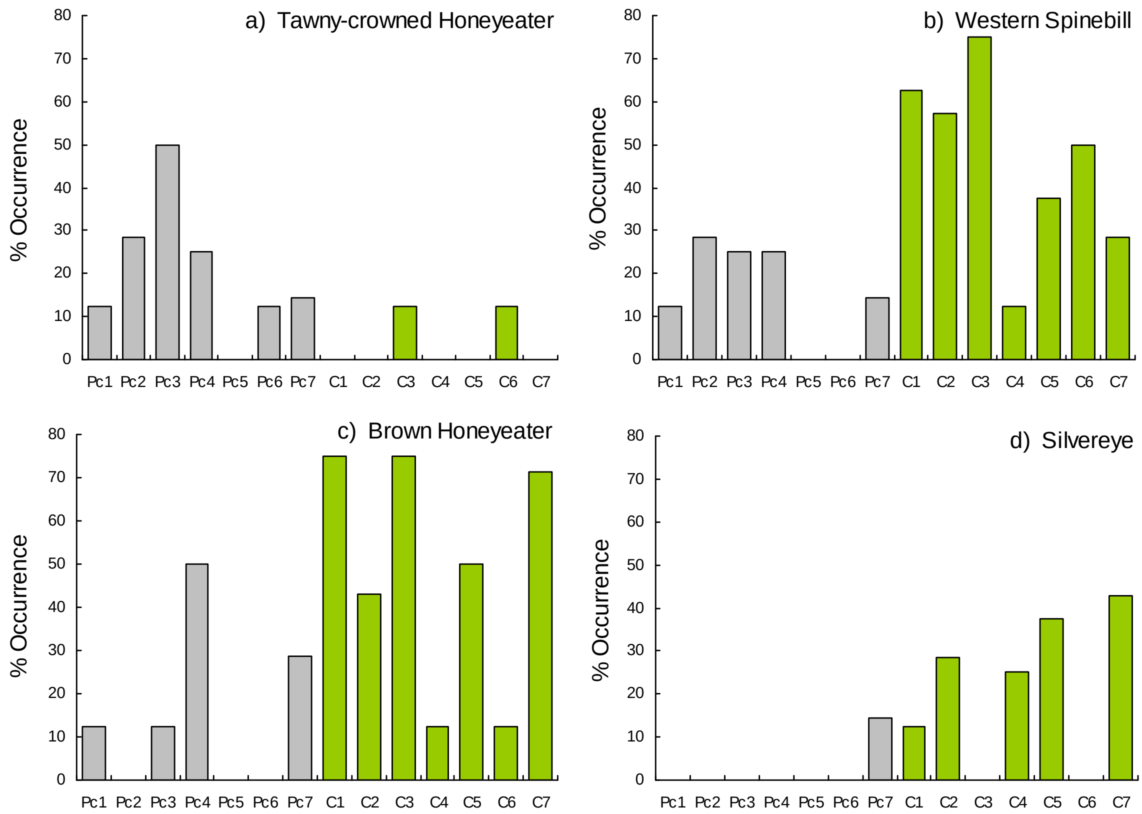



Figure 9.7: Occurrence of a) tawny-crowned honeyeaters, b) western spinebill, c) brown honeyeater and d) silvereye in dieback-affected sites (grey bars) and unaffected sites (green bars).

These preliminary findings indicate that there are impacts on the avifauna and floristics from *Phytophthora* dieback infestation. Plant species richness and canopy cover are lower in infested sites. The overstorey of transition sites at the dieback front share a greater similarity to that of infested sites, due to the collapsing overstorey of *P. cinnamomi* susceptible species such as *Banksia attenuata* and *B. menziesii*. The mid-storey and understorey of transition sites, however, have yet to experience the full effects of species loss as a result of the direct and indirect impacts of *P. cinnamomi*, sharing a greater similarity in species richness to uninfested sites than the infested sites. These preliminary analyses also indicate that species richness and density of the avifauna are also lower in infested habitats compared to uninfested habitat.

# Discussion

Although the impact of *Phytophthora cinnamomi* on flora, fauna and ecosystems has been identified as a major threatening process (DEC 2009), information on its distribution and impacts is limited. The aims of this chapter and the associated studies were to collate and review information on the impacts of *Phytophthora cinnamomi* infestation on biodiversity, and to address knowledge gaps in the occurrence and distribution of the pathogen on the GSS study area, the impacts on the biodiversity values and functional processes of the area, and outline current management regimes.

The review confirmed that *P. cinnamomi* is widely distributed in *Banksia* woodlands of the Swan Coastal Plain. However, the occurrence, population dynamics and impact of the pathogen in the community is currently poorly understood. Information on the distribution of the pathogen in the GSS study area is limited due to lack of mapping and there have been few studies on rates of spread in the area over time. Information on the effects on biodiversity has been limited to flora (susceptible species and threatened ecological communities) on the Swan Coastal Plain and there is no information on effects on fauna.

A number of knowledge gaps were addressed, and further information was obtained as a result of the assessments and projects conducted over the two-year period. Analyses of the occurrence and distribution of the pathogen on the study area established that 20 747 ha (10 %) of the area is infested with *P. cinnamomi* and that the pathogen occurs across all land uses, ranging from small urban remnants to large areas in the conservation estate. Ninety-four percent of the infested area is on the Bassendean Dune system with only minor areas on the Spearwood system and Pinjarra Plain. Our confidence in the identification of infested areas is classified as low to medium because of the limited amount of operational interpretation mapping that has been undertaken.

Analyses of data and field assessments have advanced our understanding of the impacts on the biodiversity values on the Gnangara groundwater system. Information on the susceptibility of plant species to *P. cinnamomi* was available for only 240 of the 1337 species that are known to occur in the GSS study area, and 53% of these species have been recorded as displaying a level of susceptibility to the pathogen, or to the indirect effects it has on plant communities. Eight of the ten threatened ecological communities located in the GSS study area were identified as having species susceptible to *P. cinnamomi.* Four were ranked as high risk, one at moderate risk, and five as low risk of *P. cinnamomi* impacts. '*Banksia attenuata* woodland over species rich dense shrublands' (community type 20a as described by Gibson *et al.* 1994) is considered at highest risk, based on the high number of susceptible species, and the loss of overstorey species with associated modification to vegetation structure, and indirect impacts on understorey species.

Preliminary results of field assessments of the impacts of *P. cinnamomi* on flora and fauna found that plant species richness and canopy cover are lower in infested sites compared to uninfested habitats, and that bird species richness is lower in infested habitats. Completion of plant species identification and data analyses, together with statistical analyses of the ground dwelling vertebrate and bird survey data, are required to confirm the interpretations of these results.

Although we did not directly assess impacts on functional processes, we did identify a number of remote sensing approaches, using Landsat data, capable of distinguishing *P. cinnamomi* affected areas, and assessing impacts on vegetation cover with time (also see Chapter 3). These included the use of vegetation trend analysis that may be of use in distinguishing poor condition of *P. cinnamomi* affected areas. Classification of the area using thresholds on the Projected Foliage Cover index was also fruitful.

These methods provide the opportunity for further investigations on the impacts of the pathogen on fauna habitats and vegetation community condition. In addition the methods may provide opportunities to assess ecological community function, as alterations to overstorey canopy structure have direct effects on variables such as availability of light, microclimates, litter levels and nutrients. Ongoing monitoring of vegetation to determine the vegetation community's capacity to recover or undergo secondary succession or the success of management or rehabilitation efforts may also be possible with these methods.

Changes in the rate of spread of the pathogen over a 55-year period have provided important information its dynamics on the Gnangara system. The declining rate of spread may be related to declining rainfall, or variables such as topography. Future research should investigate the relationship between the rate of spread and slope and also other landscape and topographical variables such as aspect and soil type.

Strategic planning for management of *P. cinnamomi* in the GSS study area is currently deficient. This is expected to be addressed in the strategic planning for *P. cinnamomi* in the Perth and Avon natural NRM regions which is currently being developed (Project Dieback in prep.).

A combination of interpretation of orthophotos and Landsat trend information successfully distinguished between areas infested with *P. cinnamomi* and those that were not. Mapped boundaries of infested areas identified by remote sensing and by ground surveys were not always perfectly aligned, possibly due to the fact that ground based methods can detect the death of all susceptible species, whereas orthophoto interpretation relies on the death of overstorey species. Nevertheless, the study has provided strong evidence that the methodology can be used to identify *P. cinnamomi* infestation at a landscape scale. On-ground surveys may subsequently be targeted to those areas requiring more detailed information on boundaries and management intervention.

## Recommendations

In order to manage and mitigate the effects of *P. cinnamomi* upon flora, vegetation and fauna within the GSS study area , it is recommended that

- data analyses be completed for projects undertaken on the impacts on floristics, ground dwelling vertebrates and bird
- potential changes in the abundance and cover of susceptible plant species in threatened ecological communities be assessed
- a landscape predictive model be developed for future spread of the pathogen in the GSS study area using relationships between the rate of spread and variables such as slope, aspect and soil type
- DEC-GSS contributes to the strategic plan *Managing Phytophthora cinnamomi for biodiversity conservation in the Perth and Avon NRM regions* currently being developed by Project Dieback
- a framework be developed for assessing the risk of *P. cinnamomi* to threatened species, ecological communities and geographic areas, and for ranking them as the basis for setting management priorities (see Chapter 11)

- priorities be determined for implementing management actions using risk and cost-benefit analyses. (see Chapter 11).
- management actions, such as hygiene, quarantine measures, track closures, and application of phosphite, be evaluated for use in threatened and pristine communities (see Chapter 11)
- remote sensing approaches that can identify vegetation that appears to have been affected by dieback be further developed – for example, the utilisation of high resolution data such as Urban Monitor
- long-term impacts on flora and fauna in the GSS study area be investigated.

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Gnangara Sustainability Strategy

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Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter 10: Potential Impacts of Climate Change on Biodiversity

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# 10. POTENTIAL IMPACTS OF CLIMATE CHANGE ON BIODIVERSITY

## Key points

- One of the primary global threats to biodiversity and ecosystem function is climate change. The current rate of climate change is more severe than previous changes in climate, and it is occurring in already compromised ecosystems.
- Climatic changes in Australia are already apparent. South-west Western Australia has had a significant decline in rainfall since 1970, predominantly associated with a sudden decrease in winter rainfall. Future predictions for south-west Western Australia include increases in average temperatures and decreases in average annual rainfall.
- The degree and type of impact of climate change on biodiversity is likely to be variable. Climate change is predicted to have an effect on the geographic range of species, with more generalist species (those with less specialised requirements and a wider geographic range) more likely to adapt to climate change than restricted, specialist species. Changes in phenology have also been identified in a variety of species in response to climate change, including timing of migration, hibernation, breeding seasonality and flowering phenology.
- The impact of climate change on particular species will depend upon their resistance (the ability of a species to withstand an environmental perturbation) and resilience (the ability of a species to recover from an environmental change).
- Mediterranean biomes, such as south-west Western Australia, have been identified as being at particular risk from declining rainfall and extreme weather events. Groundwater-dependent ecosystems, such as those found in the GSS study area, are threatened by reduced rainfall. The predicted continuing decline in rainfall is also likely to alter the structure and composition of *Banksia* woodlands. Changes in *Banksia* and *Melaleuca* species and Myrtaceous shrub species on the Swan Coastal Plain have occurred due to reduced rainfall in the past 30 years.
- There may be changes to other environmental factors, such as pollination syndromes, salinity, hydrology, and flowering season, which may, in turn, affect species that rely on the nectar of *Banksia* flowers (e.g. honey possums).

- Fauna are also expected to be affected by climate change, through altered species composition, community structure and ecosystem functions. Fauna that are dependent upon rainfall to facilitate breeding are particularly susceptible (e.g. frogs and the western swamp tortoise). Reptiles that are litter dependent or litter associated will also be affected by declining rainfall, due to decreased leaf shed and resultant lack of leaf litter.
- A reduction in wetland habitat has contributed to the decline of wetland birds in the GSS study area, and loss of wetlands should affect the wetland-dependent water rat. Mammal species decline is correlated with lower rainfall, and mammals in the critical weight range may be at high risk from declining rainfall. Declining rainfall may compromise the flowering potential of *Banksia*, subsequently affecting honey possum populations.
- Little is known about the potential impacts of climate change on specific taxa in the GSS study area, and further research is required to obtain baseline information for more accurate predictive modelling.

# Introduction

Climate change is acknowledged as one of the primary global threats to biodiversity and ecosystem function (Brook *et al.* 2008; 2008). Many studies have identified the impacts of climate change, both negative and positive, across species, communities and ecosystems worldwide (Parmesan 2006; Walther *et al.* 2002), and some researchers have predicted high levels of species extinction and the breakdown of community and ecosystem interactions in natural systems around the globe (Thomas *et al.* 2004). Contemporary climatic changes result primarily from anthropogenically induced increases in greenhouse gases in the atmosphere, including $CO_2$, $N_2O$ and $CH_4$. The most important greenhouse gas is $CO_2$, produced as a result of various human activities, such as industry, transportation, agriculture and forest clearance (IPCC 2007).

Although Earth has experienced many climatic changes over geological time, including previous periods of warming, current climate change differs in two important ways from past changes. Firstly, the current rate of climate change is considered to be unparalleled in the past 10 000 years. Indeed, the latest predictions from the Intergovernmental Panel on

Climate Change (IPCC 2007) indicate that global mean surface air temperatures will continue to increase throughout the next century. Furthermore, extreme high temperature and rainfall events are also predicted to become more common, while snow cover and sea ice are expected to decrease – contributing to rising sea levels. Secondly, many of Earth's ecosystems are already under pressure from other human impacts, including land clearance, habitat fragmentation, and introduced diseases and predators. These factors have reduced population sizes of some species, disrupted ecosystem function and resulted in small and isolated populations with reduced genetic diversity. Small populations are also more susceptible to stochastic events such as wildfires (Brook *et al.* 2008). The combination of these additional disturbances may make species more susceptible to the impacts of climate change.

Australia has already seen the average surface air temperature increase by more than 0.7 °C over the past 100 years; most of this increase has occurred since the 1950s (Howden 2002). This warming has come in concert with significant declines in precipitation, particularly on the east and west coasts of the continent, which is predicted to continue. These declines are expected to be especially marked in coastal regions of Western Australia, including the Swan Coastal Plain and the GSS study area (IOCI 2002; Preston and Jones 2006). The impacts of climate change on biodiversity are now apparent in ecosystems around the globe. Impacts already noted in Australian species and ecosystems include shifts in geographic range of some species (Ling 2008), apparent decline or extinction of some local populations of mammals (S Williams, pers. comm. 2009), and changes to vegetation composition (Hughes 2003).

The aims of this chapter are therefore to collate and review current literature on the apparent and predicted impacts of climate change on biodiversity, and specifically to examine the potential impacts of climate change on biodiversity assets in the GSS study area. These aims were addressed by:

- reviewing and identifying current climate change predictions for Australia, Western Australia and the Swan Coastal Plain region, specifically in terms of predicted future rainfall and temperatures
- reviewing the literature on impacts of climate change on species and ecosystems, with a focus on predicted impacts on wetland systems, and regions with Mediterranean type climates

- identifying the potential impacts of climate change on biodiversity in Western Australia, focussing on the GSS study area and, in particular, the potential impacts of declining rainfall and groundwater.

At the start of this project, there was very little information predicting species-specific responses to climate change in the GSS study area (or Swan Coastal Plain). To address this knowledge gap, we have used existing data to model potential predictions of the impacts of reduced rainfall on specific taxa in the GSS study area. These models are outlined below.

**1) Assessing the vulnerability of frogs in the GSS to declining groundwater and rainfall as a result of climate change**

The objectives were to identify frog species in the GSS study area most likely to be detrimentally affected by predicted declines in rainfall and groundwater. A new vulnerability model was developed which ranked frogs in terms of their vulnerability, based on key life history traits including larval length, dependency on predictable rainfall or permanent ponds, and potential for dispersal.

**2) Identifying extinction-prone mammals in the GSS under an increasingly arid climate**

This project involved evaluating the current extinction risk of mammals in the GSS study area based on body size (a predictor of extinction risk in Australian mammals), which is thought to interact with low rainfall to increase extinction risk.

**3) Potential consequences of declining rainfall on the abundance and reproductive success of the honey possum, *Tarsipes rostratus***

The aims of this project were to model the potential effects of various levels of reduced rainfall on the abundance of the honey possum, using published data as a baseline. We also examined how a shift in flowering phenology of key *Banksia* food species under climate change could influence reproductive success in this key GSS species.

# Climate change observations and predictions

## Global and national observations

The most extensive observations and predictions for global climate change come from the Intergovernmental Panel on Climate Change (IPCC) assessment reports; the most recent IPCC assessment report, published in 2007, affirms that 'warming of the climate system is unequivocal'. As this statement suggests, observations are that the global mean temperature has already increased and is predicted to continue to do so. In the past 100 years, the global mean temperature has increased by 0.74 °C (IPCC 2007) and temperature increases are widespread. Congruent with global predictions, the mean average temperature in Australia has increased by 0.9 °C since 1950, albeit with significant variation across regions (CAWCR 2007).

Precipitation patterns have also been observed, with increases seen in some regions in the northern hemisphere, but declines noted in some areas in the southern hemispheres. Changes in rainfall patterns in Australia also show geographic differences, with most of eastern and south-western Australia experiencing substantial rainfall declines since 1950, while north-west regions have become wetter. Extreme daily rainfall intensity and frequency has increased in north-western and central regions, but has decreased in the south-east and the south-west. Globally, the area affected by drought has increased since 1970, while the number of cold days, cold nights and frosts has decreased over land areas (IPCC 2007).

Precipitation in Australia is expected to decrease in most regions, with up to a 20% increase in drought months over most of Australia by 2030. Coastal drying is predicted to affect all urban centres except Brisbane (Pitman and Perkins 2008). Significant increases in inundation in coastal regions due to higher mean sea levels and more intense weather systems are also predicted. Finally, it is also predicted that there will be an increase of up to 70% in the number of days with very high or extreme fire danger ratings by 2050, while fire season is also likely to lengthen, meaning that the periods suitable for prescribed burns will change (Lucas *et al.* 2007; Pitman *et al.* 2007).

# Impacts of climate change on species, communities and ecosystems

That contemporary climate change is affecting species and ecosystems worldwide is now a certainty. In the past decade, numerous studies have demonstrated the impacts of climatic changes in a variety of ecosystems and communities around the world (Hughes 2000; Parmesan & Yohe 2003; Walther *et al.* 2002). However, the degree and type of climatic change, and also the impact of climate change on biodiversity, are likely to differ both on a large geographic scale and also on smaller scales across ecological gradients such as altitude and climate (Thomas *et al.* 2004). For example, owing to rising sea levels and more frequent flooding events, species inhabiting coastal wetland habitats are expected to be at extremely high risk from climate change. The IPCC predicts that within the next 70 years we could see the loss of more than 20% of the earth's coastal wetlands. Predictions also suggest that terrestrial areas will warm more than the oceans, indicating that terrestrial ecosystems and species will face greater absolute warming (IPCC 2007).

The evidence for climate induced range shifts is now confirmed by numerous studies documenting a multitude of instances where species have shifted their geographic range in response to changing climate (Parmesan 2006). The most commonly reported pattern is for species to shift their distribution towards the poles so as to follow changing climatic conditions. On average species around the globe have moved 6.1 km per decade towards the poles (Parmesan and Yohe 2003). Other species have moved upwards in altitude, with cooler adapted species retracting their lower altitude margins and becoming more restricted (e.g. Wilson *et al.* 2005). However, not all documented range shifts represent a contraction in range and a number of species have expanded their geographic range in response to a warming climate (Battisti *et al.* 2006). In general, it is predicted that more generalist species (those with a wide ecological niche) may be able to expand their geographic range under climate change, while restricted specialist species are likely to undergo range contractions (Isaac *et al.* 2009).

There have been changes to the geographic range of a variety of Australian species (Ling 2008). Further range shifts have also been predicted from bioclimatic models for other Australian species including a decline in the distribution of four large macropod species

(Ritchie and Bolitho 2008) and a number of butterfly species (Beaumont and Hughes 2002).

Changes in phenology have also been identified in a variety of species in response to the changing climate, including timing of migration and hibernation and breeding seasonality (reviewed by Crick 2003; Crick *et al.* 1997; Inouye *et al.* 2000; Reale *et al.* 2003). In some cases, these changes in phenology may have little or no impact on the species concerned. However, they can create the potential for trophic mismatch which occurs when the response of a prey species to climatic changes differs, either temporally or spatially, to that of the predator species (Figure 10.1). For example, mistiming of avian reproduction and caterpillar biomass appears to be relatively widespread and has resulted in failed breeding and population crashes in many bird species (reviewed in Visser and Both 2005). More recent research indicates that mismatch may not be limited to predator–prey interactions; Post & Forchammer (2008) demonstrate similar effects in a plant–herbivore system and Hegland *et al.* (2009) hypothesise that if the onset of flowering advances earlier than emergence of the pollinator, climate change could have implications for both plant and pollinator abundance and distribution. For example, as the climate has changed over time breeding of a predator (for instance, a migratory bird) is occurring earlier in the year.

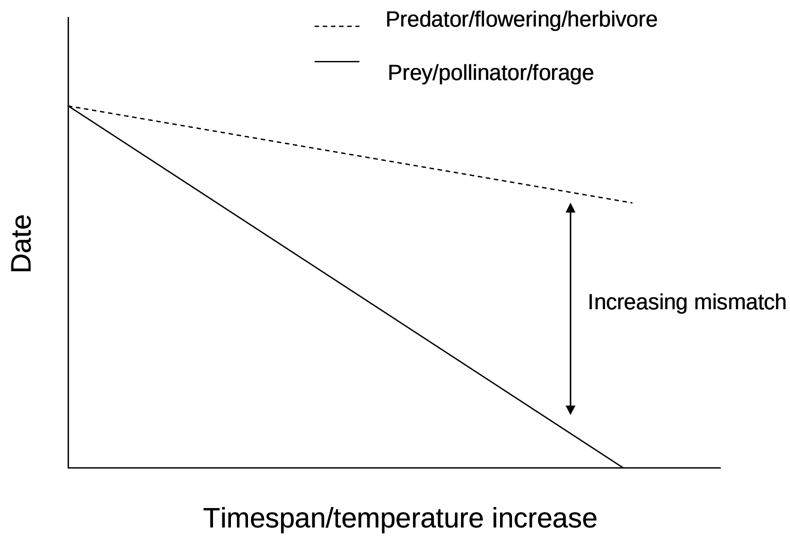


Figure 10.1: A graphical representation demonstrating how different responses to climate change can result in the decoupling of pairwise ecological relationships.

Changes to phenology in Australian species are also similar to those recorded for other species around the globe, with Australian migratory birds demonstrating changes in arrival and departure dates that are analogous to those reported elsewhere (Beaumont *et al.* 2006) and trophic mismatches have also been documented in Australian predator–prey systems (Peck *et al.* 2004). It has also been demonstrated that increased temperatures can affect flowering phenology in Australian temperate grassland plants by reducing time to first flowering (Hovenden *et al.* 2008), implying that predicted mismatches between plants and pollinators could occur if the response of pollinators is asynchronous with that of the plants.

