The Extent, Condition and Management of Remnant Vegetation in Water Resource Recovery Catchments in South Western Australia

Water Resource Technical Series

Water and Rivers Commission Report WRT 15

1999

Water and Rivers Commission

Water and Rivers Commission

Hyatt Centre
3 Plain Street
East Perth
Western Australia 6004

Telephone (08) 9278 0300
Facsimile (08) 9278 0301
Website www.wrc.wa.gov.au

Cover Photograph:
Lake Carabundup area in the Kent River catchment, May 1999

THE EXTENT, CONDITION AND MANAGEMENT OF REMNANT VEGETATION IN WATER RESOURCE RECOVERY CATCHMENTS IN SOUTH WESTERN AUSTRALIA

Report to the Natural Heritage Trust by

M. Strawbridge

Water and Rivers Commission
Resource Investigations Division
Catchment and Salinity Investigations Section

WATER AND RIVERS COMMISSION
WATER RESOURCE TECHNICAL SERIES
REPORT NO WRT 15
1999

Acknowledgments

This report was funded by the Natural Heritage Trust and the Water and Rivers Commission.

The Water and Rivers Commission wishes to thank the following landholders for kind permission to undertake vegetation surveys on their properties:

Mount Barker Research Station,
Agriculture Western Australia
B.C. and B.A. Bunker
A.F. Beech
M. and E. Hall
N.A. and C.A. Handasyde
D. and V. Drage
T.D. and V.A. Allison
P. Pavlovich

For more information contact:

Melanie Strawbridge
Plant Biologist
Catchment and Salinity Investigations Section
Hyatt Centre, 3 Plain Street
EAST PERTH WA 6004

Telephone (08) 9278 0468
Facsimile (08) 9278 0586

Reference Details

The recommended reference for this publication is:

Strawbridge, M. 1999, *The Extent, Condition and Management of remnant vegetation in Water Resource Recovery Catchments in South Western Australia,* Water and Rivers Commission, Water Resource Technical Series No WRT 15.

ISBN 0 7309 7397 2
ISSN 1327 8436

Printed on recycled stock

November 1999

Contents

Figures

Tables

Summary

There is increasing evidence to show that livestock grazing leads to degradation of remnant vegetation through destruction of the understorey, lack of native species recruitment and invasion of exotic species. Further loss of vegetation will exacerbate rising groundwater and salinisation. Under the Western Australian Salinity Action Plan, the Denmark River, Upper Kent River, Warren River and Wellington Reservoir catchments, have been designated water resource recovery catchments. The Water and Rivers Commission's aim in these catchments is to maintain or restore water quality to potable levels.

This project mapped areas of remnant vegetation in the above catchments. Native forest, modified remnants, scattered trees and plantations were mapped at 1:50 000 and 1:100 000 scale. Nine percent, 11%, 6% and 6% of the Denmark River, Upper Kent River, Warren River and Wellington Reservoir catchments were identified as modified remnant vegetation and potentially at risk from livestock grazing.

Vegetation surveys were carried out in remnants where grazing had been excluded for periods between five and 20 years. Grazing exclusion only ameliorated sites which had a history of limited grazing. Where grazing had been more intense, exclusion prevented further deterioration and the cover of some native perennial species did increase. However, the depleted soil seed bank and changed soil chemical and physical parameters did not promote recruitment of native species.

The relative effects of autumn and spring burning on weed control and regeneration in grazed remnants were examined. Neither spring nor autumn burning led to a significant increase in species recruitment. Fire resulted in increased cover of perennial native species but only in areas that had limited, previous livestock grazing. No significant effect of seasonal burning was seen on control of exotic species.

Best management practice should exclude all livestock grazing in remnant vegetation. However, under circumstances of drought or sheep weather warnings, remnants with an intact understorey may support limited grazing. Further studies are required to determine acceptable stocking rates. Use of selective and rapidly degradable herbicides are preferred in non-riparian zones over burning or grazing for weed control. Use of the latter two methods only promote further invasion of exotic species. However, intact remnants require infrequent fires for regeneration.

Monitoring any changes in vegetation is an important element of management. Seasonal or annual photographs from a fixed position (photopoints) are recommended. More intensive monitoring should be undertaken in selected remnants to improve the knowledge and understanding of remnant rehabilitation processes. This will also provide early recognition of failure of any treatments and reduce the risk of further investment into projects with low chances of success.