Although climate change is predicted to result in the decline, and even extinction, of many species (Malcolm *et al.* 2006; Thomas *et al.* 2004; Williams *et al.* 2003), there is a growing literature suggesting that some species may be able to adapt through a variety of mechanisms. For example, plasticity in niche specialisation, behavioural or microhabitat adjustments, and physiological tolerance ranges can all make a species less vulnerable to climatic changes. The evidence for ecological and behavioural adaptation to climate change is currently scant. However, research in Australia has demonstrated that some species, including lizards and mammals, can use microhabitat refugia, such as tree hollows or rock crevices, as thermal buffers to shelter from extreme temperatures (Doody *et al.* 2006; Isaac *et al.* 2008). Species could also shift their distribution to keep up with preferred climate and habitat variables, or alter their daily activity patterns in order to avoid temperature extremes (Williams *et al.* 2008).

Although it is commonly assumed that evolution by natural selection does not occur at rapid enough rates to keep up with contemporary anthropogenic pressures, significant rapid evolutionary changes in response to global change are now becoming documented in the scientific literature. For example, Grant and Grant (1993) documented rapid selection for smaller beak sizes under changing food conditions in Darwin's finches following El Niño events. Similarly, increasing temperatures over the past eighteen years have resulted in an increase in body size in common lizards (*Lacerta vivipara*) in France (Chamaille-Jammes *et al.* 2006).

The ability of a species to adapt to climate change, either behaviourally and/or genetically, depends upon its inherent level of plasticity, which may be influenced by factors such as phylogenetic constraints, genetic variability and behavioural flexibility (Williams *et al.* 2008). Generalist species, with less specialised requirements and a wider geographic range, will be more likely to adapt to climate change, while geographically restricted, ecologically specialised species – such as regional endemics – may be unable to adapt in time (Williams *et al.* 2008).

Recent work in Australia has also demonstrated the impact of extreme weather events, such as abnormally hot summer days (Welbergen *et al.* 2008). This study documented a mass mortality event (> 3500 individuals) in flying foxes during a 42 °C summer day in New South Wales (Welbergen *et al.* 2008). The effects of the extreme temperatures appeared to differ both between species and between the sexes, probably due to inherent differences in life history and behaviour.

## Climate change on the Swan Coastal Plain and predicted impacts on biodiversity

### Rainfall and groundwater

South-west Western Australia is a much studied region because of a significant decline in rainfall since 1970 (IOCI 2002); Figure 10.2). Studies show that this decline is predominantly associated with a sudden decrease in winter rainfall, which was not gradual, but more indicative of switching to a new rainfall regime (IOCI 2002). Decreased precipitation in the region has been attributed to a variety of mechanisms including global warming, natural variability, ocean warming, and land-cover change (Pitman and Perkins 2008).

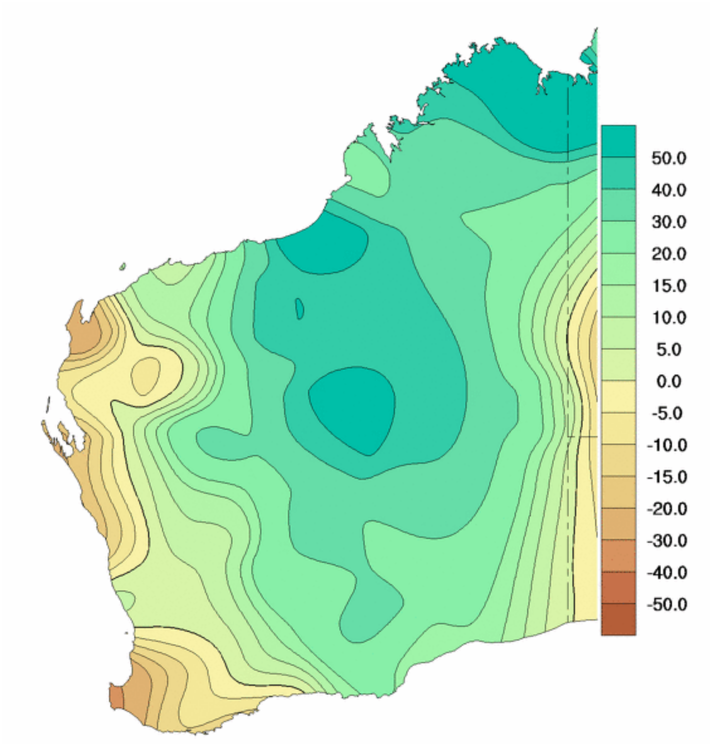


Figure 10.2: Trends in annual total rainfall, 1970–2008 (mm/10 years) in Western Australia (accessed 26/03/09 – http://www.bom.gov.au).

It has been established that this pattern of decreased precipitation is the primary cause of decreased recharge to aquifers, observed declines in groundwater and loss of wetlands in the GSS region since the mid 1970s (Government of Western Australia 2007; Vogwill *et al.* 2008). For hydrological systems in the region, the impacts of reduced rainfall have been exacerbated by increases in winter temperatures, which have resulted in increased evapotranspiration and human water consumption (IOCI 2002).

Various modelling studies concur that the main effect of climate change on the region will be decline in rainfall. The observed decline in winter rainfall is predicted to continue, but declines in spring and summer rainfalls are also predicted for the future. Climate modelling indicates that, compared with average recorded rainfalls to 1990, rainfalls will decline by as much as 20% by 2030 and 60% by 2070 (Jones and Preston 2006; Preston and Jones 2006). Additionally, models predict an increase in rain-free days in the region, particularly in winter and spring and an increase in drought months of up to 80% by 2070 (Pitman and Perkins 2008). It has been predicted that the south-west of Western Australia is likely to undergo the most intense drying of any region in Australia by 2100 (Pitman and Perkins 2008).

In the GSS study area, patterns of reduced rainfall have been documented (Chapter 1). Based on long-term average annual rainfall (Chapter 1), since 1976 there has been a 9% reduction in rainfall and since 1997, a 13% declined in rainfall. It is very likely that the current and predicted future declines in rainfall will significantly decrease streamflow and groundwater recharge and average streamflow into the Swan River from its tributaries will decline by 24% (Evans and Schreider 2002; Jones and Preston 2006; Preston and Jones 2006; Ryan and Hope 2006; Sadler 2007).

## Temperature

The south-west of Western Australia has experienced temperature increases that are generally comparable to many other Australian regions. Increases are particularly marked in winter and autumn and there has been an average annual temperature increase of approximately 1.5 °C (CAWCR 2007); Figure 10.3). Predictions are that maximum summer temperatures will rise by approximately 5% from the current mean in Western Australia and that there will be an increase in the mean temperature across all seasons of at least 1 °C (Pitman and Perkins 2008). Most models also predict an increase in the number of extremely hot days, over 35 °C, from a current 28 days to 35 days in Perth, based on a mid-emissions scenario by 2030 (CAWCR 2007).

The projected increases in temperature in all seasons are likely to exacerbate the already stressed groundwater systems in the south-west Western Australia region. Firstly, increased warming throughout the year will result in increased evapotranspiration in catchments and, as a result, reduced inflow. Secondly, higher temperatures and an increase in the number of days above 35 °C will also cause increased demand for water.

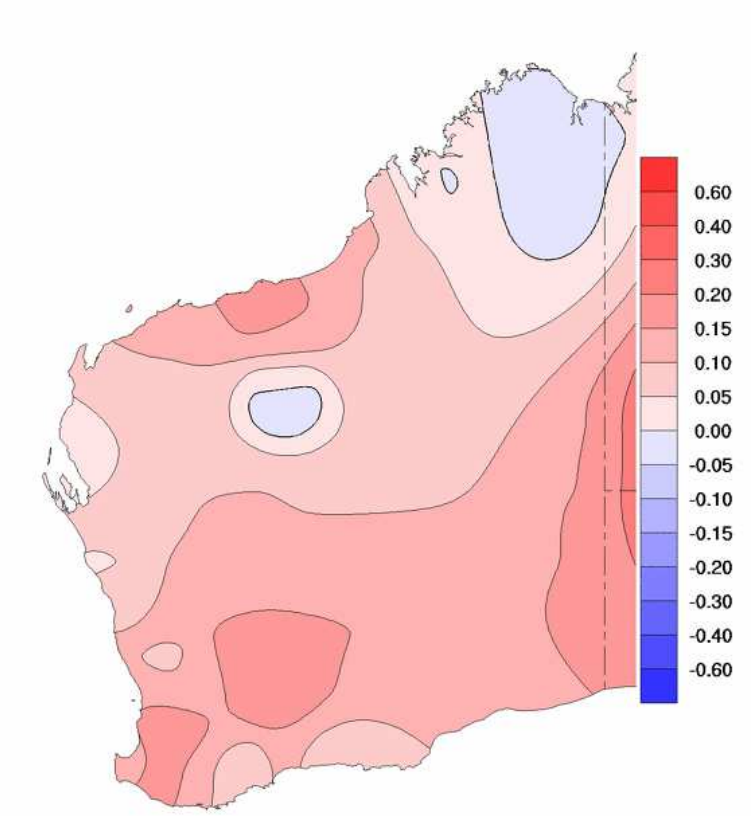


Figure 10.3: Trends in mean annual temperature, 1970–2008 (°C/10 years) in Western Australia (accessed 26/03/09 – http://www.bom.gov.au).

## Floristics and vegetation

Mediterranean biomes, including south-west Australia, are well recognised for supporting high levels of plant richness and endemism (Mittermeier *et al.* 2004). However, these ecosystems will be at particularly high risk from the impacts of climate change because of predicted decreases in rainfall, extreme weather events and sea level rises (IPCC 2007). The Mediterranean biome is projected to experience the greatest proportional loss in biodiversity by 2100 owing to its sensitivity to climate and land use change (Sala *et al.* 2000). The effects of climate change on terrestrial plant ecosystems are predicted to include modified biogeochemical cycles and changes in species composition and abundance, which will lead to changes in community structure (Mouillot *et al.* 2002).

Groundwater-dependent ecosystems, such as those found on the Swan Coastal Plain and in the GSS study area, require a predictable and consistent input of groundwater to maintain their composition and functioning (Murray *et al.* 2003). A decline in groundwater, or change in the timing or distribution of groundwater, may cause changes to associated floristic assemblages and communities. Continuing and increasing declines in rainfall and groundwater are likely to affect wetland species that are least drought tolerant. For example, *Banksia littoralis* is the only *Banksia* confined in the GSS study area to wetland fringes and low lying depressions and is thus reliant on groundwater all year round (Groom *et al.* 2000a). Thus, populations of this species, as well as other wetland tree species (e.g. *E. rudis, Melaleuca preissiana)* should decrease under drought conditions because of increasing adult and seed mortality. A broad synthesis of the impacts of reduced rainfall on groundwater-dependent ecosystems is provided in Chapter 5. Here, we explore some specific species responses to reduced rainfall in the GSS study area.

Although the tree and shrub species of the *Banksia* woodlands on the Swan Coastal Plain can tolerate soil water deficits during summer (Groom *et al.* 2000a), research indicates that terrestrial vegetation has already been affected by the marked climatic changes of the past 30 years. Groom *et al.* (2001; 2000b) investigated changes in the vigour and distribution of *Banksia* and *Melaleuca* species on the Swan coastal plain during a 30-year period and found that the two codominant overstorey species, *B. attenuata* and *B. menziesii*, displayed a reduction in foliage condition. *B. attenuata* also showed a shift in distribution, expanding its range downslope over time. Drought intolerant species such as *B. littoralis* and *Melaleuca preissiana* demonstrated the greatest loss of vigour, and *B. littoralis* was replaced in some areas by the more drought tolerant *B. prionotes*. These changes in vigour and distribution were attributed to declining groundwater in the region (see Chapter 5), caused by reduced rainfall and increased water abstraction.

Similar results have been found in Myrtaceous shrub species typical of the Swan Coastal Plain. Species which were classified as tolerant of excessive wetness, including *Astartea fascicularis, Hypocalymma angustifolium, Pericalymma ellipticum* and *Regelia ciliata*, showed the greatest reduction in population size in response to declining groundwater and rainfall (Groom *et al.* 2000b). Species which are commonly associated with dry areas, and are deep rooted, including *Melaleuca scabra* and *Scholtzia involucrata*, appeared more tolerant to declines in groundwater. However, a long-term decline of the shrub *Eremaea pauciflora* has been attributed to groundwater levels falling below the reach (6.4 m) of root systems (6.4 m, Dodd and Bell 1993; Groom *et al.* 2000b).

In an extensive review, Lamont *et al*. (2007) also identify non-sprouting species of *Banksia* as being particularly susceptible to the effects of climate change. The continuing decline in rainfall predicted for the future is likely to lead to further modifications to the structure and composition of threatened *Banksia* woodlands (Groom *et al.* 2000a). The inability to access soil water at depth causes seedlings of deep rooted species to die during the next summer period, reducing recruitment into the adult population. Drought intolerant species such as *B. ilicifolia* and *B. littoralis* may also be replaced by species such as *B. prionotes* which are less dependent on groundwater (Groom 2004).

Several recent modelling studies support the prediction that south-west Australia, and the Swan Coastal Plain region, will experience further declines and extinctions in terrestrial vegetation. For example, Malcolm *et al.* (2006) identified the vegetation of south-west Australia as being particularly vulnerable to climate change and in some scenarios predicted extinctions of more than 2000 plant species.

Fitzpatrick *et al.* (2008) also modelled the impacts of future climate change on *Banksia* woodland in Western Australia and found that in all climate and migration scenarios, 66% of *Banksia* species were predicted to decline by 2080. Under a mid-severity climate model, they predict the extinction of seven species and a decline in range of a further 80 species; only 21 species were projected to increase their range. Predicted increases and decreases for the key species found on the Swan Coastal Plain and in the GSS study area can be seen in Table 10.1.

This table shows modelled range loss categories (% of current range) for GSS *Banksia* species under three climate change scenarios. These were:

- low-severity – assumes atmospheric $CO_2$ of 520 ppm, a mean temperature increase across south-west Western Australia of 1.3 °C, and a decrease of 5% in annual rainfall
- mid-severity –615 ppm, 1.9 °C, and 12%
- high severity model –815 ppm. 4.2 °C and 40%.

The results shown in the table assume a full migration rate: a best case assumption assuming unlimited migration capacity.

The authors conclude that the effects of climate change on the flora of south-west Western Australia will be severe due to increasing drought.

Table 10.1: Modelled range loss categories (% of current range) for some *Banksia* species in the GSS under three climate change scenarios (after Fitzpatrick *et al.* 2008). The results shown here assume a full-migration rate: a best case assumption assuming unlimited migration capacity.

| Species | Low-severity[1] (%) | Mid-severity[2] (%) | High-severity[3] (%) |
|---|---|---|---|
| *B. attenuata* | 0 | 0 | 80 |
| *B. grandis* | 0 | 30 | 80 |
| *B. ilicifolia* | + | 0 | 50 |
| *B. littoralis* | 0 | 30 | 80 |
| *B. menzeissi* | 0 | 0 | 0 |
| *B. prionotes* | 30 | + | 50 |

[1] assumes atmospheric $CO_2$ of 520 ppm, a mean temperature increase of 1.3 °C, and a precipitation change of -5 % in annual rainfall.

[2] assumes atmospheric $CO_2$ of 615 ppm, a mean temperature increase of 1.9 °C, and a precipitation change of -12 % in annual rainfall.

[3] assumes atmospheric $CO_2$ of 815 ppm, a mean temperature increase of 4.2 °C, and a precipitation change of -40 % in annual rainfall.

+ indicates the potential to increase current range

Other landscape scale, threatening factors for endangered Australian flora are likely to interact with, or be exacerbated by, climate change (DEH 2004). These include pollinator disruption or loss, changes in salinity, hydrology and extreme weather events. Meta-analysis has also indicated that pollinator loss and salinity are key issues for endangered plants (Burgman *et al.* 2007). The current or potential impacts of climate change on plant-pollinator relationships in the Swan Coastal Plain and GSS study area are, at present, unclear. Most species in the *Banksia* woodland typical of the GSS study area are pollinated by honeyeaters (Lamont *et al.* 2007; Yates *et al.* 2007) and/or small mammals such as the honey possum *Tarsipes rostratus* (Wooller *et al.* 1983). Climate change has the potential to disrupt plant–pollinator relationships through phenological mismatch, and/or result in the loss of a pollinator due to a range shift or local extinction. Potential disruption of pollinator–plant mutualisms has obvious implications for plant demography (Broadhurst and Young 2007) particularly for Proteaceous species. Any independent effect of climate change on pollinators, particularly honeyeaters and honey possums, could be detrimental to the many species of *Banksia* and *Grevillea* in the GSS study area.

Dominant species of *Banksia* in the GSS study area exhibit a sequence of flowering seasons, thus providing a year round supply of nectar for nectar-feeding species (Whelan and Burbidge 1980). Research conducted elsewhere indicates that climate change has resulted in the advancement of flowering in many plant species (Lu *et al.* 2006). However, the magnitude of response can differ between species depending on their underlying ecology and phenology. For example, Lu *et al.* (2006) demonstrate that, because the climate warming has been more marked in winter and early spring in China, the dates in early flowering species have advanced more rapidly than those of species which flower in the late spring and summer.

In south-west Western Australia, the magnitude of temperature increase due to climate change has been most marked in winter, thus we might predict that flowering will become relatively earlier in *Banksia* species, which flower during the winter months, compared to species which flower at other times during the year. Any asynchrony in the response to climate change among *Banksia* in the GSS study area could result in disruption to the currently continuous supply of nectar and pollen (see Figure 10.4 a, b). Figure 10.4a shows the flowering pattern of five key species of *Banksia* found in the GSS study area, based on data of Whelan and Burbridge (1980). It demonstrates that the sequential flowering of the species provides a year round supply of nectar. Figure 10.4b shows a speculative scenario under future climatic conditions, where the winter flowering species (*B. littoralis* and *B. menziesii*) have advanced their flowering dates to a greater degree than the spring and summer flowering species.

In addition, Wooller *et al.* (1998) suggest that nectar production is likely to be predominantly dependent upon rainfall, as nectar contains 70% to 80% water and so the predicted continuing decline in rainfall in the region is also likely to result in a decrease in nectar availability. Any change to the supply and/or quantity of nectar supply, along with the predicted decline in population size of many species of *Banksia*, will have consequences for nectivorous species in the region which rely on *Banksia*, such as the honey possum (see DEC projects section of this chapter for case study).

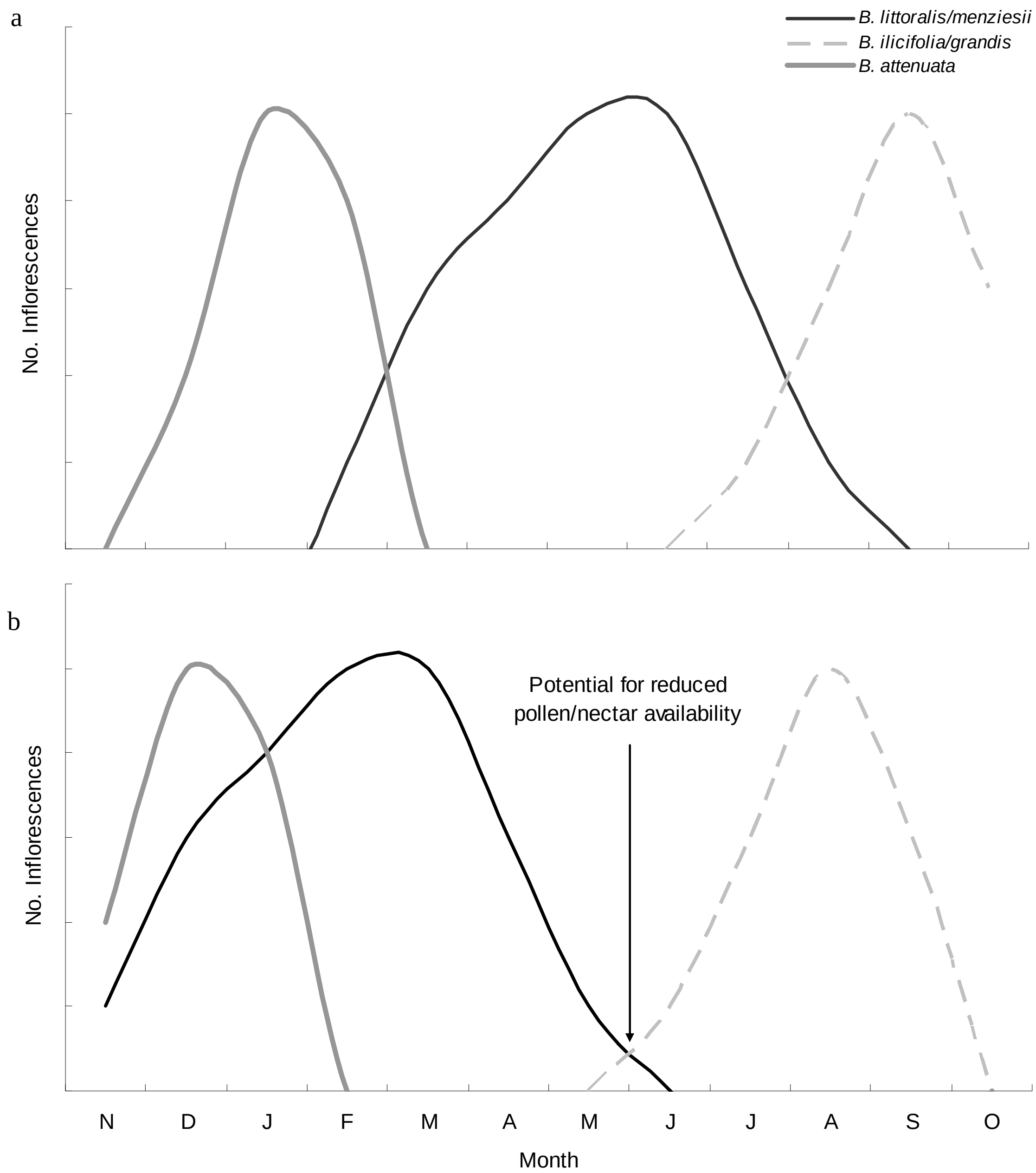


Figure 10.4: a) The flowering pattern of five species of *Banksia* found in the GSS study area, based on data of Whelan and Burbridge (1980) and b) a speculative scenario under future climatic conditions, where the winter flowering species have advanced their flowering dates more than the spring and summer flowering species.

# Fauna

## *Overview*

Climate changes are expected to have a considerable impact on faunal biodiversity worldwide, altering species composition, community structure and ecosystem function (Walther *et al.* 2002). Estimates from models suggest that fauna extinctions from climate change could be as high as 40% (Thomas *et al.* 2004). IPCC (2007) has identified a number of regions and ecosystems where faunal biodiversity is likely to be at particularly high risk from the impacts of climate change. These regions include coastal areas, wetland and Mediterranean-type ecosystems (IPCC 2007), all of which the GSS study area has. Accordingly, south-west Western Australia may be at considerable risk (IPCC 2007). For example, model outputs show species-area loss of endemic vertebrates of 2.5% to 66.1% in the south-west Western Australian biodiversity hotspot in 100 years (Malcolm *et al.* 2006).

In the GSS study area, many fauna taxa and communities will be adversely affected by climate change. There is a pressing need to determine what species are most at risk by clarifying the ecological and life history factors which influence vulnerability to current climate change. Previous studies have already demonstrated that a number of factors may make species more or less vulnerable to extinction from climate change. Evidence from the palaeontological literature suggests that two general factors act together to reduce extinction risk from environmental changes (McKinney 1997). These are resistance – the ability of a species to withstand an environmental perturbation, and resilience – the ability of a species to recover from an environmental change. Species traits that are likely to confer resistance to environmental change include a wide geographic range, high local abundance and low habitat specialisation (McKinney 1997). A GSS study area example could be the willy wagtail, *Rhipidura leucophrys*, which occurs throughout Australia in high abundances including urban areas and degraded woodlands. Conversely, an example of a species in the region expected to have low resistance is the western swamp tortoise *Pseudomydura umbrina*, which is endemic to the GSS study area, restricted to only a few wetlands and has an estimated extent of occurrence of only 100 – 150 $km^2$. The tortoise also appears to occur at low densities within its range and depends upon rare, fragmented wetland habitats with suitable breeding pools and aestivation sites (Burbidge *et al.* 2008).

While there is limited evidence from the fossil record to indicate which traits promote resilience, it is likely that species with faster generation times and a shorter life history in general will be better able to recover from adverse environmental events (Cardillo 2003). Another trait thought to promote resilience is high dispersal potential, in order for a species to reach suitable new habitat (Knapp *et al.* 2001). Under rapid climate change, a wide environment niche is also likely to promote resilience, which should facilitate the colonisation of habitats with changed or new environmental conditions (Crawford 1997). Recent studies indicate that species that have high resistance also tend to have high resilience. These traits are represented in many successful exotic species, such as rabbits and cane toads. Similarly, species with low resistance often have low resilience and such trait combinations are commonly seen in listed endangered species (Isaac *et al.* 2009).

An example of a species likely to have high resilience in the south-west Western Australia region is the western grey kangaroo (*Macropus fuliginosus*). For their size, kangaroos have relatively high reproductive rates and the ability to disperse long distances over short periods of time. They also have a wide environmental niche, occurring across a variety of habitats and climates. A species expected to have lower resilience in the face of climate change is the Carnaby's black-cockatoo (*Calyptorhynchus latirostris*). Although this species is able to migrate substantial distances, it is restricted to south-west Western Australia and has a number of specific niche requirements, including the presence of suitable tree hollows (*Eucalyptus*) for nesting and native vegetation (kwongan heathlands and *Banksia* woodlands) for foraging, which make many habitats unsuitable (Saunders 1980; Valentine and Stock 2008). These cockatoos generally raise only one offspring to independence each year and are only found in regions with more than 300 mm rainfall per year (Saunders 1986). However, the potential resilience of this species to the impacts of declining rainfall is limited by extensive habitat loss in the breeding and foraging areas.

Endemic species are also expected to be at particularly high risk from climate change. In general, endemic species have evolved in situ under specific regional environmental conditions and so develop specialised adaptations to local climatic and habitat variables (Isaac *et al.* 2009). While these specialisations are expected to increase the persistence of endemic species under a stable environment by increasing their competitive ability, they may also make them prone to extinction if conditions change rapidly (Isaac *et al.* 2009). For example, the froglet *Crinia insignifera* is restricted to the Swan Coastal Plain, has a narrow habitat niche (requires wetland habitats) and is likely to be very susceptible to declining rainfall (Chapter 3). Assessment of resistance and resilience in 163 terrestrial vertebrates in the Australian wet tropics found that endemic species consistently had lower resistance and resilience than more widespread species (Isaac *et al.* 2009) and this is likely to be the case for the regionally endemic species present in the GSS study area, and may be a potential area for further study.

The risk of extinction for already threatened species may also increase (NRMMC 2004), in particular those that now occur in isolated meta-populations due to habitat fragmentation or predation pressure. Climate change models predict that the current range of many species will decline and some habitats will become more isolated across the landscape. This may prevent species from being able to shift their geographic range in order to follow preferred environmental and ecological conditions. Conversely, climate change may favour some species, particularly those that are ecological generalists tolerating a wide range of environmental conditions (Isaac and Williams 2007).

There is little research on the groundwater dependency of terrestrial fauna. Characteristics of wetlands, such as hydroperiod, seasonal variations in water levels, presence of other species and water quality are likely to be significant predictors for responses of wetland associated species or communities (Chapters 4 and 5). In Chapter 3, based on the work by Bamford and Bamford (2003), we identified nearly 90 species of vertebrates (excluding fish) that are highly or very highly dependent on groundwater levels. As declining rainfall, coupled with groundwater abstraction, is the primary cause of declining groundwater levels, these species also represents taxa that will be susceptible to declining rainfall. Each of the following sections examines the terrestrial vertebrate fauna groups in turn, identifies the likely threats of climate change to the taxonomic group, and assesses the potential impacts on individual species found in the GSS study area.

### *Frogs*

Most frogs rely on the presence of permanent waterbodies for their survival and reproduction, and so it is likely that this group in particular will be adversely affected by declines in available water as a result of declining rainfall associated with climate change. Higher temperatures associated with climate change are also believed to contribute to the drying out of breeding pools, and as a result to the deaths of tadpoles and eggs. Additionally, as ectotherms, frogs are especially likely to be vulnerable to climate change because their basic physiological functions, including locomotion, growth, and reproductive success are influenced by environmental temperature (Deutsch *et al.* 2008). Research also indicates that an increase in ambient ultraviolet-B radiation, associated with climate warming, can result in lower offspring survival and high levels of deformity in metamorphs of some frog species (Adams *et al.* 2001).

Historically, 13 species of frog are known to have occurred in the GSS study area (Chapter 3, Bamford & Huang 2009). Recent surveys have failed to locate several of these, although these species are likely to only have been infrequent visitors to the area (Bamford and Huang 2009). Nine species have been listed as highly or very highly dependent on groundwater levels (Chapter 3). A number of these species were only found at sites which had surface water present during the winter months. They included *Crinia georgiana*, *C. glauerti*, *Litoria moorei* and *L. adelaidensis* (Bamford and Huang 2009). These apparent permanent water obligates may face local extinctions if declining rainfall and groundwater abstraction results in the loss of winter waterbodies. In addition, *Lymnodynastes dorsalis* is sensitive to reduced hydroperiods due to its long larval period and reliance upon emergent vegetation for calling and egg laying (Bamford and Huang 2009). Species at the northerly edge of their range, including *C. georgiana* and *C. glauerti* are also considered to be at risk if the species undergoes range contraction (Bamford and Huang 2009). Breeding in a number of the frogs in the GSS study area also relies upon a predictable early winter rainfall. For example, *Heleioporus eyrei* constructs breeding burrows on the margin of wetlands that experience early winter rise in water level. The vulnerability of the frogs in the GSS study area to continued declines in rainfall and groundwater is examined in detail in a later section (DEC–GSS project 1).

### ***Reptiles***

Reptiles are ectotherms and thus particularly sensitive to changes in temperature which can affect reproductive success, performance and survival. However, most terrestrial reptiles should be less vulnerable than frogs to changes in water availability and rainfall. Only five of the reptiles found in the GSS study area are significantly associated with wetlands

(Bamford and Bamford 2003), and predicted to be highly or very highly dependent upon groundwater levels (Chapter 3), particularly the two species of tortoise.

The western swamp tortoise, *Pseudemydrua umbrina* is Australia's most threatened reptile species and studies indicate that many aspects of its life history are highly dependent upon rainfall and temperature, making it likely that this species will be severely affected by climate change.

First, predicted continuing declines in annual rainfall will affect a variety of aspects of the tortoise's ecology. Females are not able to produce eggs in low rainfall years, thus two successive years of average or above average rainfall are required for effective recruitment to take place. Furthermore, hatchlings must achieve a critical body weight of about 25 g in their first six months in order to survive the following summer (Burbidge 1967; 1981) and this is not achievable in years of below average rainfall because the swamps retain water for only a short time. Declines in rainfall and groundwater levels will also clearly influence the availability of suitable ponds for *P. umbrina* as wetlands become increasingly dry.

Second, rising water temperatures in early summer appear to be a trigger for the tortoise to leave the water and aestivate, with turtles leaving the water at ~ 28 °C (Burbidge and Kuchling 1994). With increasing temperatures predicted for all seasons, water temperatures should rise earlier in spring than in the past, particularly if water depths in ponds are reduced by decreasing rainfall. This could have consequences for the survival of the tortoises because aestivating tortoises can desiccate during the summer (Burbidge and Kuchling 1994). Survival over summer is also likely to become increasingly dependent upon access to a high quality aestivation refuge which will provide buffering against extreme high ambient temperatures. Also, *P. umbrina* has extremely low reproductive output, producing only one clutch of three to five eggs per year and hatching is triggered by lowering temperatures in the winter (Burbidge and Kuchling 1994). Captive breeding experiments have shown that without a decline in temperature most embryos still develop to hatching size, but do not hatch and eventually die (Burbidge and Kuchling 1994).

Like many reptile faunas worldwide, the biology and ecology of most other reptiles in the Swan Coastal Plain and GSS study area is poorly known, making it difficult to speculate about the impacts of climate change on populations. Nevertheless, climate change should influence reptile populations through changes in the local environmental and habitat conditions, changes in phenology and life history and by altered fire regimes.

An important indirect way that climate change may affect reptiles is via increased leaf litter decomposition rates associated with increasing temperatures and changes to leaf chemical components (Aerts 1997; Couteaux *et al.* 1995; Whitfield *et al.* 2007) and decreased amounts of leaf litter habitat. Preliminary vegetation condition results indicate that reduced rainfall leads to reduced vegetation cover (Chapter 2). Consequently, there may be a reduction in the amount of leaf litter shed, resulting in a decline in leaf litter habitat. Many terrestrial lizards, particularly skinks and pygopodids, rely upon litter for refuge; studies indicate that thicker litter is often preferred (Howard *et al.* 2003 ) and that lizards show a preference for different types of litter based on their thermal properties (Valentine *et al.* 2007). Additionally, many lizards lay their eggs in shallow burrows under litter and changes in litter composition and/or depth could adversely affect reproductive output if eggs are exposed to unfavourable conditions. Many species also use litter, and other refuges, to shelter from extreme temperatures (Melville and Schulte 2001).

In the GSS study area, overall abundance of reptiles, and also the abundance of some individual species including *Hemiergis quadrilineata* and *Menetia greyii*, increases with both increasing litter depth and litter cover (Valentine *et al.* 2009). Bamford (1997) also notes that *R. a. adelaidensis* can bury itself in leaf litter up to at least 20 mm when inactive. Thus, if climate change results in altered litter quality and/or quantity, it can be speculated that this could have important implications for litter dependent species and other litter associated reptiles in the region.

Many reptiles also have temperature-dependent sex determination, whereby the sex of the offspring is determined by the environmental temperature during incubation. Climate change could be catastrophic for reptile populations having this characteristic, as rising temperatures would result in affected populations becoming extremely biased towards one sex (Janzen 1994; Mitchell *et al.* 2008; Spencer and Janzen 2004). Information on species exhibiting temperature-dependent sex determination in the GSS study area is limited, but a number of reptiles are closely related to other species that do exhibit it. For example, the dragon *R. a. adelaidensis* is closely related to *Ctenophorous* spp., (Melville *et al.* 2008).

## *Birds*

The primary effects of climate change on birds are expected to occur through changes in life history and phenology – in particular, changes in breeding season associated with warming temperatures, and also the potential for mismatch with primary food sources and/or food shortage if resources become scarce. Studies in the northern hemisphere indicate that climate change is influencing the timing of breeding in many species of birds. With the observed warmer conditions in recent years, northern hemisphere birds are typically breeding earlier (Crick *et al.* 1997; Crick and Sparks 1999; Parmesan and Yohe 2003). Warmer winter conditions, coupled with reduced winter rainfall is also predicted for south-west Western Australia (see earlier section).

The evidence for similar patterns in Australian species is currently scant, although studies so far indicate that climate change is also advancing breeding date in some regions (Norment and Green 2004). Declining rainfall has also had a negative impact on the reproductive success of some species, with Chambers *et al.* (2005) finding that mild warming and reduced rainfall have resulted in earlier laying, with reduced clutch size, in the critically endangered helmeted honeyeater in Victoria (*Lichenostomus melanops cassidix*). Similarly, a decline in reproductive output in the blue-breasted fairy wren (*Malurus pulchurrimus*) has been linked to years of below average rainfall (Brooker and Brooker 2001). Brooker & Brooker (2001) also demonstrated that in years of reduced rainfall the breeding season of *M. pulchurrimus* is reduced by up two months. We expect a similar response to rainfall in the closely related GSS study area species, the splendid fairy wren (*Malurus splendens*).

Currently, there are no published studies which specifically examine the impacts of climate change on any individual GSS bird species. However, Arnold (1988) predicted declines in some species that occur in the GSS study area, including the western rosella *Platycercus icterotis* and splendid fairy wren, and speculated that decreasing rainfall could result in a contraction in range for many Bassian species toward the wetter south-west region, while Bassian–Eyrean species were predicted to expand into the south-west. Similarly, Abbott (1999) predicts that climate change will result in a shift in species composition, allowing species currently restricted to the eastern sectors of the forest to penetrate into the south-west.

The most noticeable impact of climate change on avifauna of the GSS study area is likely to be in wetland dependent birds, which are expected to be severely affected by declining groundwater and rainfall. In Chapter 3, we identified ~ 70 species that occur within the GSS study area and are either highly or very highly dependent upon groundwater levels (Bamford & Bamford 2003). These include the black bittern (*Ixobrychus flavicollis*) and Australasian bittern (*Botaurus poiciloptilus*), both of which are now very rare in the GSS study area due to the large-scale loss of coastal plain wetlands (How and Dell 1993). Reduced rainfall, coupled with continued groundwater abstraction, will see many wetlands change to dry communities (see Chapter 5), reducing both habitat quality and quantity for wetland dependent birds.

Declining rainfall and groundwater levels will also affect non-wetland habitats, with most of the vegetation in the GSS study area being groundwater dependent to some degree (Chapters 4 and 5). This will have flow-on effects to bird fauna in the GSS study area, particularly as habitat structure and composition alters. A recent study predicted that extensive tree death will become a stark consequence of climate change if predictions of increasing severity and frequency of drought are realised (Fensham *et al.* 2009). The death of hollow-bearing trees is likely to have an impact on the reproductive success of species which breed only in hollows, such as Carnaby's black-cockatoo and the western rosella, and will also decrease the number of refuges available for all species to use as shelter on extremely hot days.

Climate change is also expected to result in the decline of many species of *Banksia* in the GSS study area. Changes in flowering phenology and a reduction in seed production is also expected in some species with continuing warming and decline in rainfall (Enright *et al.* 1998). These changes clearly have important implications for the bird fauna of the region, particularly granivorous (e.g. Carnaby's black-cockatoo) and nectar-dependent species, including a number of dispersive honeyeaters. The potential impacts of climate change on birds in the GSS study area requires further work.

## *Mammals*

Declines and extinctions of Australian mammals have been concentrated within species which fall into the 'critical weight range' – that is, medium-sized mammals with a body mass between ~ 50 g and 5 kg (Burbidge and McKenzie 1989; Johnson and Isaac 2009; McKenzie *et al.* 2007). The most likely explanation for this pattern is that these species are of the preferred prey size for introduced cats and red foxes (Johnson and Isaac 2009). Disease may also have contributed to the mammal decline (Abbott 2006). However, there is gathering evidence that rainfall patterns can also interact to increase risk for species in low rainfall areas.

In the lower annual rainfall areas of Western Australia, decline and/or extinction of mammal species are higher (Burbidge and McKenzie 1989). For example, the Darling District (located to the east of the GSS study area) was placed at the upper end of rainfall levels and lower end of species at risk – indicating that a high proportion of species that were of stable status based on rainfall level at that time. More recent studies have also demonstrated that mammal declines have been more marked in low rainfall areas (Johnson and Isaac 2009; McKenzie *et al.* 2007) and this suggests that species within the critical weight range will be most at risk from declining rainfall and increasing aridity associated with climate change (see also DEC–GSS projects section). The causes of the interaction between declining rainfall and body size are unclear. However, a study in the Northern Territory found a strong correlation between mammal declines and reduced groundwater levels, with the authors suggesting that drying water holes were the primary cause of decline in middle-sized species (Braithwaite and Muller 1996). Johnson & Isaac (2009) also suggest that medium-sized mammals are more at risk of predation by cats and foxes in arid areas with less cover.

Climate change induced rainfall declines may also affect the reproductive success of many mammals in south-west Western Australia. For example, there is a strong positive association between the body condition of females and their offspring, and rainfall, in brush-tailed phascogales in Western Australia (Rhind 2002; Rhind and Bradley 2002). Brush-tailed phascogale populations in Western Australia may be limited by food resources and thus be particularly vulnerable to annual fluctuations in rainfall (Rhind and Bradley 2002). Similarly, survival of honey possums (*Tarsipes rostratus*) has also been linked to rainfall (Wooller *et al.* 1998; see also DEC - GSS projects section). As this species is the most commonly occurring native mammal in the GSS study area, we explore the possible impacts of reduced rainfall on honey possum populations in the DEC–GSS project section below.

There is also evidence that marsupials will be more vulnerable to extreme high temperatures which are expected as a result of climate change. Previous studies have demonstrated that marsupials are less efficient at maintaining their body temperature than eutherian mammals (McManus 1969; Nicol and Maskrey 1977), with some arboreal marsupials showing signs of extreme heat stress at ambient temperatures of 20°C to 30°C (Moore *et al.* 2004). Given that the number of days over 35°C is predicted to increase, this could have implications for marsupial species unable to find refuge from the heat. Refuges that provide thermal buffering from extreme temperature events will be very important. For example, the skirts of grass trees *Xanthorrhoea preissii* provide a thermal buffer during temperature extremes, and are utilised by *Antechinus flavipes leucogaster* (mardos) (Swinburn *et al.* 2007). In general, we can predict that species which utilise refuges, such as tree hollows, rock crevices or burrows, and are primarily nocturnal, such as the possums, bettongs, phascogales, and quolls, should be buffered from extreme temperatures more than species that are active during the daytime and/or do not shelter in refuges, such as the numbat, kangaroos and wallabies.

Of the ten non-bat mammals considered to be extant in the GSS study area, four were identified as being highly or very highly susceptible to altered groundwater levels (Chapter 3). The water rat or rakali, *Hydromys chrysogaster*, is the only strictly groundwater-dependent mammal in the GSS study area and thus is significantly threatened by declining rainfall and associated declines in groundwater level (Bamford and Bamford 2003). Rakali have already suffered a significant decline in south-west Australia (Lee 1995) and are highly susceptible to loss of habitat through the contraction and drying out of lakes either through filling and draining for alternative land use, decreasing rainfall and drying climate, and reduced groundwater levels. Loss or reduction in size and quality of wetland areas should also affect the food resource for rakali, as they feed on large aquatic insects, fish, crustaceans and other wetland associated prey. The three other mammals species identified, *Isoodon obesulus*, *Rattus fuscipes* and *Macropus irma*, are strongly associated with dense vegetation surrounding wetlands, and will be susceptible to changes in vegetation structure.

Critically, many of the mammals in the GSS study area and on the Swan Coastal Plain are very short lived. Species such as the western quoll, honey possum and brush-tailed phascogale often only live for a year, or three at the most. Thus, any mass mortality events which reduce the numbers of adults, possibly combined with declines in reproductive output and offspring survival associated with reduced rainfall, could result in very rapid population crashes in some species. In support of this, Braithwaite and Muller (1996) also suggest that an annual life cycle is likely to be a very vulnerable life history strategy in times of drought.

## Summary of DEC–GSS projects (2007–09)

### Project 1: Assessing the vulnerability of frogs to reduced rainfall and groundwater

Technical report references:
1) Isaac, J, Valentine, L and Wilson, B (in prep) *Predictions of fauna responses to declining rainfall in the GSS study area*. Unpublished report prepared for the Gnangara Sustainability Strategy, Perth.

2) Bamford M J & Huang N (2009) *Status and occurrence of frog species in the Gnangara Sustainability Strategy study area*. Unpublished report prepared by MJ and AR Bamford Consulting Ecologists and the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

Assessing the vulnerability of species to climate change has been identified as crucial for conservation planning and management at the regional scale (Williams *et al.* 2008). Here, we quantify relative vulnerability to declines in rainfall and groundwater for the 13 species of frog previously recorded in the GSS study area, in order to identify those species which are most at risk in the near future.

We investigated the vulnerability of the 13 frog species previously recorded in the GSS study area using a variation of the ‘rarity’ model of Rabinowitz (1981; Rabinowitz *et al.* 1986). Rabinowitz’s model uses data on geographic range size, local abundance and habitat specificity to form a matrix which reflects different types of rarity and commonness, and can be used to rank vulnerability (Kattan 1992).

We adapted this model to include life history trait information, available for each of the 13 frog species, that are thought to be important in determining their vulnerability to hydrological changes likely to result from climate change (Bamford and Huang 2009). The traits included were:

- **Reliance on winter surface water or rainfall.** Species reliant on winter rainfall events for breeding (e.g. tadpoles that emerge from flooding burrows) and/or found only in sites with winter surface water are most likely to be vulnerable to continuing declines in winter rainfall in the GSS study area. Based on the results in Bamford and Huang (2009) and data from the literature for other species, species were scored for presence or absence of this trait.
- **Length of the larval period.** A long larval period should make a species more vulnerable to hydrological change, as mass larval mortality is more likely under drought conditions (Bamford and Huang 2009). For this trait, each species was classed as having either a long or short larval period. A long larval period was defined as longer than the median for all species of 120 days. A short larval period was equal to or less than this figure.
- **Dispersal potential.** The potential to disperse to new habitats is likely to be advantageous under climate change. Species were classified as having high or low dispersal potential based on known movements of either adults or metamorphs, primarily sourced from information in Bamford and Huang (2009).

Based on the method of Kattan (1992) we assigned a number between 1 and 8 to each geographic cell in the model to indicate vulnerability (Table 10.2). Species assigned the value 1 are vulnerable in all three traits; species assigned the value 8 are comparatively resilient in all three traits. Of the remaining cells in the model, three are rare in two traits and three are rare in only one. Intermediate values were assigned according to the following rationale. For frogs, reliance on a permanent water source will probably be the most important factor affecting vulnerability to climate change. A short larval period will then probably be more important than dispersal ability, since most frogs will be unable to disperse long enough distances to find a water source in the face of region-wide drought conditions. Thus, we ranked traits in order of the following importance: reliance on winter water or rainfall was highest, then length of larval phase, then potential for dispersal.

Using data from recent frog surveys across the GSS study area (described in Bamford & Huang (2009)), we then investigated how the final vulnerability rank correlated with estimated current abundances. Pearson's (*r*) correlation coefficient was used to assess the strength of relationships.