In order to ensure continued protection and management of remnant vegetation on private land in water resource recovery catchments, it is important that clear advice be provided on management issues. Potential sources of financial assistance for management activities like the fencing scheme, currently funded through the Salinity Action Plan, need to be identified. The catchment benefits of conserving remnant vegetation also need to be clearly demonstrated.

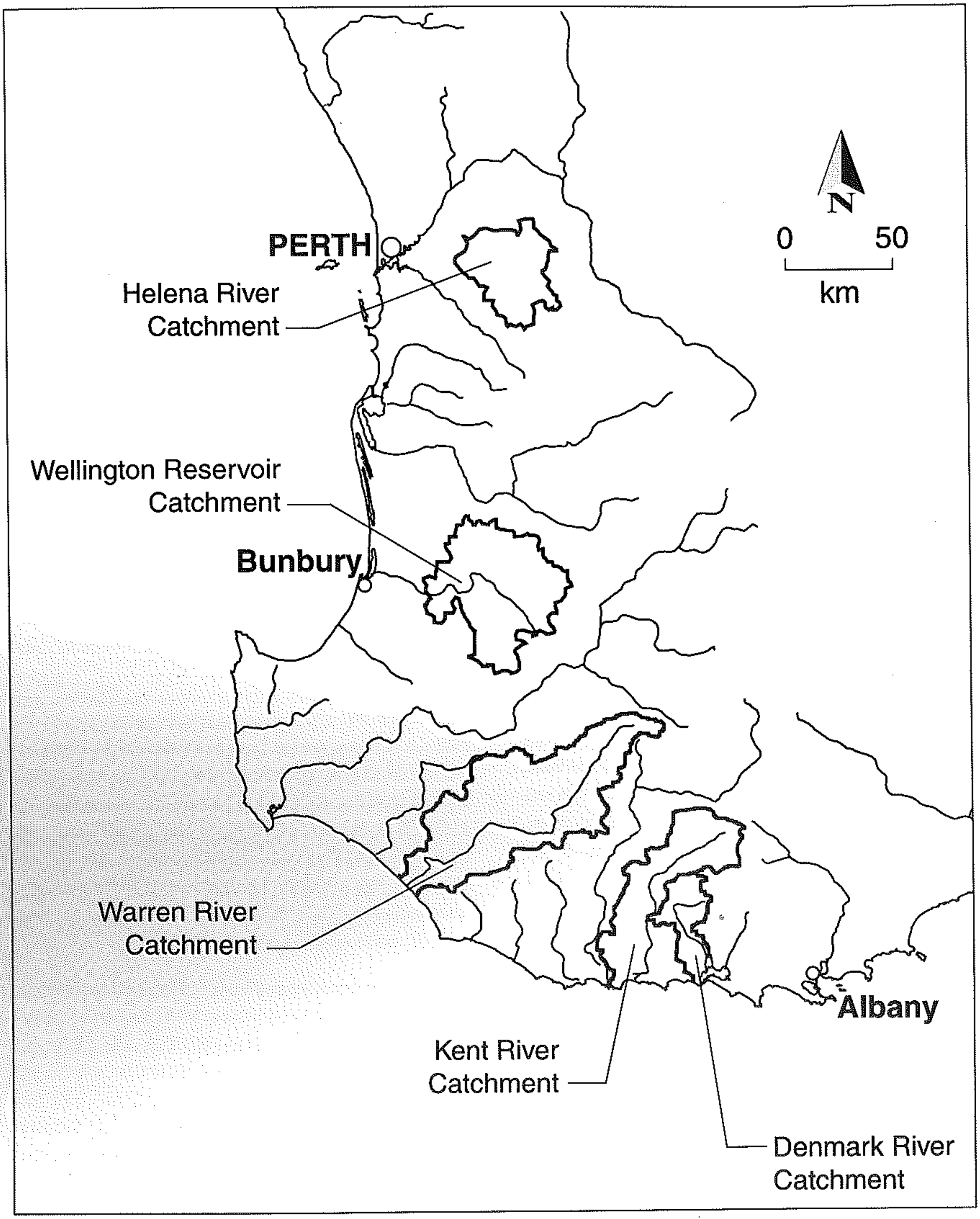

Figure 1.1 Location of the water resource recovery catchments in south Western Australia.

1. Introduction

To ensure adequate water resources for present water supply demand and to meet increasing future demands, five key water catchments have been identified in the Western Australian Salinity Action Plan as water resource recovery catchments; Mundaring Weir (Helena River), Wellington Reservoir (Collie River), Warren River, Kent River and the Denmark River (Figure 1.1). These catchments have the potential yield of 440 gigalitres per annum or twice Perth's current water supply (Western Australian Salinity Action Plan 1996). The management aim of these catchments is to maintain or restore the water quality to potable levels. Potable water is defined as having salinity of less than 500 mg L^{-1} Total Dissolved Solids (TDS), marginal water 500–1000 mg L^{-1} TDS and saline water > 5000 mg L^{-1} TDS (Steering Committee for Research on Land Use and Water Supply—Western Australia 1989).

Many areas of remnant vegetation in water resource recovery catchments are not protected from disturbance, principally livestock grazing, which leads to degradation through destruction of the understorey, lack of recruitment, invasion of exotic species, etc. This will eventually lead to the loss of these areas which will cause a further decline in water quality in these catchments.

The Natural Heritage Trust funded a three year project from 1995 to 1998. The principal objectives of this project were to map areas of remnant vegetation in water resource recovery catchments and to investigate rehabilitation and management options for the major remnant types in those catchments that have been disturbed by livestock grazing. In the Helena River catchment, isolated remnants on farming land play a much less significant role in the maintenance of water quality as it is still mostly forested. Therefore, it was not included in this project.

This report is divided into six sections. Chapter two discusses clearing history and impacts of agriculture on remnant vegetation in water resource recovery catchments. Chapter three briefly describes the mapping of the extent and condition of remnant vegetation in the catchments. Chapter four reports the results from investigations on the impact of grazing exclusion on the rehabilitation of remnant vegetation. Chapter five examines the use of fire for rehabilitation. Finally, Chapter six discusses management objectives, issues and recommendations for remnant vegetation in water resource recovery catchments.

2. Agricultural impacts on remnant vegetation

2.1 Clearing history of water resource recovery catchments in south Western Australia

Twenty million hectares of land in the Southwest land division of Western Australia have been alienated for agriculture (George *et al.* 1996). Of this, 87% has been cleared of all native vegetation. About 4.5 million hectares remain as National Parks, nature reserves and other public land, while 2.7 million hectares remain as small and scattered remnants of the original native vegetation (George *et al.* 1996).

The vegetation of the Denmark River catchment consists of mixed *Eucalyptus marginata* (jarrah) and *E. calophylla* (marri) forest, which is defined as closed tree formation of 10–30 metres high (Beard 1981) (Figure 2.1a). It also contains low forest (closed formation of low trees under 10 metres, dominated by jarrah), with swamps. The commercial exploitation of timber in the Denmark Catchment commenced in the 1870s and agriculture was mainly confined to pastoralism, when small mixed farms were established in the areas that had been clear-felled between Mount Lindesay and Lindesay Gorge (Ruprecht *et al.* 1985). Major clearing occurred in the Denmark Catchment between 1946 and 1957 and again between 1965 and 1973. Eighty percent of the clearing occurred in the upper Denmark catchment which receives 700 to 800 mm yr^{-1} rainfall (Moulds and Bari 1995a).

The Kent River catchment contains mostly woodland; open formation of medium trees, principally *E. loxophleba* (York gum), *E. salmonophloia* (salmon gum), *E. salubris* (Gimlet) and *E. wandoo* (wandoo) (Figure 2.1b) (Beard 1981). It also contains low woodland (open, low formation under 10 metres); forest; low forest and swamps. The vegetation systems in the Kent River catchment have been described in greater detail in Kelly (1995). Initial land use

Figure 2.1 ***Eucalyptus marginata*** **(jarrah)–*E. calophylla*** **(marri) forest (a),** ***E. wandoo*** **(wandoo) woodland (b) and** ***Melaleuca*** **low woodland (c). These vegetation types originally formed a mosaic of communities across south Western Australia, but are now restricted to areas of State forest, reserves and remnants on private land.**

was based on pastoral leases that were taken up on the Hay River and at Kendenup (Kelly 1995). Clearing of the land for agricultural began in the 1870s and was centred on Albany and Kendenup, then progressed to Cranbrook, Mt Barker and Denmark by 1900. From 1900, Government policy encouraged sowing of crops and pastures as opposed to sheep grazing on native pasture (Kelly 1995) and settlement of the country between Mt. Barker and Kojonup was completed by 1930 (Beard 1981).

The Wellington Reservoir catchment contains mostly remnants of mixed jarrah-marri forest. Wandoo also occurs. Clearing for agriculture began in the early 1900s and slowly expanded until the 1930s when the Depression caused agricultural expansion to virtually cease (Moulds and Bari 1995b). After the Second World War, large land releases occurred under the rigid conditions that most of it be rapidly cleared. Initial clearing was restricted mostly to valleys but with the introduction of bulldozers, in the late 1950s and 1960s land clearing accelerated and extended upslope from the valleys.