Table 10.2: Ranks for the potential vulnerability to declining water in 13 frog species of the GSS study area; species which have high vulnerability to all three traits (see text) are assigned a rank of 1, darker shading indicates a higher relative vulnerability ranking.

| | | **Length of larval stage** | | | |
|---|---|---|---|---|---|
| | | Long | | Short | |
| **Dispersal potential** | | Low | High | Low | High |
| **Winter water or rainfall dependent?** | Yes | 1 | 2 | 3 | 5 |
| | No | 4 | 6 | 7 | 8 |

The results suggest that one species (*L. adelaidensis*) is most likely to be very vulnerable to predicted changes in hydrology in the GSS study area (Table 10.3). This species relies on permanent water and/or predictable winter rainfall events to initiate breeding, has a long larval period, and a low dispersal potential. *L. adelaidensis* is thought to be an ancient relictual species from wetter times in the region and this could explain its many adaptations to a high rainfall environment. However, *L. adelaidensis* was one of the most abundant frog species in the region during recent surveys (Bamford and Huang 2009). *L. adelaidensis* has a relatively long lifespan and thus, even if current decreases in rainfall have resulted in a decline in breeding success, we might expect a significant lag before declines in the adult population are noted in the field. Similarly, *C. glauerti* is predicted to be highly vulnerable (Table 10.3), but was observed at several sites (Bamford and Huang 2009). This species also has a relatively long life span, so again, there may be a time lag between when declines occur and when they are identified in the field.

Concurring with the conclusions of Bamford and Huang (2009), we found that *L. moorei*, the motorbike frog, was also highly vulnerable due to a reliance on permanent water and having a very long larval period. However, this is a large species which has been found considerable distances away from known breeding sites, suggesting relatively greater potential for dispersal compared to other species, which may mitigate the impacts of declining water in some local populations (Table 10.3).

Table 10.3: Summary of the potential vulnerability to declining water or rainfall in the 13 frog species previously recorded in the GSS study area. Darker shading indicates a higher relative vulnerability ranking.

| **Species** | **Surface water/winter rainfall dependent?** | **Larval period** | **Dispersal potential** | **Vulnerability to declining rainfall** |
|---|---|---|---|---|
| *Litoria adelaidensis* | Yes | Long | Low | 1 (high) |
| *Crinia glauerti* | Yes | Long | High | 2 |
| *Litoria moorei* | Yes | Long | High | 2 |
| *Helioporus albopunctatus* | Yes | Long | High | 2 |
| *Helioporus psammophilus* | Yes | Long | High | 2 |
| *Crinia georgiana* | Yes | Short | Low | 3 |
| *Myobatrachus gouldii* | Yes | Short | Low | 3 |
| *Pseudophryne guentheri* | No | Long | Low | 4 |
| *Neobatrachus pelobatoides* | No | Long | Low | 4 |
| *Helioporus barycragus* | Yes | Short | High | 5 |
| *Lymnodynastes dorsalis* | No | Long | High | 6 |
| *Heleioporus eyrei* | No | Long | High | 6 |
| *Crinia insignifera* | No | Short | Low | 7 (low) |

In general, there was a positive relationship between occupancy, in terms of the number of sites a species was recorded from during field surveys (Bamford and Huang 2009), and the vulnerability rank found in our analysis (Figure 10.5), (Pearson correlation coefficient, $r$ = 0.47). Species which were found at only a few sites were found to have high vulnerability, while those found in many sites tended to have low vulnerability to predicted changes in rainfall and water availability.

The two outliers to this relationship were *L. adelaidensis* and *C. glauerti*, both of whom have relatively long life spans, possibly resulting in a time lag between declines and their detection. When these two outliers were removed, the correlation coefficient between abundance and vulnerability was substantially higher ($r$ = 0.89). This confirms that the current abundance of frog species in the GSS study area can be explained, in part, by their sensitivity to prevailing hydrological conditions. However, data on the life history and ecology of many of the frog species are limited, and further, more detailed vulnerability analyses should be conducted on the basis of rigorous field studies into behaviour and ecology of priority species.

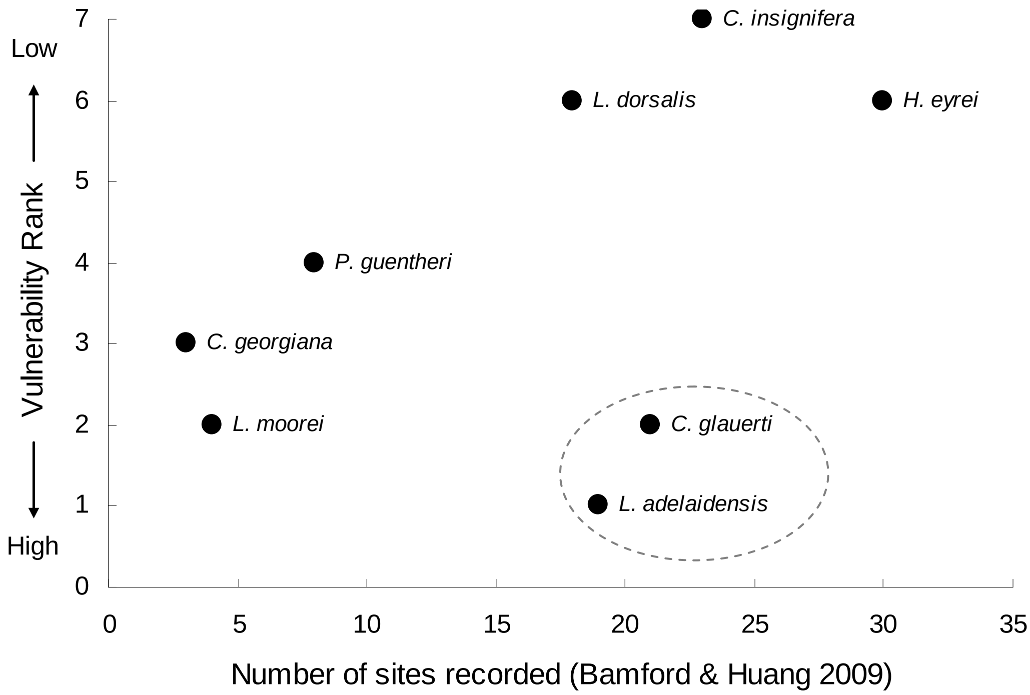


Figure 10.5: The relationship of relative abundance (in terms of number of sites recorded in Bamford and Huang (2009)) and predicted vulnerability to declining rainfall for eight frog species found in the GSS study area (species with no abundance estimates were excluded). Dashed line indicates the two outlier species.

## Project 2: Identifying extinction-prone mammals in the GSS study area under an increasingly arid climate

Technical report reference: Isaac, J., Valentine, L. and Wilson, B. (in prep) *Predictions of fauna responses to declining rainfall in the GSS study area*. Unpublished report prepared for the Gnangara Sustainability Strategy, Perth.

Medium-sized mammal species appear historically prone to declines and extinctions in Australia, and there is some evidence that body size can interact with declining rainfall to increase extinction risk in medium-sized mammals. This is thought to be due to increased predation risk from introduced cats and foxes in arid, open environments and also the impacts of a lowered watertable on water holes (Braithwaite and Muller 1996; Johnson and Isaac 2009). Additionally, Johnson *et al.* (2007) suggest that dingo control is common in arid regions and that the removal of dingos can allow mesopredators, such as introduced cats and foxes, to increase in density with detrimental impacts on native mammals whose mass is within the critical weight range. Here, we investigate which mammal species in the GSS study area are within this range, and assess their current population persistence relative to other mammal species which occur in low rainfall regions.

Using data from Johnson and Isaac (2009), we investigated the relationships between the current persistence (as defined in Johnson and Isaac (2009)) and the body mass of native mammals (excluding rodents and bats) known to occur, or to possibly occur, in the GSS study area, and those in the greater Swan Coastal Plain region, relative to other mammal species across Australia which fall into the ‘low rainfall’ category (≤ 500 mm/year). Persistence was measured as the current range on mainland Australia as a proportion of the range at the time of European arrival. High values represent species that have suffered little or no decline while low values indicate severe declines or extinctions. Assumptions and methodologies are available in Johnson and Isaac (2009).

The analysis shows that most GSS study area mammal species (66%) are within the critical weight range, and that all of these species are showing varying signs of decline (across their entire Australian geographic range). Combining all species in the GSS study area and Swan Coastal Plain, 54% are within the critical weight range, and Swan Coastal Plain species have lower persistence (average = 0.88) than all other species from low rainfall areas (average = 1.02). For the full data set, there is a significant non-linear (second order polynomial) relationship that demonstrates a decline in persistence within the critical weight range species (Figure 10.6; 1: $r^2 = 0.41$, F = 16.29, $P < 0.0001$), with increasing persistence in the smaller and larger species. This relationship is also significant for just the GSS study area species (although the F-value is lower, probably due to the reduction in sample size), and the $r^2$ value is actually greater (Figure 10.6; 1: $r^2 = 0.55$; F = 4.86, $P = 0.04$). This means that for GSS study area species, body mass explains more (14%) of the variation in persistence than it does for all species from arid regions. The regression line for GSS study area species is also lower than the line for all species, indicating that in general, they have lower persistence for body mass compared to the entire data set.

Given the apparent intrinsic vulnerability of medium-sized mammal species in the GSS study area, and the evidence for an interaction between declining rainfall and extinction risk, it is likely that future declines due to climate change will be most evident in these species. Of particular concern are declines in the brushtail possum, southern brown bandicoot, chuditch and the bettong species. However, this dataset is limited in that it does not include rodents due to data limitations, and a similar, more extensive analysis which includes rodents is recommended.

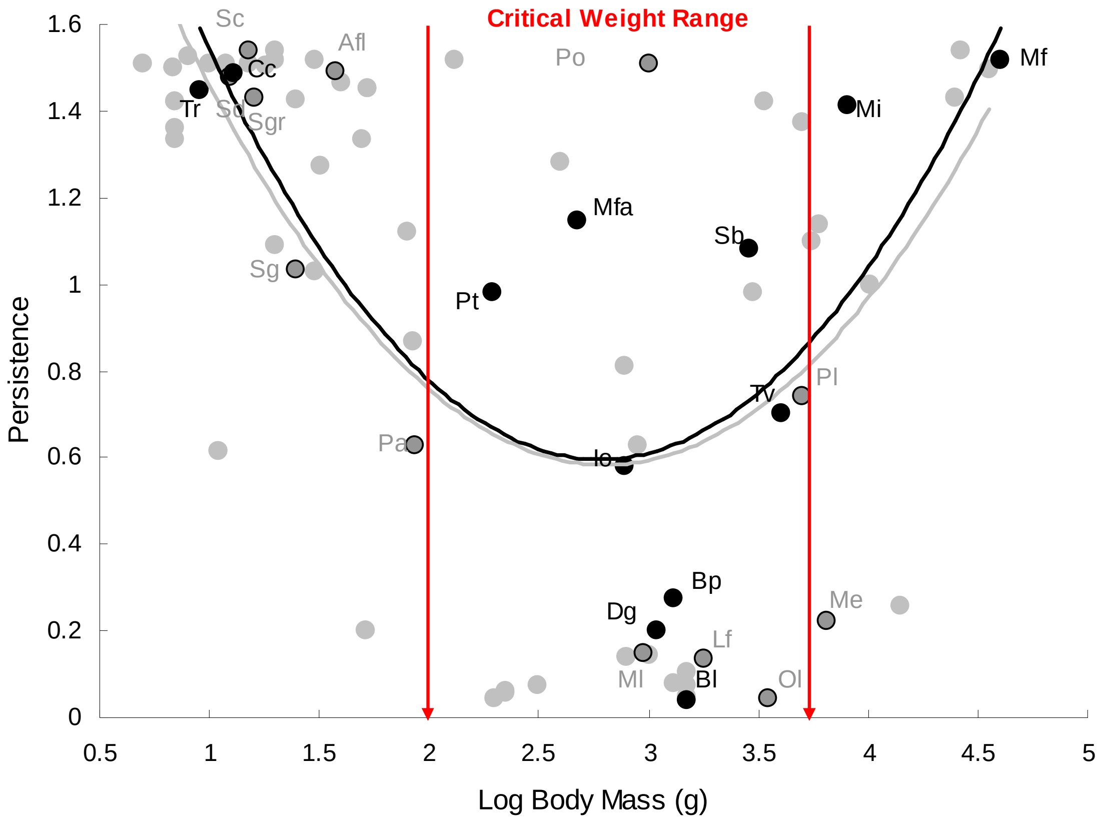


Figure 10.6: The relationship between log body mass (g) and persistence in Australian mammal species from low-rainfall habitats (light grey circles), those species with historical occurrence on the Swan Coastal Plain (dark grey circles, black outline) and in the GSS study area specifically (black circles). The limits of the 'critical weight range' are shown in red. The black line shows the polynomial relationship between body mass and persistence for all species; the grey line shows the same relationship for GSS species. GSS species codes (in black text): Tr: *Tarsipes rostratus*, Cc: *Cercartetus concinnus*; Pt: *Phascogale sp.*; Mfa: *Myrmecobius fasciatus*; Sb: *Setonix brachyurus*; Tv: *Trichosurus vulpecula*; Io: *Isoodon obesulus fusciventer*; Bp: *Bettongia pencillata*; Dg: *Dasyurus geoffroii*; Bl:

*Bettongia lesueur*; Mi: *Macropus irma*; Mf: *Macropus fuliginosus*. Swan Coastal Plain species codes (in grey text): Sd: *Sminthopsis dolichura*; Sg: *S. granulipes*; Sc: *S. crassicaudata*; Sgr: *S. grisoventer*; Afl: *Antechinus flavipes leucogaster*; Pa: *Parantechinus apicalis*; Po: *Pseudocheirus occidentalis*; Pl: *Petrogale lateralis*; Lf: *Lagostrophus fasciatus*; Ml: *Macrotis lagotis*; Ol: *Onychogalea lunata*; Me: *Macropus eugenii*.

## Project 3: Modelling the potential impacts of climate change on the honey possum

Technical report reference: Isaac, J., Valentine, L. and Wilson, B. (in prep) *Predictions of fauna responses to declining rainfall in the GSS study area.* Unpublished report prepared for the Gnangara Sustainability Strategy, Perth.

The honey possum is an obligate nectarivore, with a diet restricted to nectar and pollen from Myrtaceae, Proteaceae and Epacridaceae plants. Thus, their continued persistence is completely dependent upon the year round availability of flowering plants. Currently, there is little information on the abundance or ecology of honey possums in the GSS study area. However, long-term studies on the ecology of honey possums have been conducted in Scott National Park (Bradshaw *et al.* 2007) and Fitzgerald River National Park (Wooller *et al.* 1981; Wooller *et al.* 1998). This research has illustrated that honey possum capture rates are correlated with rainfall from the previous year or previous two years (Bradshaw *et al.* 2007; Wooller *et al.* 1998), with a reduction in the number of possums following low rainfall attributable to a decrease in nectar production (Bryant 2004; Wooller *et al.* 1998). Given this relationship, honey possums should be vulnerable to declining rainfall associated with climate change. We used available data on honey possums to model the potential consequences of declining rainfall on their abundance, and to investigate how a change in phenology in *Banksia* flowering patterns could affect breeding success. Due to limited data on honey possums in the GSS study area, we assume that the ecology and biology of honey possums on the Swan Coastal Plain is comparable to that of the south coast areas of Western Australia.

## *Direct impacts of reduced rainfall on honey possums*

Using the rainfall and capture data from Wooller *et al.* (1998), we investigated the potential effect of declining rainfall on the density of honey possums, using capture rate as a surrogate for population density (see Wooller *et al.* 1981). First, we calculated a linear equation for the relationship between capture rate and the annual rainfall from the previous year, as below:

Capture rate (per 1000 trap nights) = 0.12 x annual rainfall (mm) + 2.84 ($r^2$ = 0.65)

Then, we modelled the effect of declining annual rainfall on the predicted density of honey possums using a variety of scenarios, ranging from no decline in annual rainfall to a maximum of 50% decline. Initial values (i.e. assuming no decline) were calculated based on the long-term estimated annual rainfall for the GSS study area of 807 mm (taken from Bureau of Meteorology data, average from 1905–2007; see Chapter 1, Figure 1.5). We then overlaid the shorter-term annual rainfall averages (post-1976 and post-1997) and annual rainfall from recent years (2006 and 2007).

The data from Wooller *et al.* (1998) indicates a considerable decline in the density of honey possums with decreased rainfall (Figure 10.7). The extreme scenario modelled a 50% decline in rainfall, and resulted in a 49% decline in honey possums. The GSS study area is already experiencing declines in annual rainfall, with a 9% decline in rainfall since 1976, and a 13% decline in rainfall since 1997, indicating that honey possum rates may have already declined in the GSS study area (Figure 10.7). The annual rainfall of two recent years (2006 and 2007) indicates that extremely dry years are currently occurring in the GSS study area, and that honey possum abundances would be predicted to be quite low (prediction of ~ 37% reduction in capture rates in 2006). As honey possums typically only live for 1 to 3 years, there is the potential for successive dry years to affect the persistence of honey possum populations.

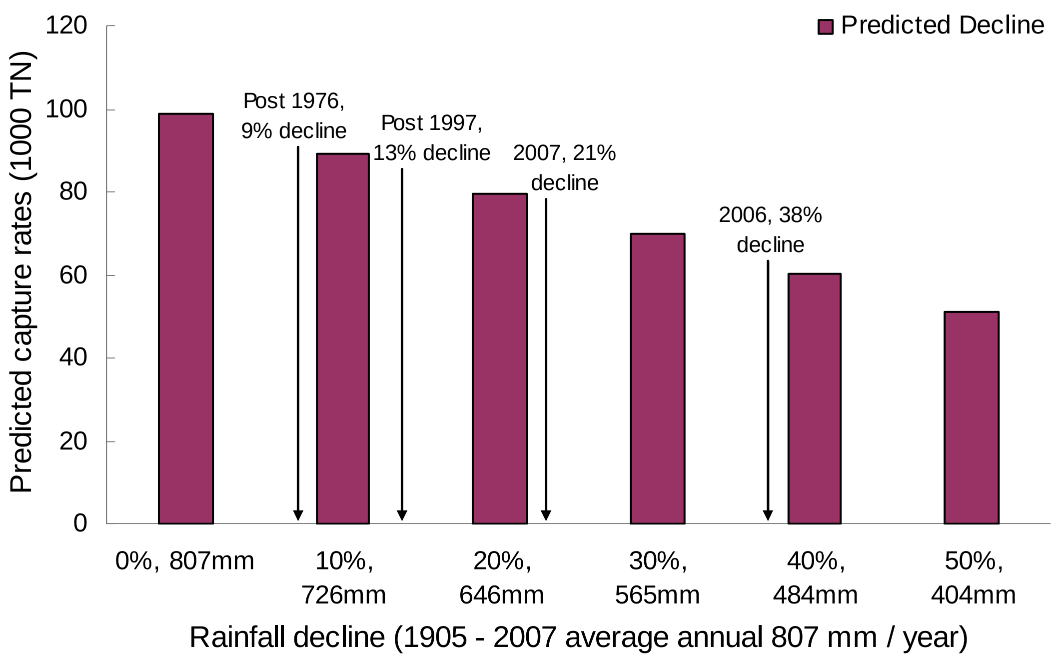


Figure 10.7: The predicted impact of incremental declines in annual rainfall (based on the long-term average annual rainfall of 807 mm/year) on the density of honey possums (capture rates are used as a surrogate for density). Figures were derived from the relationships found in Wooller *et al.* (1998). Average annual rainfall (post 1976 and post 1997), and annual rainfall from recent years (2006, 2007) is also shown.

## *Potential phenological mismatch of* **Banksia** *and honey possums*

Although honey possums can breed year round, breeding peaks during the winter months (Bryant 2004). If a decline in nectar availability (see Figure 10.4) coincides with peak breeding season, and the species is unable to adjust its breeding phenology at the same rate as the plants, breeding success and survival of offspring could be impaired. To investigate the potential impact of a shift in flowering phenology in *Banksia* species, we used data already presented in the vegetation and floristics section of this chapter on the current and potential (under climate change) annual flowering pattern of key *Banksia* species in the GSS study area (see Figure 10.4 and accompanying text) in conjunction with breeding seasonality data (Wooller *et al.* 2000), which reports the number of females caught with pouch young in each month.

The model results illustrate that currently the peak in breeding in honey possums (in south-west Western Australia) coincides with the peak flowering time of *Banksia* in the GSS study area (Figure 10.8a). However, if *Banksia* shift their flowering in response to climate change (as outlined in the vegetation and floristics section of this chapter), and honey possums are unable to shift their breeding seasonality to the same degree, peak breeding season could occur at a time with reduced nectar availability due to phenological mismatch (Figure 10.8b). In this hypothetical scenario, if the honey possums do not change their reproductive seasonality, peak breeding will occur when nectar is least available.

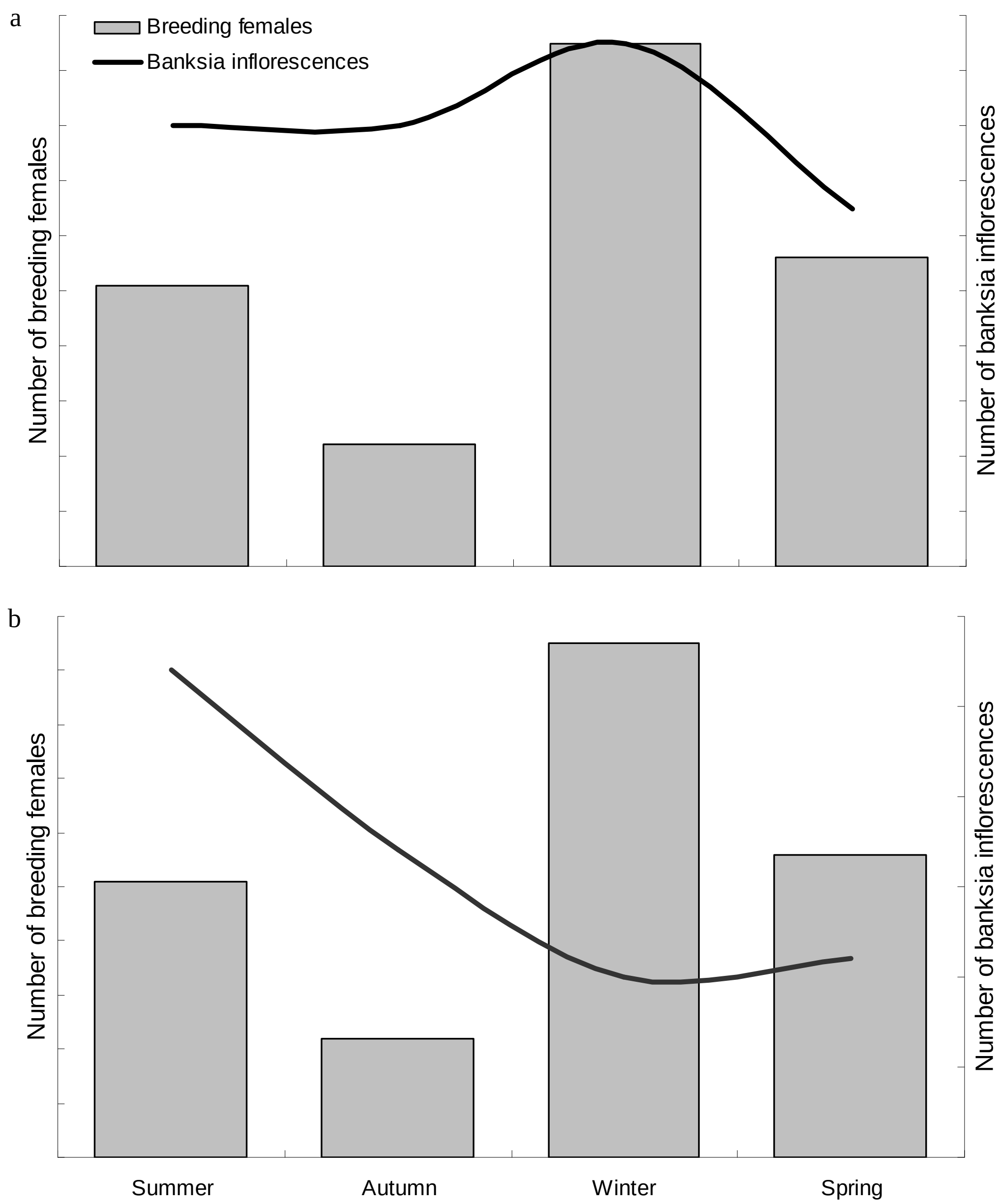


Figure 10.8: a) The number of female honey possums (grey bars) caught with pouch young during each season in south-western Western Australia, based on Wooller *et al.* (2000) trapping data from 1984–1995, and the total number of inflorescences of five species of

*Banksia* (black line) during each season in the years 1976 and 1977 (based on Whelan and Burbidge 1980); b) hypothetical example of phenological mismatch, assuming that the breeding seasonality of honey possums remains the same, while the flowering phenology of *Banksia* has changed in response to climate change, based on the model shown in Figure 10.4.