The Warren River catchment contains three vegetation categories. Jarrah–marri forest; tall forest (closed, medium density canopy greater than 30 metres) dominated by *E. diversicolor* (karri) either in pure stands or mixed with jarrah and marri; and low woodland of *Melaleuca rhaphiophylla* and other *Melaleuca spp.* (Figure 2.1c). Most of the clearing occurred in the 1950s mainly in the upper reaches of the catchment in the 600–800 mm yr^{-1} rainfall area.

The private remnants and reserves which remain are of differing size and isolated to different degrees. The original vegetation was a mosaic of communities which ranged from forests and tall open woodlands to low heathlands, which were distributed according to landform and soil type (Beard, 1990). Remnants are not representative of the original vegetation of the region because of selective land clearing. Woodlands in particular were used by settlers as an indicator of the best soils for agriculture (Beard and Sprenger 1984). Jarrah, *Banksia attenuata* (fire wood banksia) and *B. menziesii* (Menzies' banksia) were associated with inferior soils, while wandoo and to a lesser extent marri, were used to indicate more fertile soils (Bell and Heddle 1989). Most existing remnants occur on deep sand, rock outcrops and gravel ridges (Merriam and Saunders 1993; George *et al.* 1995). One hundred and four species of plants have become extinct in Western Australia, with a further 132 plant species listed as rare and endangered (Saunders and Hobbs 1989). Species of animals are now also isolated within these remnants. Of the 46 species of native mammals in the area before European settlement, 13 have disappeared; nine of them extinct on the Australian mainland (Saunders and Hobbs 1989).

2.2 Increasing stream salinity and enactment of clearing control legislation

Before clearing, evapotranspiration accounted for over 95% of rainfall (George *et al.* 1996). Very little water moved past the deep roots of the native vegetation and entered the deep groundwater. Recharge rates on catchments in south Western Australia were in the order of 0.1mm yr^{-1} (George 1992). Shallow water tables of less than 2 m only occurred at the bottom of large catchments under wheatbelt salt lakes or near springs adjacent to rocky outcrops in higher rainfall areas.

The average stream salinity of the Kent River, Warren River and Wellington Reservoir catchments increased significantly from less than 500 mgL^{-1} TDS prior to 1955 to greater than 1000 mg L^{-1} TDS after 1990 (Table 2.1). There was also an observable relationship between area of catchment cleared and resulting stream salinity. In the Warren River catchment, the Dombakup River sub-catchment was 15% cleared, and had a stream salinity of 160 mg L^{-1} TDS (Davies and Bari 1995). In contrast, the Tone River sub-catchment was 64% cleared and had a stream salinity of 900 mg L^{-1} TDS (Davies and Bari 1995). Moulds and Bari (1995a) recorded a similar relationship in the Denmark River catchment. TDS for the Yate Flat sub-catchment (60% cleared) was 1823 mg L^{-1} and 709 mg L^{-1} for the Mt. Lindesay sub-catchment (17% cleared). In addition, groundwater rose to be on average 6 m, 4.5 m and 1.5 m below the soil surface respectively for the three Denmark River sub-catchments.

Concerns were first raised in the 1950s about the continued clearing of forested land causing increases in the salinity of streams. As a result further land alienation was stopped in Water Resource Catchments in 1960. However, clearing of previously alienated land continued and salinity levels also continued to rise. In November 1976 the Country Areas Water Supply Act 1947 was amended to provide control of clearing of indigenous vegetation within the Wellington Reservoir Catchment Area (Collie/Harris River) by the then Public Works Department (PWD). In

Table 2.1

Average stream salinity of water resource recovery catchments prior to 1955 and after 1990

Catchment	Stream Salinity Total Dissolved Solids (mg L^{-1})	
	Prior to 1955	After 1990
Denmark River	–	709 – 1823 (Moulds and Bari 1995a)
Kent River	280 (Hawkins 1997)	1240 (Hawkins 1997)
Warren River	120 – 350 (Collins and Barrett 1980)	160 – 4900 (Davies and Bari 1995)
Wellington Reservoir	280 (Hookey and Loh 1985)	980 (Moulds and Bari 1995b)

December 1978 the Act was further amended to extend the controls to the Mundaring Weir (Helena River) and Denmark River Catchment Areas and to the Warren River and Kent River Water Reserves.

Guidelines were drafted for government officers to administer the clearing control legislation by granting or refusing licences to clear. Issues related to clearing emerged which were not covered by the original guidelines and these were handled with special and interim guidelines. These were reviewed in 1994 to include all the relevant issues and to redefine and document a policy and guidelines that enabled applications for clearing licences to be assessed and dealt with equitably. The review focused on the maintenance of water quality protection principles. This not only featured strong protection of sustainable remnant forest areas but also fostered landowner participation in sustainable land management practices that can provide economic, social and environmental benefits to the local and wider community.

The guidelines for the granting of licences outline how discretion will be exercised. In recognition of the variation in potential for saline discharge following clearing, each catchment has been divided into zones A to D (decreasing hazard; all catchments do not have all zones) in order that decisions on licensing might be consistent and realistic. In Zone A licences will not be granted for broad acre clearing. In Zones B and C, clearing licences are generally granted for areas less than 10 ha and 25 ha respectively. In Zone D licences will normally be granted subject to 10% of the property remaining uncleared. A Claim for Compensation may arise when the Water and Rivers Commission refuses a part or all of an application or the Minister refuses a subsequent appeal. In the assessment of a claim, compensation is not payable for remnant vegetation amounting to 10% of the property. Compensation is paid at market price for bush and at least 10% is added for the special value of the bush to the property if it had been cleared.

2.3 Impact of livestock grazing and other disturbances on remnant vegetation

In negotiating settlement claims for compensation, it was accepted that there would be 'limited' grazing of domestic stock in uncleared areas based on the premise that degradation would be minimal. Since the introduction of clearing controls no monitoring of the effects of this grazing has taken place. However, it has become obvious that degradation of the vegetation is occurring in the Kent River catchment (True *et al.* 1992) and the Wellington Reservoir catchment (Pettit and Froend 1994). Post 1978, many remnants had been actively grazed year round. Continuous grazing over a number of years has caused, in some cases, complete loss of native understorey species, weed invasion, grazing damage, soil compaction and other problems. There is also a lack of recruitment of tree and shrub species and a decline in the health of existing trees. This, in the long term, is a form of insidious clearing as the old trees die and are not replaced. The potential loss of these areas will, in effect, negate the purpose of the clearing bans of preventing salinity increases in the catchment.

Assessment of 130 sites in the Kent River catchment in 1992, showed that 62% of the remnants had been significantly disturbed, with a further 30% having had moderate disturbance; no sites were undisturbed (True *et al.*, 1992) (Table 2.2). Grazing of domestic livestock was recorded for 87% of the sites with other factors affecting remnant health including waterlogging (17% of remnants) and salinity (18%). Native herbivores caused significant damage to the vegetation in only 7% of the remnants. Only 10% of remnants had been protected from livestock grazing by fencing. Of the sites, 38% were less than five hectares in size, 51% were less than ten hectares and 91% were less than fifty hectares with the average size being nineteen hectares. However, 60% of sites were directly connected to other areas of native vegetation increasing the potential for corridors to be established between remnants, increasing their functional size.

Figure 2.2 **Sixty-two percent of surveyed remnants in the Kent River catchment had been subject to significant disturbance. Livestock grazing reduced remnant vegetation to two structural layers, trees and grass (a), and inhibited the recruitment of native species. Salinity (b) effectively reduced remnant vegetation to the tree layer only.**

Table 2.2

Disturbance factors and levels of disturbance seen in a survey of 130 remnants in the Kent River catchment (True *et al.* 1992).

Parameter	Sites Affected (%)
Level of Disturbance	
• undisturbed	0
• slightly	8.0
• moderate	30.0
• significant	62.0
Disturbance Factor	
• fire	25.4
• salinity	17.7
• livestock grazing	87.0
• native herbivore grazing	7.0
• soil compaction	44.7
• waterlogging	17.0
• weed invasion	
0–20% of total species present	42.3
20–80%	32.3
> 80%	25.4
• rubbish	10.0
• logging	50.8
• mining	1.5
• dieback	5.4
Other	
• fully fenced	10.7
• connected to other areas of bush (remnants, windbreaks, reserves, vegetated roadsides)	60.0

In 53 of the 130 sites surveyed in the Kent River catchment, at least 20% of the total plant species present were exotic. These introduced species are generally annual grasses or herbs invading from surrounding agricultural land. Altered conditions such as reduction in vegetation cover, elevated soil nutrient levels and increased soil disturbance primarily from livestock grazing, allow these species to readily invade remnants. This alters the vegetation structure of the remnant from one dominated by native perennial species to one dominated by annual exotic species, resulting in reduced hydrological effectiveness of the remnant and increased vulnerability to erosion.