These preliminary analyses suggest that both the overall abundance and breeding success of honey possums could be negatively affected by declining rainfall in the GSS study area due to climate change. Because of their limited ability to move long distances (Wooller *et al.* 1998), honey possums will be unlikely to avoid drought-induced food shortage by moving to other habitats, especially if the impacts of drought are widespread. Local or widespread extinctions of honey possum populations may thus result. Declines in *B. ilicifolia* populations on the Scott River plains, due to a lowered watertable (Groom *et al.* 2000a), have previously been linked to a decrease in honey possum populations (Phillips *et al.* 2004). However, our models do not include the potential for honey possums to shift their breeding seasonality to follow potential changes in flowering phenology of *Banksia* species. Given that honey possums generally live for a few years, and that lowest densities occur around late spring following the winter breeding peak (indicating a main die-off period), opportunity for shifts in breeding period may be limited.

## Discussion

Climate change is a primary threat to global biodiversity values and ecosystems function. Contemporary climate change is a significant threat because the rate of change is rapid, and may not allow species and ecosystems adequate time to adjust to the new environmental conditions. In addition, the climatic changes are occurring in ecosystems that are already pressured by numerous human-mediated disturbances. Broad climate changes that have started to occur, and are predicted for the future in south-west Western Australia, include increases in average temperatures and decreases in average annual rainfall (IOCI 2002; IPCC 2007). Furthermore, extreme weather events, such as high temperatures and very low rainfall years, are predicted to occur more frequently.

South-west Western Australia is predicted to undergo the most intense drying of any region in Australia by 2100 (Pitman and Perkins 2008). Modelling indicates that rainfall will decline by as much as 20% by 2030 and 60% by 2070 (Jones and Preston 2006; Preston and Jones 2006). Compared with long-term averages, average annual rainfall since 1997 has already declined by 13% (Chapter 1). The declines in rainfall will predominantly occur during winter months, although declines in spring and summer rainfall are also expected.

Species responses to climatic changes are predicted to be variable but may include:

- a shift in species distribution or range (Parmesan 2006)
- changes in phenology, such as advanced flowering in plants (Lu *et al.* 2006)
- changes in biology, such as altered sex ratios (Spencer and Janzen 2004)
- altered species composition and structure (IPCC 2007).

The response of a particular species to climate change will depend on its resistance (ability to withstand environmental changes) and resilience (ability to recover from environmental changes). Broadly, generalists species with a wide geographic range, high local abundance and fast generation time, high dispersal potential, and less specialised habitat requirements will be more likely to adapt to climatic changes and/or shift their range accordingly. In contrast, species with a restricted distribution, patchy or low local abundances and low reproductive output, limited dispersal potential, and specialised habitat requirements are likely to be more susceptible to the impacts of climate change.

The focus of this chapter has been on the impacts of declines in rainfall induced by climate change, which, coupled with abstraction, have resulted in reduced groundwater levels. Although one of the obvious consequences of declining rainfall is a loss of wetland habitat (Chapter 5), research also indicates that terrestrial vegetation in the GSS study area has already been affected by rainfall declines (Groom *et al.* 2001; Groom *et al.* 2000b). For example, a drought intolerant species such as *Banksia littoralis* has been replaced in some areas by the more drought tolerant *B. prionotes* (Groom 2004). Few long-term studies have been conducted on the fauna of the GSS study area, and there have been no detailed empirical studies on the impacts of declining rainfall. However, at least 90 species of vertebrate (excluding fish) are highly dependent on groundwater and are likely to be affected by declining rainfall (Chapter 3; Bamford and Bamford 2003), including several wetland dependent bird species that have already exhibited population declines.

Surprisingly, we know very little about the detailed biology and ecology of species in the GSS study area, which makes predicting future impacts of declining rainfall difficult. Studies have so far identified that future rainfall declines are likely to modify the structure and composition of some *Banksia* woodland (Groom *et al.* 2000a). Modelling of potential impacts of climate change also predicts a 66% decline of *Banksia* species by 2080 (Fitzpatrick *et al.* 2008), and there is the potential for disruption of pollinator–plant relationships *via* phenological mismatch. Fauna that are dependent upon rainfall to facilitate breeding (e.g. frogs, western swamp tortoise) are very susceptible to reduced rainfall. The western swamp tortoise may be particularly susceptible to long-term reductions in rainfall. Our preliminary models also indicate that the biology of several frog species make them very vulnerable to reduced rainfall. The predicted changes to *Banksia* woodland structure and composition are also likely to have flow-on effects for fauna. Altered vegetation cover and leaf litter habitat may affect the thermal range and shelter resources available for reptiles. Declining rainfall may also affect fire regimes, and work conducted in the GSS study area (Chapter 7), indicates that reptile communities in *Banksia* woodlands are strongly influenced by time since fire.

The populations of mammal and bird species in the GSS study area have already fallen substantially. This is thought to be caused primarily by a combination of habitat loss and introduced predators (Chapters 6 and 8). Changes in the structure and composition of *Banksia* woodland have flow-on effects for species that depend on them for their food resources, such as nectar-dependent honeyeaters and Carnaby's black-cockatoo. Our modelling suggests that mammals in the critical weight range in the GSS study area may be very susceptible to the impacts of declining rainfall. These include the southern brown bandicoot. In addition, the water rat (rakali) is a water-dependent mammal, and the loss of wetland habitat is likely to lead to local population declines or extinctions. Species that exhibit a positive relationship between population abundance and rainfall, such as the honey possum, are likely to decline under future rainfall scenarios. As a relatively short-lived species, the honey possum may also be susceptible to extreme weather events, such as sequential very low rainfall years.

Without detailed biological and ecological information on the species of the GSS study area, it is difficult to make accurate predictions on the impacts of declining rainfall. Further work focusing on some species (e.g. susceptible reptile and frog species, Carnaby's black-cockatoo, southern brown bandicoot, water rat and honey possum) will substantially improve modelling predictions. For example, using advanced climatic envelope models such as MaxEnt to explore possible responses to declining rainwater in susceptible species. In addition, identifying potential refuge areas for such species may be critical for future conservation management. In the GSS study area, there are a number of processes that are threatening species, including habitat loss and fragmentation (Chapter 6), altered fire regimes (Chapter 7), introduced predators (Chapter 8) and diseases (Chapter 9). Understanding the potential interactions between these processes and declining rainfall is particularly important for future management decisions, and is discussed in Chapter 12. Management of other threatening processes may increase the resilience and resistance of species to declining rainfall.

## Recommendations

In order to more accurately predict species responses to climate change, it is recommended that:

- detailed biological and ecological data of potentially susceptible species persisting within the GSS study area be collected to provide baseline information for predictive models
- models of the potential impacts of climate change on species in the GSS be developed using biomechanistic approaches that take into account both climatic factors and the biology and ecology of the species (e.g. MaxEnt).
- an integrated approach be developed that combines individual species vulnerability to climate change scenarios, potential for range expansions and contractions, capacity for adaptation (genetic or behavioural) and trophic relationships with interactions of other threatening processes.

In order to conserve and manage biodiversity under altered environment conditions resulting from climate change, it is recommended that:

- the vulnerability of priority species to climate change impacts be assessed and that this include potential adaptive management options to minimise impacts and increase adaptive capacity (for example, the framework suggested by Williams *et al.* (2008)).

- potential refuge areas within the GSS study area for susceptible species and communities be identified
- plan to manage an altered faunal assemblage – integrating new species while protecting those which still remain, with a focus on endemic species that may be unable to shift their range (e.g. western swamp tortoise, Carnaby's black-cockatoo and the honey possum) due to ecological and/or movement constraints.

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Gnangara Sustainability Strategy

An across-government initiative

Groundwater – Biodiversity – Land use

Biodiversity values and threatening processes of the Gnangara groundwater system

Edited by Barbara A. Wilson and Leonie E. Valentine


# Chapter Eleven: Risks and Priorities

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Our environment, our future

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Government of Western Australia
Department of Water
Department of Agriculture and Food WA
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Department of Environment and Conservation
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CSIRO
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# 11. RISKS AND PRIORITIES

## Key points

- A systematic and transparent planning process is used by organisations to ensure biodiversity values and features are protected and to identify conservation priorities.
- Ranking of biodiversity assets and identifying the risks that threatening processes cause are essential steps in the process of formulating conservation objectives and priorities.
- Decision making tools such as mathematical optimisation or qualitative methods such as multi-criteria decision analysis, ecological risk assessment or cost-benefit analysis are employed to determine transparent conservation objectives and priorities.
- Of the GSS study area, 43% had biodiversity assets which are rated as being of high significance and these are largely located in the north and central areas of the study area.
- Of the remnant vegetation across the Gnangara Mound, 19% is currently at high risk from *Phytophthora* dieback and a further 52% is at moderate risk.
- Priority areas for *Phytophthora* dieback management were identified for high and moderate risk areas.
- Projected costs of implementing *Phytophthora* dieback management, for susceptible remnant vegetation, across the GSS study area were investigated.
- Development of criteria and spatial datasets relating to biodiversity assets will improve the ranking models.
- Development of spatial models to predict the distribution of the *Phytophthora* dieback pathogen more accurately will improve the assessment of risk.

## Introduction

Setting priorities for biodiversity conservation is a necessary requirement in any region that supports a large number of high value biodiversity assets, especially if these are facing multiple threatening processes, and if there are only limited resources available for conservation. Conservation priorities should reflect both the risks that ecosystems and species are facing and how well they are included or represented in existing protected areas. In order to sustain biodiversity, decision makers invest in a diverse range of activities such as invasive species control, fire management, acquisition of land for formal reservation, rehabilitation and monitoring. Formal and transparent conservation planning processes need to be developed so that the limited funds available can be allocated among this diverse range of possible activities and between regions or subregions in such a way that biodiversity benefits are maximised (Margules and Pressey 2000; Possingham *et al.* 2008; Wilson *et al.* 2008; Wilson *et al.* 2007).

An increasing number of planning frameworks and decision making tools have been developed to assist planners to identify conservation objectives and determine funding priorities. Examples of frameworks include the 'Systematic conservation planning' approach of Margules and Pressey (2000) and the 'Conservation investment (cost-benefit) framework' of Wilson *et al.* (2007). Both qualitative and quantitative decision making tools are used during conservation planning. One example of qualitative decision making is the ranking of biodiversity assets using multi-criteria decision analysis, and an example of quantitative decision making is the 'mathematical optimisation' of Possingham (2001). Decision planning frameworks that have been widely used in other fields are also drawn upon. An example of this is risk assessment for which an Australian Standard (AS/NZ 4360) has now been developed for its application in an ecological context (AS/NZS 2004; Walshe 2005).

The aims of this chapter are to collate and review information on:

- approaches to conservation planning
- the methods used by DEC for conservation planning and for identifying conservation priorities
- the roles of decision theory, ecological risk assessment and cost-benefit analyses in biodiversity conservation.

These aims were addressed by reviewing literature. In addition, two DEC–GSS projects were undertaken to address identified knowledge gaps for the GSS study area. These projects are outlined below.

**Project 1: Ranking of biodiversity assets**

The objective of this project was to utilise two multi-criteria evaluation models to rank the biodiversity assets on the GSS study area.

**Project 2: *Phytophthora* dieback risk assessment of biodiversity assets**

*Phytophthora cinnamomi* has been identified as a major threatening process in the GSS study area. The aims of this project were to determine the extent of areas at high to moderate risk of being impacted by *P. cinnamomi*, to identify specific management actions and to calculate the total area costs for cost-effective management of the disease.

# Systematic, 'best practice' planning for conservation

The focus of systematic conservation planning is to locate, design and manage protected areas that comprehensively represent the biodiversity of each region (Mace *et al.* 2006). It is considered to be a transparent planning process whereby clear choices can be made as to what biodiversity values and features should be protected, and where explicit goals (preferably with quantitative operational targets) are set for their protection (Margules and Pressey 2000; Possingham *et al.* 2008). 'Systematic protected areas planning' is an important tool for ensuring the integrity of the broad ecosystem by achieving regional-scale goals, while catering for local requirements to influence management of each individual site in aspects such as size, shape, and use as appropriate (Possingham *et al.* 2008).

Systematic conservation planning involves several planning stages or steps (Groves *et al.* 2002; Margules and Pressey 2000; Noss 2003; Possingham *et al.* 2008) and methodologies will vary depending on the size, biogeography, nature of the ecosystems and land-use history of the planning region (Noss 2003). The process is not unidirectional – rather it is iterative and dynamic (Groves *et al.* 2002; Margules and Pressey 2000; Noss 2003). For example, changes may be required to incorporate new data, or new sites may need to be identified (Margules and Pressey 2000). The planning process is also an adaptive one and therefore should be reviewed and refined as planning cycles are completed (Groves *et al.* 2002; Noss 2003).

Possingham *et. al*. (2008) identified a number of features that all good conservation planning methods should have from the literature. Firstly it is important to identify and involve stakeholders, as collaborative decision making facilitates a good understanding of the process and increases accountability of those leading the planning process. Secondly, the definition of clear goals and objectives for the protection and restoration of biodiversity is integral to the planning processes and should include socio-economic goals and objectives.

It is essential to compile spatial data when designing a network of protected areas, as this provides valuable information on the location and extent of conservation features and on socio-economic and land use constraints. However, because of the complexity of biodiversity and lack of knowledge and spatial data, surrogate measures often have to be used (Margules and Pressey 2000) and input from experts is often sought (Groves *et al.* 2002).

Conservation targets need to be quantified, including how much of each conservation feature (such as habitats or species) to protect in the network of protected areas. To ensure ecological integrity and persistence, design principles must be established to address factors such as size, shape, number and connectivity of sites. Proximity of the sites to modified habitat should also be considered (Margules and Pressey 2000).

A process of review is required to determine to what extent conservation targets and network goals have been achieved through existing protected areas and to identify the shortfalls. Following this review, selection of new protected areas can occur where the focus is on identifying alternative configuration options of the protected area network that can fill the gaps identified and also meet design principles. From the options presented new sites for protection will be identified through consultative processes. Decision support tools should be used in this step so the effectiveness of alternative protected area networks can be assessed and socio-economic trade-offs can be considered. A number of decision support tools are available including Marxan (Ball and Possingham 2000) and C-Plan (Pressey 1999).

Implementation of conservation actions follow this planning and will include decisions on fine-scale boundaries of proposed protected areas and management measures or approaches for the sites.

Where limited resources prevent all sites being protected or managed immediately, a framework to set priorities for action, or implementation schedule, should be developed based on criteria such as degree of existing protection, conservation value, feasibility and threat (Groves *et al.* 2002). Interim protection measures for sites should also be considered.

Ongoing management and monitoring of sites is required if the ecological integrity of the network is to be preserved. Monitoring is also required to determine if the goals and objectives that were set are being achieved.

A number of factors have contributed to the need for a more systematic approach to conservation planning. The growing number of endangered species has highlighted that proactive conservation approaches are required to complement the reactive measures of species recovery programs (Groves *et al.* 2002). There is more recognition of the importance of conserving the underlying ecological processes that support the patterns of biodiversity and that conservation efforts must be at multiple spatial scales and must target appropriate levels of biological organisation (Balmford *et al.* 1998; Schwartz 1999). There has also been acknowledgment that previously used ad hoc or site-by-site approaches to planning have led to biased reserve networks with some ecosystems being over-represented and others under-represented (Groves *et al.* 2002; Margules and Pressey 2000; Possingham *et al.* 2008).

One benefit of systematic conservation planning is that it facilitates a transparent, inclusive and defensible planning process which allows decisions to be critically reviewed (Margules and Pressey 2000; Noss 2003; Possingham *et al.* 2008). It is a flexible planning process, and this characteristic is essential when faced with issues of competing land uses (Margules and Pressey 2000). The planning process is effective in identifying efficient reserve networks – those that meet conservation objectives for the least possible cost (Margules and Pressey 2000; Possingham *et al.* 2008). It is also considered ‘best practice’ as it incorporates other fundamental planning principles of comprehensiveness, spatial arrangement, complementarity and selection frequency (Possingham *et al.* 2008).

## Conservation planning within DEC

The Department of Environment and Conservation (DEC) is the lead agency for coordinating biodiversity conservation in Western Australia, and has statutory responsibility for the management of public land and waters for conservation and wildlife conservation. Within DEC, the Nature Conservation Division is responsible for various aspects of conservation of flora and fauna in Western Australia, including:

- the development and implementation of programs for the conservation of flora and fauna, threatened species, ecological communities and commercially exploited species
- the protection of representative ecosystems
- providing support for nature conservation services and policies (DEC 2006).

The Nature Conservation Division has primary responsibility for coordination of the Nature Conservation Service (NCS) (formally known as the Nature Conservation Output), which is involved in strategic conservation planning and identification of conservation priorities. The NCS has moved towards an outcome-based or value-driven management approach, where 'desirable biodiversity-related outcomes (conservation targets) drive the need for what activities and tasks should occur' (DEC 2009a; b). This new approach has also been united with work to better define priorities and targets for the NCS, and the adoption of an active adaptive management framework for major projects and programs (DEC 2009b).

The NCS is guided by the following key documents:

- the draft *A 100-year biodiversity conservation strategy for Western Australia*
- the draft framework *Biodiversity conservation appraisal system* (DEC 2009b)
- Nature Conservation Service region plans for each of the nine regions in DEC – discussed below.

The central element of this new approach is the development and implementation of the rolling five-year NCS regional plans for the department's nine regions. These plans are vertically integrated with the targets of the draft *A 100-year Biodiversity Conservation Strategy for Western Australia* and the *Nature Conservation Service Strategic Plan* (DEC 2009a).

An active adaptive management approach has been adopted to the delivery of some programs (DEC 2009a). This approach uses management programs established experimentally to compare policies or practices, by evaluating alternative hypotheses about the system being managed. The aim is to continually improve management policies and practices by learning from the outcomes of operational programs (DEC 2009b; MFR 2004).

The draft framework *Biodiversity conservation appraisal system* (DEC 2009b) has the potential to help in the efficient delivery of NCS programs. This draft framework measures and reports on biodiversity outcomes and effectiveness of management. It also clarifies a biodiversity conservation business model, within the DEC context, which will result insetting priorities that are better suited to an active adaptive management approach (DEC 2009b). It will also improve knowledge of biodiversity and management standards throughout the organisation (DEC 2009b).

The NCS regional plans clearly define priorities and targets for the service (DEC 2009a). The current Swan Region Plan sets out a strategic direction for biodiversity conservation for the DEC Swan Region, in which the GSS study area is located, for the period 2009–2014. The plan provides a strategic framework for the development of NCS operational strategic plans, annual works programs and adaptive management approach projects within the Swan Region (DEC 2009a). See Chapter 1 for more details on the policy framework that the Swan Region Plan fits into.

A brief summary of the planning process that was used for all of the nine Nature Conservation Service regional plans is given below.

The first planning stage involved the development of a 25-year aspirational goal, in line with the NCS goal of ‘conserving biodiversity’. Conservation targets were then identified for a number of asset classes such as landscape, protected area, wetland, ecosystems at risk and species at risk. This was done using a variety of tools and reference material from statutory and strategic plans, at the state-wide, regional and subregional levels, and by referring to threatened species and ecological communities and to recovery plans. As part of this, a value/threat matrix was constructed to help identify which threats were a priority for management and research. This process identified the ‘specific high priority biodiversity values requiring direct management at a range of spatial scales in order to recover unacceptable decline or maintain biodiversity’ (DEC 2009a).

Candidate actions were then assigned to each conservation target, including the strategy for implementation, and were described using *what, where, why* and *who* parameters. Costs of each candidate action were estimated and then prioritised for investment. Lead officers for each action were identified. Performance indicators and measures for candidate actions and milestones were then developed to assist in the prioritisation process and as an evaluative mechanism.

## Decision making tools for biodiversity conservation

Conservation agencies are increasingly drawing on decision theory or frameworks when determining what conservation actions to invest in so that the benefits, constraints, uncertainties and trade-offs can be explicitly stated. This improves transparency and ensures that the best long-term outcomes are delivered for the money invested (Possingham 2001; Possingham *et al.* 2002; Stephens *et al.* 2002; Walshe 2005). Decision theory involves any mathematical, economic, or social science that helps us make decisions (Wilson *et al.* 2008) and decision making tools are often used within decision theory (Possingham 2001). These include mathematical optimisation tools which are designed to 'provide the best solutions to well-defined problems' (Possingham 2001; Wilson *et al.* 2008). Alternatively, more qualitative tools can be used such as 'multi-criteria decision analysis and ecological risk assessment' (Possingham 2001) or cost–benefit analysis.

Decision theory approaches are considered to have merit as they provide frameworks to assess the relative worth of diverse conservation outcomes (Stephens *et al.* 2002) at both micro and macro scales (Possingham 2001); and can distinguish among, and integrate, various goals held by stakeholders (Possingham 2001). One such framework, detailed by Possingham (2001), suggests a process which includes first specifying the management objectives and management options. This is followed by specifying the system properties and developing a conceptual model of the dynamics of the system. Walshe (2005) considers that it is important to specify constraints and uncertainty by being clear about what we don't know. Finding solutions to the problem is the next aim and it is here that decision tools are used extensively.

Details of the functional relationships that relate assets to threats and management interventions to improvement in condition is required when using decision theory (Possingham 2001; Stephens *et al.* 2002) and often this information is not available. Surrogates can sometimes be used to overcome this constraint (Possingham *et al.* 2002). Active adaptive management approaches can be taken to build into the monitoring process the exploration of functional relationships between assets, threats and management interventions (Possingham 2001).

## Qualitative tools

### *Multi-criteria decision analysis*

A number of biological, social, political and economic factors need to be considered when ranking and prioritising natural areas. Therefore, conservation planners are often faced with multi-criteria decision problems involving a large number of criteria. A number of multi-criteria decision analysis methods and tools have been developed that can assist planners (Moffett and Sarkar 2006; Smith and Theberge 1987).