Many remnants in the Kent River catchment have been reduced to only two structural layers consisting of the tree layer and a dense sward of exotic annual grasses (e.g. *Lolium ridgidum*—annual rye grass) which can prevent recruitment of native species (Figure 2.2a). Some sites have been reduced to effectively only one structural layer (trees) where the site has been affected by salinity or erosion or on lateritic hilltops (Figure 2.2b). These sites with only one or two structural groups will provide the greatest challenge for management and rehabilitation.

In 1991 the Water and Rivers Commission established sites in the Wellington Reservoir catchment and is examining the long-term effects of grazing by livestock in remnants of native vegetation. Analysis after three years showed that livestock grazing within remnant vegetation resulted in a change from a community dominated by native perennial species, to one dominated by exotic annual species (Pettit and Froend 1994). High grazing intensity, length of grazing history, climatic variability and effects on the soil are the major factors affecting the observed responses of the vegetation to grazing. Natural regeneration in degraded remnants is possible if livestock are excluded, however, initially, this may consist of a different community than the original. Where a long history of high grazing intensity exists, rehabilitation of some sites is also required.

3. Mapping of remnant vegetation

3.1 Introduction

A major objective of this project was to map areas of remnant vegetation in order to quantify how much was potentially at risk from livestock grazing and provide a tool for the formation of management plans. In addition, areas of reforestation were also mapped as plantation forestry on private land is increasing significantly in the water resource recovery catchments. Areas of plantation are critical components of catchment modelling to predict groundwater levels, stream flows and stream salt loads in response to land use changes in water resource recovery catchments.

3.2 Preparation of vegetation maps

Base maps of the catchments were prepared from Landsat TM data and cadastre overlays, using Microstation software (Intergraph 1991). In the Kent River catchment, only the northern part of the catchment (upper Kent River) was mapped. Aerial photographs were then used to confirm finer details of remnants and plantations. Finally, the maps were checked on site for condition of remnants, recent areas of plantations and degraded areas. Remnant vegetation was placed into three categories according to the level of understorey disturbance. Native forest (intact understorey); Modified (understorey disturbed principally by livestock grazing); and Scattered (no native understorey and low numbers of trees).

The Upper Kent (1995; reviewed 1998) and Denmark River catchment (1998) vegetation maps are 1:50 000. The Wellington Reservoir (1997) and Warren River catchment (1997) maps consist of a 1:100 000 map of the entire catchment and a series of 1:50 000 maps for greater clarity. Due to the loss of detail which would result from sizing the maps at A4 or A3, they are separate to this report. If required, copies of the maps may be obtained from the Water and Rivers Commission.

3.3 Summary

A significant proportion of each catchment remains forested (Table 3.1). However, salinity has increased in the rivers as discussed in Chapter two. Nine percent of the Denmark River catchment, eleven percent of the Upper Kent River catchment, six percent of the Warren River catchment and six percent of the Wellington reservoir catchment were identified as modified remnant vegetation and potentially at risk from livestock grazing. Any further vegetation loss will only contribute to salinisation of the catchments. It will also result in loss of biodiversity for those vegetation associations which are not well represented due to selective clearing.

In the Wellington Reservoir catchment, ground water levels were reduced ten years after reforestation of 15% and 28% of the Maringee Farm and Maxon Farm sub-catchments (Moulds and Bari 1995b). Stream flow and salt load were not significantly reduced, however levels did not increase. Modelling of the hydrological effects of vegetation in the Upper Kent River catchment estimated that at least 37% of the cleared areas would need to be reforested to return the salt concentration to potable levels (Dixon *et al.* 1998). In addition, studies in the Blackwood catchment have shown that stream flow and salt load began to reduce five years after reforestation of 72% of the cleared area in the Padbury Road sub-catchment (Moulds and Bari 1995b).

Up until 1997, 16% of cleared land had been reforested in the Denmark River catchment, 15% in the Upper Kent River catchment, 10% in the Warren River catchment and 22% in the Wellington Reservoir catchment. Comparing these levels of reforestation to the studies summarised above indicates the significant amount of reforestation still required for reduction of stream flow and salinity in the water resource recovery catchments. However, current reforestation levels may be enough to halt further increase and reduce groundwater levels in the Wellington Reservoir catchment. It is anticipated that the maps will be upgraded every three to five years to show how many modified remnants have been protected from livestock grazing and how much of the catchment is being reforested with plantation species.

Table 3.1 The extent and condition of remnant vegetation in water resource recovery catchments.