When ranking natural areas, criteria are employed so an assessment of the values of each area can be undertaken with respect to each criterion (Smith and Theberge 1986). The assessment can be either quantitative or qualitative and can be prepared at a range of scales such as local, regional or at the national scale (Smith and Theberge 1986; 1987). Criteria used to rank natural areas fall into one of four generic types:

- biotic or abiotic (Smith and Theberge 1986)
- biodiversity process – the ecological, evolutionary and genetic mechanisms necessary for genetic diversity and for speciation (Smith et al. 1993)
- socio-political – related to how humans use the landscape (e.g. recreational criteria), or value the landscape (e.g. cultural, economic, industrial resources) (from Moffett and Sarkar 2006; Smith and Theberge 1986)
- planning and management – e.g. management costs (Smith and Theberge 1986).

A summary of commonly used classes of criteria used to rank natural areas is provided in Table 11.1. The majority of these are abiotic and biotic criteria. Where possible, specific criteria should be developed that relate to the primary attributes of ecosystems including composition, structure and function (see Chapter 1 and Noss 1990). Criteria also need to be targeted at the multiple levels of biotic organisation and different spatial and temporal scales (Margules and Pressey 2000; Noss 1990). The hierarchy concept proposed by Noss (1990), and outlined in Chapter 1, provides a framework for this. Ideally criteria should be selected that cover the regional or landscape, community or ecosystem, and population or species levels of organisation (Chapter 1 Table 1.1).

The nature of the criteria used to rank natural areas is very much dependent on the availability of regional spatial data relating to biotic, abiotic and biodiversity processes. Generally, regional datasets of the spatial distributions of species and populations do not exist or are biased and therefore of limited use (Ferrier 2002; Margules and Pressey 2000; Pressey 2004). Biodiversity conservation planners therefore work with what information is available and often use surrogate measures of biodiversity (see Chapter 1; (Margules and Pressey 2000). Surrogates generally relate to the community or ecosystem level of organisation (e.g. vegetation types). The advantages with these types of surrogates are that data is readily available at the regional scale, they integrate biodiversity process (see comments below) and empirical studies have found that these broad environmental variables are good indicators of spatial patterns of species (Margules and Pressey 2000). The disadvantage is that they lose biological precision (Margules and Pressey 2000). Surrogates based on spatial distribution models for species that have been developed using available location data from museum or herbarium records are also increasingly been used (Margules and Pressey 2000) and in data poor regions modelling of collective properties of biodiversity is being considered (Ferrier 2002).

When ranking natural areas, higher order surrogates (often referred to as a coarse filters), such as vegetation types, are often used in combination with other criteria and datasets that relate to fine filter biodiversity features (Higgins *et al.* 2005; Lieberknecht *et al.* 2008; Margules and Pressey 2000; Moffett and Sarkar 2006). Examples of these include the location of threatened species or ecological communities, endemic species or special habitats which are not adequately represented by the higher order surrogates (Lieberknecht *et al.* 2008; Margules and Pressey 2000). The advantage of this approach is that these fine filter datasets provide a greater degree of precision thereby refining the coarse filter assessment.

Table 11.1: Commonly used criteria used in the ranking of natural areas.

| Criteria | Examples of the criteria's use in regional planning across the Swan Coastal Plain |
|---|---|
| Rarity | • Bush Forever – 'considered from an ecological community and individual species perspective' (1)<br>• Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |
| Diversity | • Bush Forever – 'richness, diversity or complexity for their physical or biological attributes at the community, species or genetic level' (1)<br>• Local government biodiversity planning guidelines for the PMR – considered in a general way only by considering upland and wetland structural plant communities (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3) |
| Representativeness | • Bush Forever – 'representation of each floristic community type within each vegetation complex' and 'each natural wetland group and wetland types within each wetland group' (1)<br>• Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |
| Maintenance of ecological processes | • Bush Forever – large areas with relatively intact natural processes (1)<br>• Local government biodiversity planning guidelines for the PMR – considered in regard to connectivity (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3) |
| Productivity | • First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |
| Fragility/Stability | • Project Dieback uses susceptibility to *Phytophthora cinnamomi* (5) |
| Importance for wildlife | • Bush Forever – fauna habitats specific for feeding/breeding/nursery functions, wildlife corridors and habitats for significant populations of migratory birds (1)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |
| Size | • Bush Forever – lower size limit of 20 ha used though smaller areas were considered if a complex or community was threatened or poorly reserved (1)<br>• Local government biodiversity planning guidelines for the PMR (2) |
| Shape | • Bush Forever – compact shape is preferable to an irregular or an elongate shape (1)<br>• Local government biodiversity planning guidelines for the PMR (2) |
| Condition | • Bush Forever – order of preference for remnants were those that were (summarised from 1):<br>1. largely undisturbed<br>2. basic vegetation structure intact<br>3. in lesser condition but able to be regenerated<br>• Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3) |
| Threat | • Project Dieback – Autonomous spread of *P. cinnamomi*, proximity to infested areas, density of roads (5) |
| Naturalness | • Bush Forever<br>• Second – tier evaluation of wetlands in Wedge Island to Mandurah area (3) |
| Educational value | • Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |
| Historical significance | • Bush Forever (1)<br>• Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |
| Scientific value/research investment | • Bush Forever – scientific or evolutionary importance (1)<br>• Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |
| Recreational value | • Bush Forever – 'area is a regional recreation resource' (1)<br>• Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |

| Criteria | Examples of the criteria's use in regional planning across the Swan Coastal Plain |
|---|---|
| Ecosystem services | • Bush Forever – wetlands are also recognised for the role they play in maintaining ecological functions associated with the hydrological cycle, and river foreshores and coastal vegetation are recognised for the role they play in maintaining stability in these environments (1)<br>• Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |
| Icon species or ecological communities | • Local government biodiversity planning guidelines for the PMR (2)<br>• First-tier evaluation of wetlands in Wedge Island to Mandurah area (3 and 4) |

(1) Government of Western Australia (2000)
(2) Del Marco *et al.* (2004)
(3) Hill *et al.* (1996)
(4) Leprovost *et al.* (1987)
(5) Strelein *et al.* (2008)

Previous regional conservation planning exercises across or within the Swan Coastal Plain IBRA Region have generally used a combination of coarse filter and fine filter criteria and surrogates to identify areas of high biodiversity value. Criteria relating to the level of representation of vegetation complexes or wetland groups are often used as the coarse filter whilst criteria relating to rarity, diversity, maintenance of ecological processes, productivity, size, shape and condition provide the fine filter (Table 11.1). As in most regions, a lack of adequate data on species distributions at a regional scale limit how the population or species level of organisation can be considered. The majority of previous conservation planning exercises in the Swan Coastal Plain also included a number of socio-political criteria in their rankings (Table 11.1). These have not been used to determine the regional biodiversity significance of the natural area but rather to rank natural areas which have similar biodiversity values or levels of significance (Government of Western Australia 2000).

In recent years there has been a greater recognition of the importance of conserving biotic processes required to produce and maintain biodiversity pattern (Pressey 2004; Pressey *et al.* 2003; Smith *et al.* 1993). Therefore, criteria relating to biodiversity processes are now being included in multi-criteria analyses that rank natural areas. In a review of recent studies, undertaken by Pressey *et al.* (2003), four approaches to including biodiversity process have been identified:

- Incidental – by considering only biodiversity pattern some processes that do not need large areas are likely to persist even when not explicitly targeted. This approach ignores important population and ecological processes.

- Generic design criteria – e.g. size, shape, connectivity. The approach can assist to maintain processes such as disturbance regimes, but their effectiveness is limited by the fact they don't consider requirements of specific processes.
- Process-specific design criteria – parameterising the generic design criteria with quantitative requirements for persistence of specific processes. This requires adequate (functional) information. Parameters could be related to natural disturbances (setting minimum size), spatial requirements for select species or defining core geographical range for species where persistence is more likely.
- Specific spatial attributes associated with processes – identifies locations defined by specific physical or climatic features, associated with processes of interest (e.g. refugia).

On the Swan Coastal Plain biodiversity processes have been considered largely based on incidental and generic design criteria. Biodiversity pattern has been considered through levels of representation of vegetation complexes and wetland types (Del Marco *et al.* 2004; Government of Western Australia 2000; Hill *et al.* 1996). Design criteria relating to the size of intact natural areas and connectivity of remnant vegetation or linked wetland systems have also been employed (Del Marco *et al.* 2004; Government of Western Australia 2000; Hill *et al.* 1996). To enable approaches based on process-specific design criteria and specific spatial attributes associated with processes to be used on the Gnangara Mound information will need to be compiled relating to the parameters. Surrogate measures will then need to be identified and spatial datasets relating to these will need to be developed.

### Evaluation methods for ranking natural areas

A number of multi-criteria evaluation methods can be used to combine the scores relating to individual criteria into a single composite index which can then be used to rank natural areas (Moffett and Sarkar 2006; Smith and Theberge 1987). The development of composite indices have a number of benefits including reducing the large amount of information relating to biodiversity features to a single measure, thereby providing new perspectives on biodiversity pattern and process, and also making information on biodiversity value more accessible to non-specialists (Smith and Theberge 1987).

Any evaluation method used to rank natural areas must have ecological and mathematical validity (Smith and Theberge 1987). The scale of measurement used to assess the criteria will determine what evaluation methods can be used. A number of evaluation methods can be used when criteria are measured using quantitative data. However, there are more limitations when only qualitative data is available (Moffett and Sarkar 2006; Smith and Theberge 1987). A number of assumptions specific to each evaluation method may have to be met such as independence of criteria (Moffett and Sarkar 2006 (see Appendix 1); Smith and Theberge 1987).

In regional conservation planning two of the most commonly used evaluation methods are 'additive weighting', which ranks alternatives based on the sum of the criteria scores (Smith and Theberge 1987), and the 'Maximax' decision model, which ranks alternatives based on the highest score across all criteria (Hwang and Yoon 1981). Requirements and assumptions for both of these models are listed in Kinloch (in prep.). Project 1 in this chapter assesses the effectiveness of these two evaluation methods for ranking the biodiversity assets across the GSS study area. It should be noted that both of these evaluation methods rank (or score) natural areas in isolation and cannot evaluate the contribution that a site makes to the overall reserve network or to meeting regional biodiversity targets (Wilson *et al.* 2008). In conservation planning, information on what minimum set of natural areas are required to represent all species and biodiversity features is required (Possingham *et al.* 2008).

## *Ecological risk assessment*

The objectives of risk analyses are to identify the source of risk, the likelihood of occurrence, and the magnitude of the consequences (OGTR 2005). Analyses can be qualitative or quantitative depending on the type and quality of data or information available, and the costs of undertaking the analyses. 'Ecological risk assessment' has been defined as the process of estimating likelihoods and consequences of the effects of human actions or natural events on plants, animals and ecosystems (Barnthouse and Suter 1986).

Fundamental issues in conservation planning and natural resource management involve the evaluation of the loss or degradation of conservation and biodiversity values. Risk assessment can be employed to determine the best management strategies to minimise risk, prioritise limited resources for the best outcomes and to identify knowledge gaps that require research or monitoring (Walshe 2005). Risk assessments have been developed to estimate the risk of human activities, and other threatening processes (e.g. pest plants and animals, inappropriate fire regimes and climate change) on flora, fauna, communities and ecosystems (Barnthouse and Suter 1986; Hart *et al.* 2003; Suter II 1993). Analyses can assess the likelihood and extent of effects and thus provide a basis for comparing and prioritising risks so that managers can make informed decisions.

The benefits of undertaking risk assessment in natural resource management include:

- transparency of the process
- clear documentation (so that changes in personnel or advances can be applied to work undertaken previously)
- informed decision making
- input into priority setting
- reduced costs for environmental practitioners by using standard methods (Beer and Ziolkowski 1995; Burgman 2001; Carey *et al.* 2004).

There are a number of problems facing ecological risk assessments. Estimating the likelihood and consequence of various hazards (threats) is difficult due to the lack of detailed understanding of many ecological systems (Walshe 2005). In particular a lack of quantitative data is a major problem, so that many of the models of stressor effects on values are semi-quantitative with descriptive ratings for assessing risk. These assessments are thus faced with subjective influences and also high levels of uncertainty (Carey *et al.* 2004; Hart *et al.* 2003; Walshe 2005). Many ecological risk assessments focus on single species and stress factors. However, when dealing with risk to communities there are increased complexities (Beer and Ziolkowski 1995; Hayes 2002a; b).

Steps in the continuous improvement cycle underpinning the Australian risk assessment standard (AS/NZ 4360; AS/NZS 2004) include:

- establishing the context by identifying important ecological values and defining the scope of the assessment
- identifying relevant hazards, threats or stressors

- analysing the risks by assessing the consequences and likelihood for each of the hazards
- evaluating the risks by comparing, ranking and prioritising them in terms of their seriousness with respect to the management objectives identified in the initial problem formulation.

The Australian Standard provides a risk analysis matrix which defines the risk of a hazard as the product of its consequence and its likelihood (Table 11.2) (Walshe 2005). To utilise this matrix, the probability of the hazard being present (likelihood) and its likely impact on biodiversity values (consequence) must be assessed and assigned a ranking number. Risk is then determined by multiplying the likelihood and consequence ranking numbers. The risk score is then assigned to one of three risk categories (low, moderate or high risk, Table 11.2). Hart *et al.* (2005) recommends that a quantitative assessment of uncertainty and risk be undertaken when a qualitative assessment (using the AS/NZ 4360 standard) indicates a high risk, or where there is disagreement amongst experts on the importance of a hazard.

Table 11.2: Semi-quantitative descriptors of consequence and likelihood used to rank risk. A scale of five levels is used to describe the likelihood and consequence of a hazard. Unshaded = low risk, light grey = moderate risk, dark grey = high risk (Walshe 2005)

| **Likelihood** | | **Consequence** | | | | |
|---|---|---|---|---|---|---|
| | | Insignificant **(1)** | Minor **(2)** | Moderate **(3)** | Major **(4)** | Catastrophic **(5)** |
| Almost certain | **(5)** | 5 | 10 | 15 | 20 | 25 |
| Likely | **(4)** | 4 | 8 | 12 | 16 | 20 |
| Moderately likely | **(3)** | 3 | 6 | 9 | 12 | 15 |
| Unlikely | **(2)** | 2 | 4 | 6 | 8 | 10 |
| Rare | **(1)** | 1 | 2 | 3 | 4 | 5 |

As mentioned previously, assessing the likelihood and consequence of a hazard can be difficult due to lack of knowledge and the inherent variability of natural systems and therefore variability in how they are affected by threats (Walshe 2005). Conceptual models should be used to document assumptions about cause and effect and the models should preferably be quantified so the uncertainty in the risk assessment can be clearly communicated (Hart *et al.* 2005). Conceptual models that can be used include logic trees (including decision trees, event trees and fault trees) and Bayesian belief networks (Walshe 2005).

### Case study of *Phytophthora* dieback

The disease in natural systems caused by *P. cinnamomi* is listed as a key threatening process under the EPBC Act. The National Threat Abatement Plan for *Phytophthora* was developed with the major objectives being to promote the recovery of threatened species and ecological communities under threat, and to limit spread of the pathogen (CPSM 2006; Environment Australia 2001). Projects to address these objectives have developed processes and criteria to assess the risk to biodiversity (Wilson *et al.* 2005), and provide national best practice benchmarks for management (O'Gara *et al.* 2005). The essential requirements for risk assessment and management of disease are: accurate knowledge of where the disease occurs (see Chapter 9 project 2), which species and communities are threatened (see Chapter 9 projects 1 and 4), and where risks and consequences of infestation are likely.

Although Standards Australia (AS/NZ 4360; AS/NZS 2004) describes risk as being a measure of consequences and likelihood, the term is used inconsistently in *P. cinnamomi* management, and often only describes the probability of an event such as pathogen transmission and/or impact. A range of risk processes have been developed in Australia, using indicators of consequence and/or likelihood, to assist in planning or in setting priorities for the management of *P. cinnamomi*. These are described below and a *Phytophthora* dieback risk assessment for the GSS has been developed using the Australian Standard (see project 2 of this chapter).

A risk assessment process was developed for national adoption through the National Threat Abatement Plan for *P. cinnamomi* (Wilson *et al.* 2005). It includes procedures for assessing the risk of *P. cinnamomi* to threatened species, ecological communities and areas, and for ranking them as the basis for setting management priorities. The procedures identify the source of risk, the likelihood of occurrence, and the magnitude of the consequences to biodiversity. The processes are semi-quantitative (i.e. qualitative criteria are assigned scores) and are based on current scientific knowledge or expert opinion. The process is viewed as iterative, and reviews are recommended to be undertaken as new data and knowledge become available.

The potential distribution and impact of *P. cinnamomi* in Victoria was assessed for Parks Victoria (Gibson *et al.* 2002). This assessment included development of a strategic level map showing the potential distribution and impact of the pathogen across the state. The risk map was constructed with GIS overlays of:

- topographic and climatic parameters suitable for the pathogen
- known distribution of the pathogen
- distribution of susceptible species
- the distribution and density of roads and tracks as a surrogate for the probability of pathogen transmission (Gibson *et al.* 2002).

The risk classification system has been incorporated into the Parks Victoria environmental management system (Parks Victoria 2004).

In Western Australia, ‘Project Dieback’ was developed at a regional scale and strategic level, to protect biodiversity of areas assessed as significant, and at risk from dieback caused by *P. cinnamomi* (Strelein *et al.* 2008). The project involved the identification of significant disease-free areas, an assessment of the risks of introduction of *P. cinnamomi* into these areas, mapping of disease symptoms in infested areas, and risk analysis of the relationship between biodiversity assets and threats. The risk analysis was based on the collation, modelling, analysis and overlay of spatial data themes using the GIS forest management information system. Asset scores and threat scores were assigned, based on a set of ranking criteria, and were then employed as part of a decision matrix of appropriate management actions.

## *Cost benefit analysis*

Cost-benefit analysis is widely used in a number of fields. The process estimates the economic costs and benefits of a proposed action, project or policy to inform decision making (Arrow *et al.* 1996; Naidoo and Ricketts 2006). This type of analysis can help determine whether the aggregate benefits of a particular action outweigh the aggregate costs (Arrow *et al.* 1996; Naidoo and Ricketts 2006) and can be utilised to compare the cost-effectiveness of quite diverse actions or projects.

Until recently conservation planners have only considered biological data and information on threats when identifying conservation priorities. They are now starting to incorporate information on economic costs by using cost-benefit approaches (Naidoo and Ricketts 2006; Wilson *et al.* 2007). This change in practice will lead to a more effective and transparent approach to allocating conservation investment as greater understanding of costs and benefits is developed. Naidoo and Ricketts (2006) outlined a number of broad costs and benefits associated with conserving biodiversity. These include the costs associated with protecting and sustaining biodiversity (e.g. land purchase, management costs, rehabilitation and invasive species control) and, opportunity costs (e.g. foregone alternatives including alternative land uses). Two categories of benefits are recognised including the direct benefits of protecting species and ecosystems and indirect benefits in the form of ecosystem services (e.g. flood control from wetlands and carbon sequestration from woodlands). Quantifying the costs and benefits is quite complex and is an area of current research (Smith 2000 and Turner et al. 2003 cited in Naidoo and Ricketts 2006). Spatially explicit data on costs and benefits will be of the greatest benefit to conservation planning but the availability of such data is still limited (Naidoo and Ricketts 2006). This would enable 'win–win' areas (areas of high biodiversity value and high economic benefit from conservation) to be identified more easily as well as areas where trade-offs will need to be considered (areas of high biodiversity value and low economic benefit from conservation; Naidoo and Ricketts 2006).

Cost-benefit analysis in combination with a decision analysis approach was used by Possingham *et al.* (2002) to identify priority actions for sustaining biodiversity. This assessment was undertaken at the national scale and involved:

- outlining the risks and management options for each threatening process
- quantifying the risk in terms of the number of species under threat
- calculating the biodiversity benefit by estimating the effectiveness of the management options
- estimating the financial cost of each management option
- converting this to a cost per species secured
- estimating the value of the indirect benefits of each management option.

Management options were then ranked according to the number of species secured for a fixed financial investment and the ratio of the indirect benefits to total financial cost.

Another example of the application of cost-benefit analysis in biodiversity conservation is the ‘Conservation investment framework’ developed by Wilson *et al*. (2007). The approach taken is similar to that used by Possingham *et al.* (2002) in that it focuses on identifying actions to abate threats and then on costing these actions. Benefits are defined as the number of plant and vertebrate species predicted to persist in the region after investment in an action and are termed ‘biodiversity benefits’. A ‘maximise short-term gain’ heuristic (rule of thumb) is used to determine the optimal investment schedule by identifying which actions ‘provide the greatest short-term increase in biodiversity benefit per dollar invested’ (Wilson *et al.* 2007). The conservation investment framework approach was used by Wilson *et al*. (2007) at the regional scale to prioritise investment for three possible management actions (revegetation, *Phytophthora* dieback management and invasive predator control) on the Swan Coastal Plain. The analysis revealed that the most cost-effective way to deliver biodiversity benefits would be to initially invest all funds into *Phytophthora* dieback management. Such analyses would need to be repeated prior to each round of funding, as the initial investment in a particular action is likely to have provided biodiversity benefits. These benefits need to be factored in as they could change management priorities.

## Quantitative tools

### *Optimisation*

The first quantitative approaches to identify ‘preferred’ or ‘the best’ candidate sites for reservation were based on a numerical scoring to rank sites in terms of multiple criteria (Possingham *et al.* 2008; Smith and Theberge 1986; Williams *et al.* 2004). Usually sites with the highest scores were nominated as ‘the best’ sites for reservation (Possingham *et al.* 2008; Smith and Theberge 1987). The drawbacks with this approach are that it takes a large number of sites to represent all species or features as the highly ranked sites often contain similar species, so a large number of sites will need to be reserved in order that all species or features are represented (Williams *et al.* 2004). Further, sites are ‘scored’ or ‘valued’ in isolation, when in fact for reserve networks the concept that the ‘whole is more than the sum of the parts' needs to be incorporated through an assessment of how each site contributes to meeting regional biodiversity targets (Wilson *et al.* 2008).

An alternative approach is to use decision support tools that use optimisation algorithms that determine the minimum number of sites, or minimum total area, necessary to represent all species or biodiversity features (Possingham *et al.* 2008). These decision support tools integrate mathematical programming and spatial design criteria and are able to deal with complementarity when determining the optimum reserve network that can meet user-defined biodiversity targets (Possingham *et al.* 2008).

One such decision support tool is 'Marxan' (Ball and Possingham 2000). It uses heuristic (non-exact) algorithms ('simulated annealing') to determine 'a number of good, near-optimal solutions' that planners and stakeholders can then consider (Possingham *et al.* 2008) . Sites (referred to as planning units in Marxan analysis) that are selected more than 50% of the time are often considered important for efficiently meeting biodiversity targets (Possingham *et al.* 2008). In conservation planning, heuristic algorithms are currently preferred as they can provide a range of solutions to complex reserve design problems in a timely manner (Possingham *et al.* 2008).

The Marxan decision support tool can be used to answer questions such as: what are the gaps in the current reserve system and how efficient is it; what are the priorities and options for filling the gaps identified; and what are the socio-economic costs to meeting conservation objectives (Martin *et al.* 2008). This ability to assess not only ecological, but also economic and social considerations is important for any conservation planning exercise.

# Summary of DEC–GSS projects (2007–09)

## Project 1: Ranking of biodiversity assets across the GSS study area

Technical report reference:

1) Kinloch, J. (in prep.) *Criteria and ranking of biodiversity assets in the GSS study area*. Unpublished report prepared by the Department of Environment and Conservation for the Gnangara Sustainability Strategy, Perth.

2) Eco Logical Pty Ltd. (November 2008) *Gnangara conservation significance assessment for the Gnangara Sustainability Strategy*. Unpublished final report to Western Australian Department of Environment and Conservation, Perth.

Ranking of biodiversity assets and identifying the risks that threatening processes pose to these assets are essential steps in the process of formulating conservation objectives and priorities for the GSS study area. The objectives of this project were to use and compare two multi-criteria evaluation models to rank the biodiversity assets in the study area. The models used were the ‘Summed weighted rank model’ and the ‘Highest weighted rank model’.

The rankings were based on readily available spatial data on biodiversity assets and for both models expert input was sought during the development of the criteria, the scaling, scoring and weighting of criteria and the definition of significance categories (Eco Logical Pty Ltd 2008; Kinloch in prep.). The methods and criteria used in each of the models are summarised in Table 11.3. In both models a combination of coarse filter criterion (e.g. representation of vegetation complexes or conservation status of wetland) and fine filter criterion (e.g. presence of threatened flora, fauna and threatened ecological communities) were employed to rank biodiversity assets, which meant that some of the criteria were not independent.

Table 11.3: Summary of methods, criteria and scoring used in each model.

| | **Summed weighted rank model (SWRM)** | **Highest weighted rank model (HWRM)** |
|---|---|---|
| **Methods** | | |
| Measurement scales of biodiversity asset data | Interval | Interval |
| Brief description of ranking procedures undertaken in the GIS | Vector based GIS. Data layers for each asset were combined and attributed to indicate what assets were present in each polygon and the final rank for each polygon. | Raster based GIS. Data layers for each asset was converted to a 100 m grid using the Spatial Analyst and the Cell Statistics function (maximum) calculated the final rank for each 100 m grid cell. |
| Evaluation method | Additive weighting | Maximax (Moffett and Sarkar 2006; Smith and Theberge 1987) |
| Criterion score range | 1 (lowest) to 5 (highest)<br>The weightings (scores) reflect the significance that the experts place on the asset (Eco Logical 2008). Final scores ranged between 0 (no assets) and 33 (highest). | 1 (highest) to 10 (lowest)<br>The weightings (scores) reflect the overall value that the experts place on the asset. Final scores ranged between 1 (highest) and 10 (lowest). |
| Significance categories | The final scores were summarised into six significance categories | The final scores were summarised into five significance categories. |
| **Criteria** (broad criterion type as per Table 11.1) | | |
| **Ecological community status** (rarity and representativeness) | Criterion relating to degree of threat to threatened and priority ecological communities.<br>Criterion relating to vegetation community (complex) status (VC asset dataset version 1[1]). | Criterion relating to degree of threat to threatened and priority ecological communities and whether they are restricted to the GSS.<br>Criterion relating to vegetation community (complex) status (VC asset dataset version 2[1]). |
| **Terrestrial flora rarity and endemicity** (Rarity) | Within 500 m of an existing threatened (declared rare flora or priority) flora record. | Criterion relating to degree of threat of declared rare flora and priority flora and whether the species is locally[2] or regionally[3] endemic. |
| **Fauna rarity and endemicity** (Rarity) | Criterion relating to the known habitat and EPP area for the critically endangered western swamp tortoise (Government of Western Australia 2003). | Criterion relating to the known habitat and EPP area for the critically endangered western swamp tortoise (Government of Western Australia 2003). |
| **Wetlands** (Representativeness, Productivity, Importance For Wildlife, Condition, Ecosystem services, Icon Ecological Communities) | Criterion relating to the conservation status of wetlands (wetlands of national importance or conservation category, resource enhancement or multiple use wetlands) | Criterion relating to the conservation status of wetlands (Ramsar wetlands, wetlands of national importance or conservation category, resource enhancement or multiple use wetlands) |
| **Connectivity/ecological linkages** (maintenance of ecological processes, size) | Within a plantation linkage (Brown *et al.* 2009) | Criterion relating remnant vegetation cover over a 2 $km^2$ area. Based on thresholds identified for sensitive avifauna (Brooker *et al.* 2008).<br>Criterion relating to the GSS ecological linkages (Brown *et al.* 2009). |
| **Bush Forever sites** (all criteria) | Bush Forever site or Northern Crown Reserve | Covered in second ecological linkage criterion above. |
| **Remnant vegetation patch size** (size) | Criterion relating to Patch size | Not included (covered in first ecological linkage criterion). |

[1] VC asset dataset version 1 was undertaken prior to the completion of Kinloch *et al.* (in prep.) so criteria and weightings are slightly different to VC asset dataset version 2
[2] local endemic – refers to a species or community which is only found within the Swan Coastal Plain IBRA region
[3] regional endemic – refers to species or community which is only found within the South Western Australian Floristic Region

The ranking of the biodiversity assets using the summed weighted rank model reveal that 15.2% of the study area has biodiversity assets rated as being of extremely high or very high significance and 28% as being of high significance (Table 11.4a and Figure 11.1a). In comparison the highest weighted rank model revealed that 43% of the Gnangara Mound has biodiversity assets which are rated as being of very high significance (very high or extremely high in Table 11.4b and Figure 11.1b). These areas have been identified as being significant as they scored highly on criteria relating to ecological community status, terrestrial flora rarity and endemicity, fauna rarity and endemicity and wetlands in both models, remnant vegetation patch size in the summed weighted rank model and connectivity/ecological linkages in the highest weighted rank model (Eco Logical Pty Ltd 2008; Kinloch in prep.).

The majority of areas in the aforementioned top significance categories for both models are located in the north and central areas of the study area where there has been limited agricultural or industrial development (Figure 11.1a and b, Eco Logical Pty Ltd 2008). However, small, usually isolated, pockets of land that are ranked in the top significance categories do occur in the developed areas in the south, east and far north (Figure 11.1a and b). Even though these are surrounded by cleared areas they often represent one of the last remaining areas for a particular species or ecological community and therefore are significant.

The similarity in results between the two models was expected as the criteria used in both models were similar and the biodiversity assets used were either the same or very similar.

Limitations of a number of the base mapping datasets used in this ranking created some artificial boundaries to the significance classes in both models. These included the circular or uniform buffers for threatened flora and ecological communities (Figure 11.1a and b) and straight line (tenure) boundaries that divide contiguous vegetation (affects the summed weighted rank model only – see Figure 11.1a).

Table 11.4: Extent and proportion of land area in each of the significance categories for the (a) Summed weighted rank model and (b) Highest weighted rank model.

| **Significance category** | **Total area of the Gnangara Mound ha** | **Proportion of the Gnangara Mound %** |
|---|---|---|
| **(a)** | | |
| No assets | 69 537 | 32 |
| Very low (score 1–2) | 24 402 | 11 |
| Low (score 3–4) | 13 457 | 6 |
| Moderate (score 5–6) | 15 023 | 7 |
| High (score 7–9) | 60 162 | 28 |
| Very high (score 10–19) | 31 837 | 15 |
| Extremely high (score 20–28) | 421 | 0.20 |
| **Total** | **214 839*** | **100** |
| **(b)** | | |
| No assets | 72 400 | 34 |
| Low | 9 838 | 5 |
| Moderate | 51 | 0.024 |
| High | 39 773 | 19 |
| Very high | 86 131 | 40 |
| Extremely high | 6 704 | 3 |
| **Total** | **214 896** | **100** |

* Note: the summed weighted rank model analysis was undertaken by consultants prior to the boundary of the GSS study area being finalised. Therefore the total area of the Gnangara Mound reported for this model is slightly smaller than that reported in highest weighted rank model. The majority of the areas not included are open water.

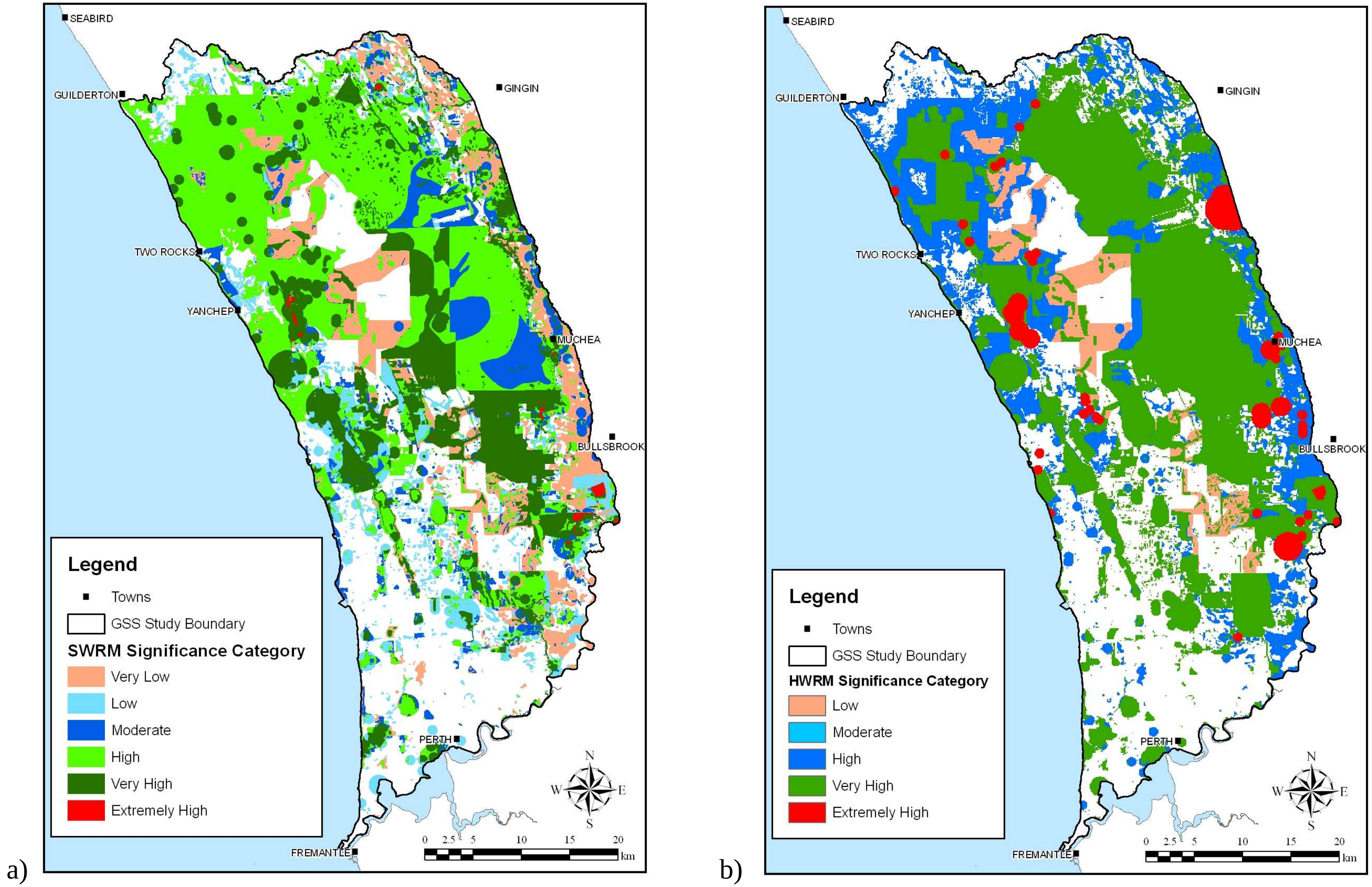



Figure 11.1: Ranking of biodiversity assets across the Gnangara Mound using (a) summed weighted rank model and (b) highest weighted rank model.

A direct comparison of the ranking of biodiversity assets in the two models is not possible as not only were the evaluation methods different but also the criteria and some of the base asset datasets. Despite this, some of the differences in the rankings can be directly related to the type of evaluation model used, highlighting the strengths and weaknesses of each. The summed weighted rank model has been found to be more effective than the highest weighted rank model at discriminating areas which contain multiple high value biodiversity assets from those areas which contain only one. One of the drawbacks of the summed weighted rank model evaluation method is that potentially areas ranked in the very high or high significance categories may not be important in respect to any individual criteria. This is not a problem for the highest weighted rank model where we can be certain that those areas in the significance category of very high all have at least one high ranking biodiversity asset present. Another drawback of the summed weighted rank model evaluation method is that it is more susceptible to skewing the ranks towards those areas where there has been a greater amount of biological survey effort.

One purpose in undertaking this type of evaluation is to reduce the large amount of information on biodiversity assets to a single rank or index. Another purpose is to provide new perspectives and insights into what areas have significantly high biodiversity values (Smith and Theberge 1987). Both models presented have been successful in achieving these aims and the analysis of the pros and cons of each above indicate that neither model has outperformed the other. Rather, both can provide valuable insights into the location, extent and significance of biodiversity values across the Gnangara Mound.

Further development of the criteria is required and consideration should be given to including criteria relating to the shape of remnant vegetation patches (e.g. using an index such as the perimeter to area ratio). Additional surrogate measures of biodiversity would certainly strengthen the criteria, especially those relating to both terrestrial and aquatic species and ecological community diversity, threats, productivity, condition and ecological processes, but at this stage the lack of available spatial data prevent their inclusion. Since a combination of coarse and fine filter criterion are required on the Swan Coastal Plain the independence of criterion also needs to be taken into consideration. Further development of some of the spatial data used in these models is also required.

Whilst both models provide valuable insights into what areas have significantly high biodiversity values, one flaw in them is that they score sites in isolation (Wilson *et al.* 2008) and therefore cannot provide insights into what are the minimum number of sites required to represent all species and biodiversity features (Possingham *et al.* 2008). To answer this type of question and others decision support software such as Marxan (Ball and Possingham 2000) will need to be used. The development of biodiversity surrogates and spatial layers for these two models will act as good groundwork for any future Marxan analysis.

## Project 2: *Phytophthora* dieback risk assessment of Gnangara mound biodiversity assets

Technical report reference:

Kinloch, J. and Wilson, B. (in prep.,). *Phytophthora dieback risk assessment of Gnangara Mound biodiversity assets*. Unpublished report prepared for the Department of Environment and Conservation and the Gnangara Sustainability Strategy, Perth.

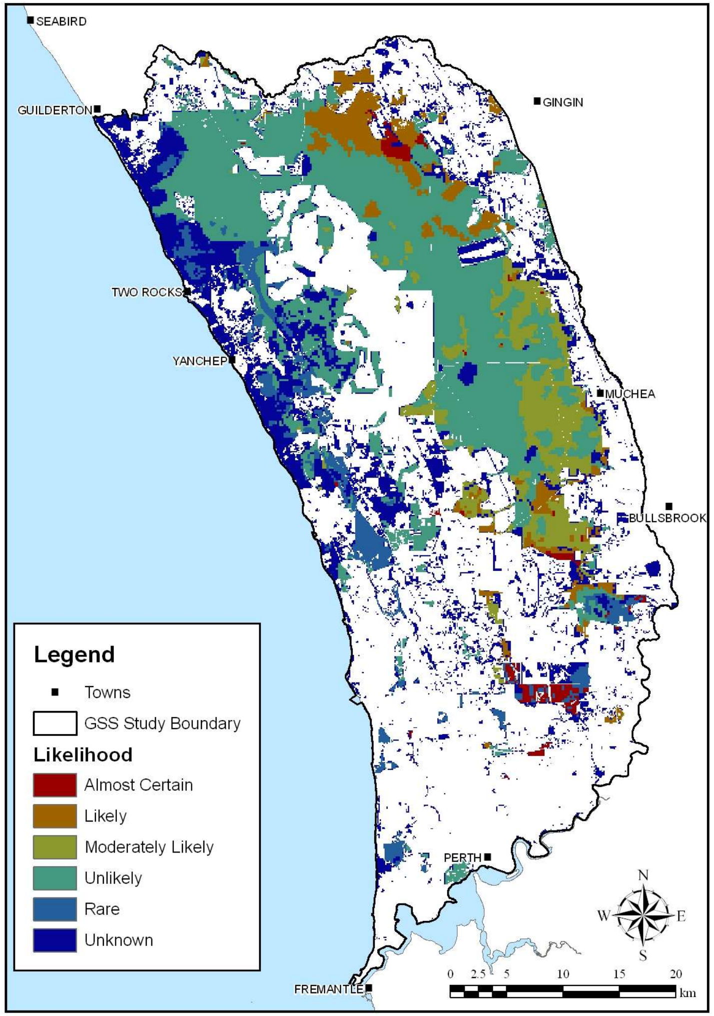
Identifying the likelihood and consequences of key threatening processes on biodiversity assets and determining the most appropriate and feasible conservation actions to abate threats are essential steps in the process of formulating conservation priorities. *Phytophthora cinnamomi* has been identified as a major threatening process on the study area with the potential to have serious negative impacts on flora, fauna and ecosystems. The aims of this project were to determine the extent of areas at high to moderate risk of being affected by *P. cinnamomi*, identify specific management actions and calculate the total area costs for cost-effective management of the disease.

### ***Risk assessment***

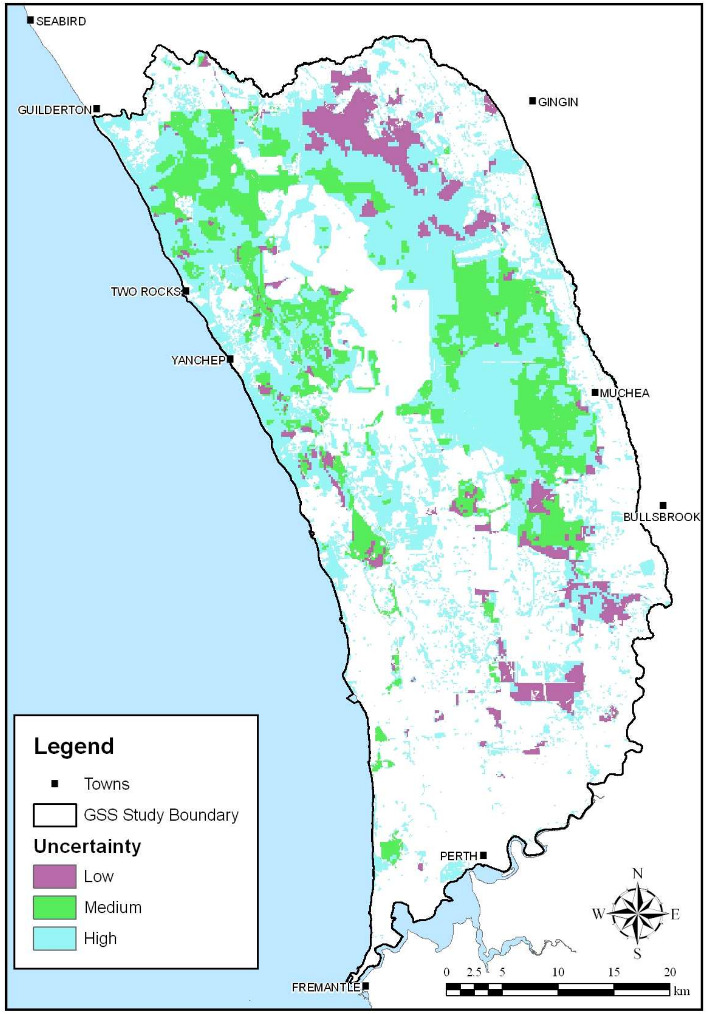
The *Phytophthora* dieback risk assessment was undertaken using the Australian Standard 4360 where the risk of a hazard (threat) is defined as the product of its consequence and likelihood and standard semi-quantitative descriptors of likelihood and consequence are used (AS/NZS 2004; Walshe 2005). The risk assessment used information on biodiversity assets from the ‘Highest weighted rank model’ (Project 1 above) and our analysis of perimeter to area ratio of remnant vegetation patches (Chapter 6 Project 1).

Likelihood was defined as the likelihood that an area is currently infested with *Phytophthora* dieback. Usually, risk assessments assess the likelihood of a hazard affecting biodiversity assets in the future. Unfortunately no reliable spatial information relating to the future likelihood of *Phytophthora* infection currently exists for the whole of the Gnangara Mound. However, the Project Dieback interpretation data (DEC 2008a) provides spatial information on the current likelihood of *Phytophthora* infection across the Gnangara study area. Ratings of the likelihood of current infestation were developed from information from Project Dieback (interpretation mapping categories and susceptibility assessments of Beard's vegetation types; (DEC 2008a; b) and applied to AS/NZ (2004) likelihood categories by an expert panel (see Kinloch and Wilson in prep. for details). The majority of the areas where *Phytophthora* dieback is likely to be currently present (almost certain, likely or moderately likely categories) are in the Bassendean sands soil types (Figure 11.2a). No current mapping exists for a large proportion of the study area and this was classed as 'unknown'.

It is important in risk assessments to be explicit about the level of certainty on estimates of the likelihood of hazards (Walshe 2005). Therefore a rating of uncertainty was also estimated using information on the level of confidence of the Project Dieback interpretation data and whether the susceptibility of the Beard's vegetation type to dieback was known (see Kinloch and Wilson in prep. for details). Uncertainty is low for only a small proportion of the Gnangara Mound revealing the limited extent of operational interpretation mapping that has been undertaken (Figure 11.2b).




a)




b)

Figure 11.2: (a) Likelihood that an area is currently infected with *Phytophthora* dieback and (b) the uncertainty in this assessment

*Phytophthora* dieback has the potential to highly modify or destroy susceptible vegetation communities, but the consequence of the loss will vary depending on the biodiversity values (at the species, community or landscape level) that are sustained by these ecological communities. The likely impact (consequence) of *Phytophthora* dieback in regard to overall loss of biodiversity values was assessed using data from two different sources:

- rankings of biodiversity assets from the highest weighted rank model, Figure 11.1b
- perimeter to area ratio data of remnant vegetation patches (Chapter 6 Figure 6.3).

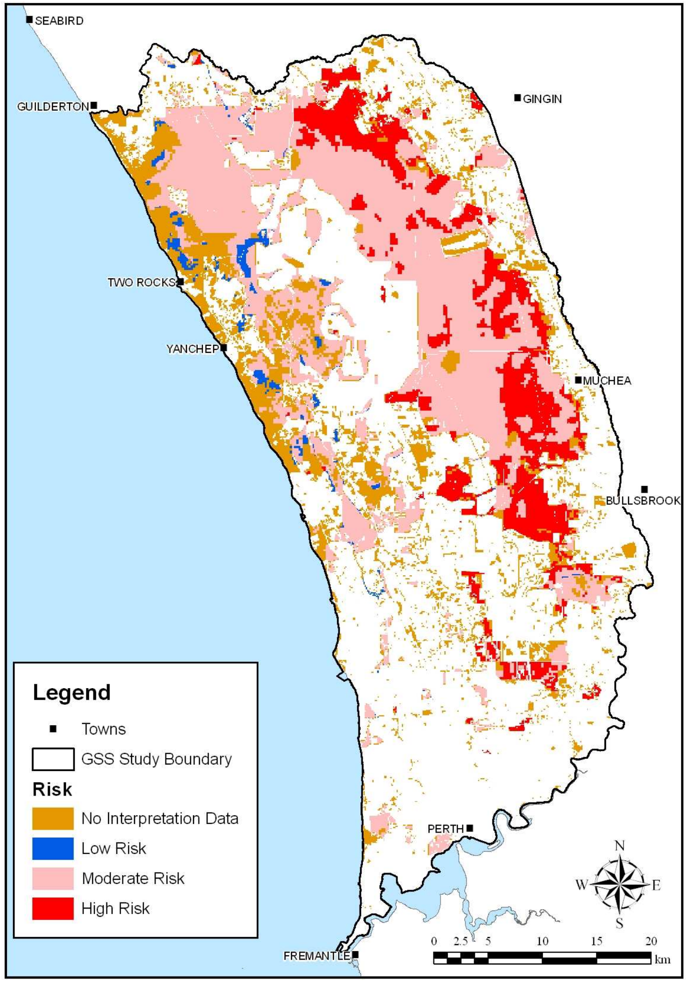
An expert panel assigned the ranks (highest weighted rank model) or classes (perimeter to area ratio) to one of five consequence categories: insignificant, minor, moderate, major or catastrophic impact (see Kinloch and Wilson in prep. for details).