Catchment	Denmark (1998)	Upper Kent (1998)	Warren (1997)	Wellington Reservoir (1997)
Total Area (km^2)	758	1108	4406	2827
Native Forest (%)	64	21	56	67
Modified Remnant (%)	9	11	6	6
Scattered Vegetation (%)	2	4	3	2
Cleared (%)	20	52	27	18
Reforestation (%)	4	9	3	5
Cleared Land Reforested (%)	16	15	10	22

4. Grazing exclusion and rehabilitation of remnant vegetation

4.1 Introduction

Grazing disturbance in native plant communities usually results in changes to the structure and composition of vegetation communities. Exotic annual species are favoured by the changed conditions resulting from livestock grazing while most native species, particularly perennials, decrease in number (Williams 1969; Hacker 1984; Belsky 1992; Milchunas and Lauenroth 1993; Scougall *et al.* 1993; Pettit *et al.* 1995; Abensperg-Traun *et al.* 1996). In many Australian habitats, grazing has lead to the loss of particular functional types of plants and this may have important implications for the ecological functions within the habitat (Hobbs 1991a). For example the loss of perennial shrub species may increase the risk of soil erosion, disrupt nutrient cycling, decrease animal habitat and decrease food sources for pollinators, all of which would lead to major changes in the function of habitats.

In the 1992 Kent Catchment survey, livestock grazing affected 87% of the 130 remnants surveyed. This was the most significant disturbance. At that time, only 10% of remnants had been fenced. Livestock grazing had resulted in changed vegetation structure, one in which the community was now dominated by annual exotic species.

This grazing-induced shift from a plant community dominated by native perennial species to one dominated by exotic annual species was also seen in jarrah–marri and jarrah–wandoo remnants in the Wellington Reservoir catchment (Pettit and Froend 1994). After three years, exclusion of grazing resulted in an increase in the number of native perennial species. This regeneration was derived from resprouting rather than germination from the soil seed bank.

Results from the Wellington Reservoir catchment surveys after eight years are reported here. In addition, a separate study was conducted that examined the effects of long periods of grazing exclusion in jarrah-marri and wandoo remnants for periods up to twenty years (Kent-Denmark River catchment surveys). Unlike the Wellington Reservoir surveys, information on the species present prior to exclusion was not available. However, by measuring several sites, effects of grazing exclusion over greater time periods than that so far covered by the Wellington study, can be measured. Specifically, the objective of these surveys was to examine the ability of jarrah-dominant and wandoo-dominant remnants to recover their floristic composition after livestock grazing is excluded and no other amelioration is attempted.

4.2 Materials and methods

4.2.1 Wellington Reservoir catchment surveys

The materials and methods for the Wellington Reservoir catchment grazing exclusion surveys are briefly summarised here. A more detailed description is found in Pettit and Froend (1994) and Pettit *et al.* (1995).

At twenty seven sites, 10 m × 10 m quadrats were established in remnants considered to be representative of those that occur in the eastern Wellington Reservoir catchment. The sites were characterised by the dominant overstorey species and included 16 jarrah-marri sites, 8 jarrah-wandoo and 3 wandoo-jarrah sites. Grazing intensity at each site was given a subjective ranking based on grazing history, presence of exotic species and condition of vegetation at the site. These grazing intensity categories were; never grazed; light grazing; moderate grazing; heavy grazing; and severe grazing (Table 4.1). In sites that were

Table 4.1

Sites details for the grazing intensity categories from the Wellington Reservoir catchment grazing exclusion trials.

Grazing intensity category	Site number
Never grazed	4, 5, 9, 11, 13, 16, 17, 21, 26
Least grazed	7, 18, 19, 20, 22, 24
Moderately grazed	1, 2, 6
Heavily grazed	3, 8, 12, 14
Severely grazed	10, 15, 23, 25, 27

grazed, quadrats were fenced, with an adjacent unfenced quadrat pegged out. Single quadrats were pegged out in sites that were not grazed. The initial vegetation survey of sites was completed in November 1991. Species richness (the number of species present) and relative cover of each species were determined for each plot. The estimate of the cover of each species was made from the Domin-Krajina scale (Mueller-Dombois and Ellenberg 1974).

4.2.2 Kent-Denmark River catchment vegetation survey

Sites for survey were chosen from aerial photographs then examined in the field. Sixteen jarrah-marri sites were selected in the Upper Denmark Catchment. Sites were grouped into four categories of grazing history; currently grazed, grazed four to five years ago, grazed six to ten years ago and never grazed. In addition there were two size categories, large (> 10 ha) and small remnants (< 10 ha) (Table 4.2).