In a geographic information system, risk was then calculated for each 100 m grid cell of remnant vegetation by multiplying the likelihood information by the consequence information. Risk scores were assigned to three risk categories (Figure 11.3a and b) as per AS/NZ (2004) and summary statistics were calculated for the two risk assessments (Table 11.5).

Table 11.5: Extent and proportion of land area in each of the *Phytophthora* dieback risk categories based on the assessment of biodiversity assets using the highest weighted rank model and perimeter to area ratio of remnant vegetation patches.

| **Risk category** | **Highest weighted rank model** | | **Perimeter to area ratio** | |
|---|---|---|---|---|
| | **Total area ha** | **Proportion %** | **Total area ha** | **Proportion %** |
| High risk (score 15–25) | 18 351 | 18.8 | 15 124 | 15.5 |
| Moderate risk (score 5–12) | 50 816 | 52.2 | 50 653 | 52.0 |
| Low risk (score 1–4) | 1 942 | 2.0 | 5 741 | 5.9 |
| No dieback interpretation data (score 0) | 26 299 | 27.0 | 25 890 | 26.6 |
| **Total area of remnant vegetation** | **97 408** | **100.0** | **97 408** | **100.0** |

a)




b)

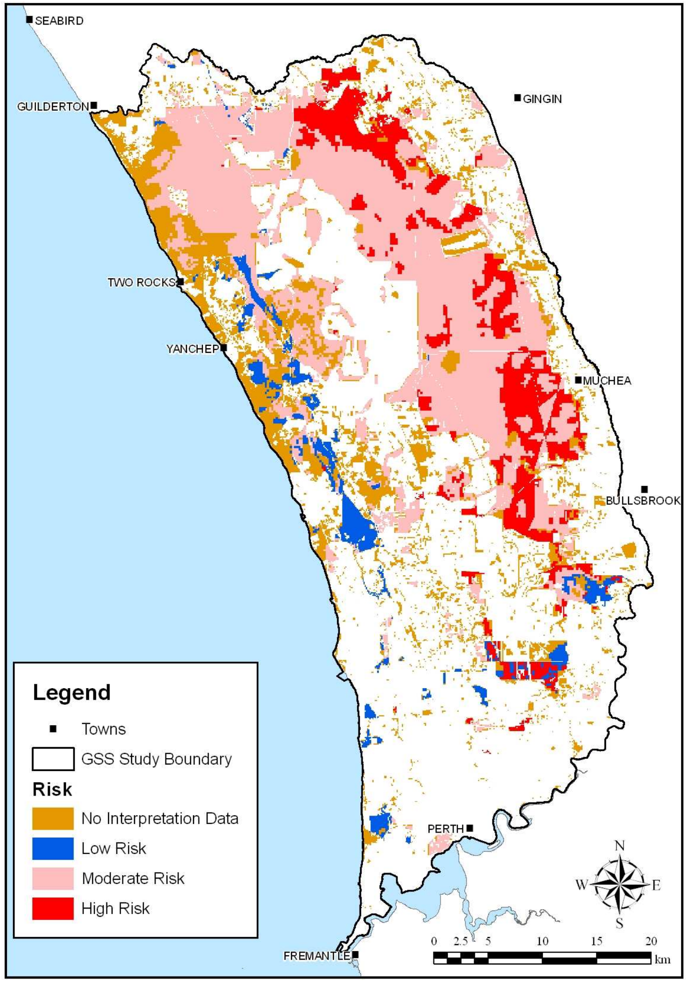



Figure 11.3: Current risk to biodiversity values of *Phytophthora* dieback across the Gnangara Mound using information on biodiversity assets from (a) highest weighted rank model and (b) perimeter to area ratio of remnant vegetation patches.

This risk assessment, using information on biodiversity assets from the highest weighted rank model, has revealed that 19% of remnant vegetation across the Gnangara Mound is currently at high risk of *Phytophthora* impact (Table 11.5 and Figure 11.3a). At the very least, it is moderately likely that *Phytophthora* dieback is currently in these areas (Figure 11.2a) which support significant biodiversity assets (Figure 11.1b). Uncertainty in the assessment of likelihood of *Phytophthora* impact is low or medium in these areas due to the availability of operational interpretation mapping. Specific management actions for these high risk areas are discussed in the 'Priority areas for *Phytophthora* dieback management' section below.

Comparison of the highest weighted rank model and perimeter to area ratio risk assessments show that the location and extent of areas assessed as being high or moderate risk are very similar (Figure 11.3a and b). In the perimeter to area ratio risk assessment the extent of areas categorised as high or moderate risk is slightly lower (Figure 11.3a and b, Table 11.5). This is not surprising as the greater number of biodiversity assets included in the consequence assessment in the highest weighted rank model risk assessment will result in a greater differentiation of areas, in terms of the significance of their biodiversity assets. The inclusion of assets such as the occurrence of threatened flora and ecological communities and the level of representation of vegetation complexes in this model has meant that fewer areas are categorised as low risk. Therefore it appears that a more comprehensive estimate of risk will be achieved if a broad range of biodiversity assets are used to estimate consequence. However, where this information is not available the perimeter to area ratio of remnant vegetation patches could be used as a surrogate.

As mentioned previously this risk assessment has only been able to assess the *current* likelihood that an area is infested with *Phytophthora* dieback. We have not been able to factor in the likelihood that an area will become infested in the future, which is a severe limitation. For example, in Whiteman Park, where operational interpretation mapping is available, areas of moderate risk lie alongside areas of high risk (Figure 11.3a). If the likelihood of an area being infested with *Phytophthora* dieback over a 30-year time period was assessed and the proximity to known infestations, roads and other linear infrastructure as well as the rate of autonomous spread were considered then it is possible that all of Whiteman Park would fall into the high risk category. This highlights the urgent requirement for a landscape predictive model for *Phytophthora* dieback as recommended in Chapter 9.

Overall, this analysis has shown that it is feasible to undertake a relatively quick risk assessment of *Phytophthora* dieback using the framework outlined in Australian Standard 4360 (AS/NZS 2004) and readily available spatial data. This type of risk assessment will provide information to decision makers on the location of priority areas for management or additional mapping.

### *Priority areas for* **Phytophthora** *dieback management*

Information from the highest weighted rank model risk assessment, ranking of biodiversity assets (Project 1 of this chapter) and mapping from Project Dieback (DEC 2008a) has been used to identify priority areas and specific actions for managing *Phytophthora* dieback . Priority areas are those which were identified as having high to moderate risk of *P. cinnamomi* affecting biodiversity values (Figure 11.3a). The limited availability of operational interpretation mapping for the GSS study area means that this risk assessment is probably under-estimating the extent of high or moderate risk areas across the GSS. Therefore investment in *Phytophthora* dieback management across the GSS should not just focus on limiting the spread and impact of the disease in priority areas but also on gaining a greater understanding of the extent of areas infected by interpretation and mapping of *P. cinnamomi*.

Specific management actions for moderate or high risk areas are listed in Table 11.6. Both the potential impact on biodiversity values (high versus moderate risk, Figure 11.3a) and the uncertainty of the assessment (level of confidence of the presence or absence of the pathogen, (Figure 11.2 b) were considered when identifying these actions. *P. cinnamomi* mapping and interpretation was considered a priority management action where the risk is moderate or high and level of uncertainty is medium or high or where no interpretation data exists. This accounts for 71 187 ha of susceptible remnant vegetation. Hygiene and quarantine measures or phosphite application (depending on the dieback status of the vegetation) were the recommended specific management actions for 19 860 ha (see Table 11.6).

Table 11.6: Extent and proportion of susceptible remnant vegetation within the GSS study area and specific recommended initial management actions for *P. cinnamomi*.

| **Highest weighted rank model risk category** | **Uncertainty Assessment** | **Total area of remnant vegetation that is susceptible ha*** | **Proportion of remnant vegetation that is susceptible %** | **P. *cinnamomi* specific management actions** |
|---|---|---|---|---|
| Low risk | High | 47 | 0.05 | Not a priority for action |
| | Medium | 50 | 0.05 | Not a priority for action |
| | Low | 320 | 0.33 | Not a priority for action |
| Moderate risk | High | 30 804 | 31.81 | *P. cinnamomi* interpretation and mapping |
| | Medium | 14 457 | 14.93 | *P. cinnamomi* interpretation and mapping |
| | Low | 1 533 | 1.58 | Hygiene, quarantine measures (if uninfested)<br><br>Phosphite application (if infested) |
| High risk | High | 0 | 0 | |
| | Medium | 9 371 | 9.68 | *P. cinnamomi* interpretation and mapping , and as a 2nd priority action (if infested) phosphite application<br><br>*P. cinnamomi* interpretation and mapping, and as a 2nd priority action (if uninfested) hygiene, quarantine measures |
| | Low | 8 956 | 9.25 | Hygiene, quarantine measures (if uninfested)<br><br>Phosphite application (if infested) |
| No dieback interpretation data | | 16 555 | 17.1 | P. *cinnamomi* mapping and interpretation |

*Total area of susceptible remnant vegetation is slightly lower than that reported in the text above. This is due to the risk analysis being undertaken in a raster format and vegetation type mapping not extending to the GSS boundary in some coastal areas.

Hygiene and quarantine is considered necessary for high and medium risk areas where *P. cinnamomi* is known to be absent, whereas phosphite application and monitoring were considered necessary for those high and medium risk areas where *P. cinnamomi* is known to be present. This intervention may reduce the impact of the disease on the infested areas and may also limit its spread to adjoining un-infested areas. Rehabilitation of infested high risk areas in conjunction with ongoing phosphite application may be a useful management action but requires further investigation to determine if it will provide a long-term benefit to biodiversity on the GSS.

In areas where biodiversity assets are of very high significance and the pathogen is currently known to be absent or thought likely to be absent, quarantine and exclusion to limit access (e.g. strategic track and road closures, restricted access and fencing) should be implemented to prevent the entry of *P. cinnamomi* (Figure 11.4). This represents a significant proportion of remnant vegetation across the GSS and occurs over a range of tenures including conservation reserves, state forest (including areas proposed for protection as a conservation reserve) and unallocated Crown land.

Approximately 17% of the susceptible GSS remnant vegetation has no associated interpretation data (Table 11.6) and therefore the risks to biodiversity assets located on these areas is unknown. Interpretation mapping is the specific management action recommended for these areas and is an urgent priority for those areas which contain biodiversity assets of very high significance (Figure 11.4). These priority areas are scattered throughout the central and north-eastern areas of the GSS, they are usually relatively small in size and the majority occur on private land or small Crown reserves.

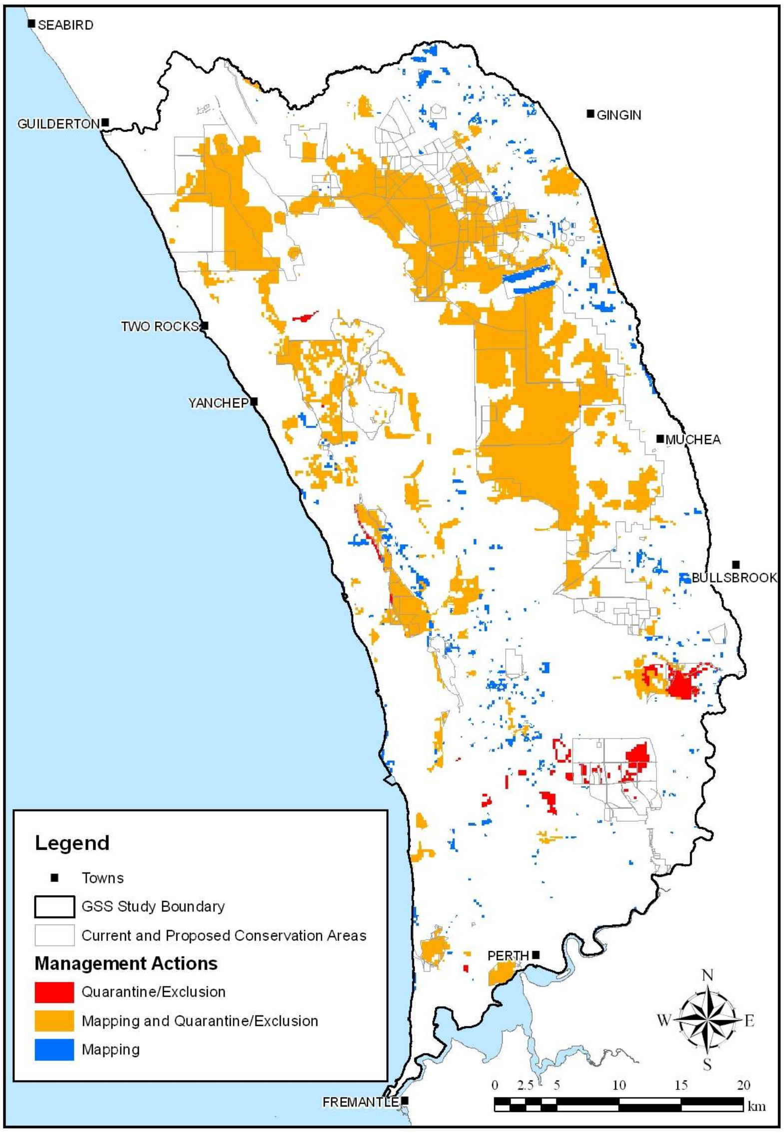



Figure 11.4: Extent of areas with biodiversity assets of very high significance where the management actions include exclusion management (uninfested – high confidence), a combination of mapping and exclusion management (uninfested – low and medium confidence) and mapping (no interpretation data available). Current and proposed conservation areas also include northern Crown reserves (outside the Bush Forever study area).

## *Broad Phytophthora dieback management costs*

The management of *Phytophthora cinnamomi* has been identified as the most cost-effective way of delivering biodiversity benefits for the Swan Coastal Plain by Wilson *et al.* (2007). This assessment was undertaken at the broad ecoregion scale and was based on a formula developed by Wilson *et al.* (2007). This calculates the per area cost of implementing *P. cinnamomi* management, including the cost of:

- phosphite application
- monitoring
- planning
- mapping of *P. cinnamomi*
- policy development
- risk mapping
- communication
- research
- establishing conservation covenants.

For the Swan Coastal Plain this cost was estimated to be $6541 per ha over 20 years. This figure is based on the Wilson *et al.* (2007) estimate of US$514 626 per km$^2$ over 20 years which was converted to Australian dollars using a conversion rate of 1 USD = 1.271 AUD.

The broad cost of implementing *P. cinnamomi* management across the GSS study area was estimated using this formula and information on the extent of susceptible remnant vegetation from Project Dieback (DEC 2008b). Of the remnant vegetation on the GSS study area 85 214 ha or 39.7% is believed to be susceptible to *P. cinnamomi* and the cost of implementing management for all of this area was projected to be $557m over 20 years.

This projected cost, using the Wilson *et al.* (2007) formula, is based on data at the broad ecoregion scale. Costs of managing *Phytophthora* dieback in the GSS can be further refined using finer-scale spatial data from the highest weighted rank model risk assessment of biodiversity assets and more up-to-date information on the cost of *P. cinnamomi* management actions. The highest weighted rank model risk assessment allowed us to identify specific management actions for susceptible remnant vegetation across the GSS (Table 11.6). More specific costings have been estimated for implementing *P. cinnamomi* management actions for the GSS study area and are listed in Table 11.7. It must be noted that these are indicative estimates of costs only. Costs will vary depending on topography, vegetation type and other site characteristics, as well as degree of infestation and size of area to be surveyed or treated. Due to the economies of scale, small areas are likely to have higher cost per ha than large areas.

Table 11.7: Indicative costs of *Phytophthora* dieback management as of July 2009 in the south-west of Western Australia.

| **Management action** | **Estimated yearly cost/ha for each survey or application** | **Repetition rate** |
|---|---|---|
| *P. cinnamomi* broad area interpretation of *Banksia* woodland (using linear survey technique with aerial photographic interpretation) | $7.84 (1) | Every 1 to 3 years |
| Broad area aerial phosphite application, including planning, phosphite, aerial delivery (two applications) and monitoring | $400–$500 (2) | Every 1 to 2 years |
| Targeted, ground-based phosphite application, including planning, phosphite injections, ground spraying and/or slow release tablets and monitoring | $3000–$4000 (2) | 2+ years |

(1) Costs based on interpretation and mapping undertaken by DEC within the GSS study area and outlined in Pez & Swinburn (2009) and Meharry & Swinburn (2009)
(2) Based on phosphite application conducted on the south coast of Western Australia (C. Dunne pers. comm.)

For those areas where interpretation and mapping were identified as a management action (see Table 11.6) the cost of undertaking interpretation mapping is estimated to be $428 k (high and moderate risk areas) and $129 k (risk unknown). To keep information on the extent of *Phytophthora* dieback across the GSS study area current, it is suggested that interpretation mapping be updated at least once every three years. So an ongoing budget for interpretation mapping should also be considered. For those areas where phosphite application is a possible management action (see Table 11.6), the cost of aerial phosphite application and monitoring is estimated to be $9.9m per year with additional finer-scale, targeted ground application of phosphite where necessary. It must be noted that this is certainly an overestimation of the total cost of phosphite application as interpretation mapping may reveal that some of these high and moderate risk areas are not infested with *Phytophthora* dieback. In such instances, hygiene and quarantine measures are recommended.

## Discussion

Identification of biodiversity conservation priorities and risks is essential for the GSS study area, given that the area supports a large number of high value biodiversity assets, the escalating threats these assets are facing and the limited resources available for conservation. The aims of this chapter were to collate and review approaches to conservation planning, including DEC's approach, to review the role of decision making tools in biodiversity conservation, and to address knowledge gaps in the ranking of biodiversity assets and assessment of *Phytophthora* dieback risk across the GSS study area.

The review identified a number of conservation planning frameworks which can assist planners identify conservation objectives and priorities. Systematic conservation planning is a transparent and iterative process that is designed to locate, design and manage protected areas that comprehensively represent the biodiversity of each region. Explicit goals and quantitative targets are set for the protection of biodiversity values and features during the process. It is a proactive conservation approach that complements the reactive measures of species recovery programs and also recognises the importance of conserving underlying ecological processes that support patterns of biodiversity. DEC's approach to conservation planning reflects this systematic approach. In the Swan Region, DEC has determined conservation targets for a number of asset classes (landscape, protected area, wetland, ecosystems at risk and species at risk) and identified conservation priorities using information from a value–threat matrix.

The review also confirmed that decision making tools or frameworks have the potential to provide valuable information on the relative worth of diverse conservation outcomes. Therefore they can play a significant role in determining priority conservation actions and levels of investment in these actions. It must be noted that the application of these decision making tools can be limited by our lack of detailed understanding of ecosystems including how they respond to threats. It is also limited by the availability of spatial data on the location of biodiversity assets and the extent of threatening processes. In some instances surrogate measures of biodiversity can be used to overcome these limitations.

Both qualitative and quantitative decision making tools are used in biodiversity conservation. One qualitative tool is multi-criteria decision analysis which employs criteria relating to biotic, abiotic, biodiversity process, and socio-political (including economic) factors to assess the values associated with different natural areas. A number of different qualitative or quantitative evaluation methods are available to combine the scores for individual criterion into a single index which can be used to rank natural areas. Many previous biodiversity planning initiatives on the Swan Coastal Plain have utilised multi-criteria decision analysis tools to rank natural areas.

Ecological risk assessment is another qualitative tool which can be used to identify and estimate the risk of human activities and threatening processes on flora, fauna and ecological communities. In this type of assessment the source of the risk is identified along with the likelihood of its occurrence and the magnitude of the consequences. Ecological risk assessment has been widely used across Australia to assess the risks associated with the disease *Phytophthora* dieback, which is listed as a key threatening process under the EPBC Act. The last qualitative tool or framework identified was cost-benefit analysis. These frameworks incorporate not only biotic, abiotic and information on threats into conservation priority setting but also information on the economic cost and economic value of the benefits.

Limitations associated with the ranking of natural areas using multi-criteria methods have driven the development of quantitative tools using optimisation algorithms. These tools determine the minimum number of sites necessary to represent all species or biodiversity features by integrating mathematical programming and spatial design criteria and by dealing with complementarity.

A number of knowledge gaps were addressed. Multi-criteria decision analysis tools were used to compare two evaluation methods to rank the biodiversity assets across the GSS study area. Both models determined that 43% of the GSS study area had biodiversity assets which are rated as being of high significance and these are largely located in the north and central areas of the study area where there has been limited development. However, small, usually isolated, pockets of land ranked as highly significant occur in other areas. Limitations were identified with both evaluation methods but despite these, both models provided new perspectives and valuable insights into those areas of the GSS study area having significant biodiversity assets.

A *Phytophthora* dieback risk assessment of biodiversity assets across the GSS study area was also undertaken. This determined that 19% of remnant vegetation across the GSS study area is currently at high risk of *Phytophthora* impact and a further 52% is at moderate risk. However, it must be noted that the uncertainty in the assessment of the likelihood of *Phytophthora* impact for some of these areas is high due to the lack of operational interpretation mapping. It was also determined that when information on a broad range of biodiversity assets is not available, data on the perimeter to area ratio of remnant patches could be used as a surrogate in these types of assessment. Priority areas for *Phytophthora* dieback management were identified for high and moderate risk areas and the projected cost of implementing *P. cinnamomi* management, for susceptible remnant vegetation, across the GSS study area was investigated.

## Recommendations

In order to improve conservation planning, priorities and cost-effective management it is recommended that:

- criteria, surrogate measures of biodiversity and spatial data for biodiversity assets be further developed so that ranking models can be refined
- the tools available in Marxan be used to develop additional decision support information for input into conservation planning across the GSS study area
- the *Phytophthora* dieback risk assessment be further developed once a landscape predictive model for the future spread of the pathogen in the GSS has been developed (see Chapter 9 recommendations)
- the costs associated with managing *Phytophthora* dieback, across the GSS study area, be further refined using finer spatial scale data relating to priority areas for management
- mapping and interpretation of *Phytophthora* dieback be undertaken for 71 187 ha
- phosphite application be undertaken for 19 860 ha of high and moderate risk susceptible remnant vegetation
- in addition to the measures recommended above, exclusion management should also be considered where an area is currently uninfested in areas where biodiversity assets of very high significance occur
- mapping should be considered where no data is currently available.

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