Ten sites were selected for assessment of wandoo communities, in the Upper Kent catchment. Sites were grouped into four categories of grazing history; currently grazed, grazed five to ten years ago, grazed ten to twenty years ago and never grazed. There were not enough sites to assess differences between large and small remnants as was done with the jarrah-marri communities in the Denmark catchment.

In October 1995, two 50m transects were established at each site in the wandoo remnants. Transects were placed at least 20 metres within remnants to avoid edge effects. This process was repeated in the jarrah-marri remnants in September 1996. A 1 m × 1 m quadrat was placed at 5m intervals along each transect. Species richness, the number of individuals of each species (abundance) and the relative cover of each species were recorded. Species were identified using local floras (Blackall and Grieve, 1980; Blackall and Grieve,1981a; Blackall and Grieve, 1981b; Blackall and Grieve, 1982; Marchant *et al.*, 1987a; Marchant *et al.*, 1987b; Blackall and Grieve, 1988; Hussey *et al.* 1997). Relative cover was estimated using the Domin-Krajina scale (Mueller-Dombois and Ellenberg, 1974). The presence and cover of the seedlings of overstorey species were recorded and analysed as part of the understorey. Species importance values (IV) were calculated for all species occurring in sites in each grazing class. This was calculated as:

IV = relative abundance + relative cover + relative frequency (Kent and Coker 1996).

Table 4.2

Site details for floristic surveys of understorey species in jarrah-marri communities in the Upper Denmark River catchment and wandoo communities in the Upper Kent River catchment.

Site No.	Community	Grazing History	Site Size small = < 10 ha large = > 10 ha
1	jarrah-marri	current	small
2	jarrah-marri	current	small
3	jarrah-marri	current	small
4	jarrah-marri	current	large
5	jarrah-marri	current	large
6	jarrah-marri	4 - 5 years ago	small
7	jarrah-marri	4 - 5 years ago	small
8	jarrah-marri	4 - 5 years ago	small
9	jarrah-marri	4 - 5 years ago	large
10	jarrah-marri	6 - 10 years ago	small
11	jarrah-marri	6 - 10 years ago	small
12	jarrah-marri	6 - 10 years ago	large
13	jarrah-marri	never grazed	small
14	jarrah-marri	never grazed	large
15	jarrah-marri	never grazed	large
16	jarrah-marri	never grazed	large
17	wandoo	current	
18	wandoo	current	
19	wandoo	current	
20	wandoo	5 - 10 years ago	
21	wandoo	5 - 10 years ago	
22	wandoo	5 - 10 years ago	
23	wandoo	10 -20 years ago	
24	wandoo	10 -20 years ago	
25	wandoo	10 -20 years ago	
26	wandoo	Never grazed	

4.2.3 Kent-Denmark River catchment survey site characterisation

At each site the soil profile was described to 1m (McDonald *et al* 1990). Three soil samples were taken at each site (0–10 cm) and analysed for nitrate, phosphorus and potassium (Rayment and Higginson 1992). Litter was removed and a surface soil sample (top 1cm) was taken every 10m along each transect (2 × 5 samples per transect) and analysed for water repellency (WR) using the molarity of ethanol drop method (MED) (King 1981). Nine surface cores were also taken per site and the bulk density (BD) of the top 10 cm of soil determined (Loveday 1974).

4.2.4 Statistical and non-parametric analyses

Statistical and non-parametric analyses of the data were completed using Statistica 5.1 software (Statsoft 1984–1996). All species richness and soil data were examined for normal distribution using the Shapiro-Wilks W test (Zar 1984). Analysis of Variance (ANOVA) was then used to examine the relationship between species richness and; grazing exclusion and grazing severity (Wellington survey); years since grazing and site size (Kent/Denmark survey). Bartlett's test was conducted to determine if the data met the assumption of homogeneity of variance (Zar 1984). Where they did not, species richness data were transformed to their log values and the soil data to log values. Transformed data were then analysed for variance. Multivariate analysis of variance (MANOVA) was used to determine which soil chemical and physical properties differed significantly between sites with different grazing histories and site size. Post hoc comparison of means was undertaken with the Neuman-Keuls test and power analysis of each ANOVA was also undertaken (Zar 1984).

Relative cover data were ordinated using Multidimensional Scaling. The ordination matrix was prepared by using the coefficient of squared Euclidean distance measures (Kent and Coker 1996) with unweighted pair-group method using arithmetic averages (UPGMA) linkage rules (Sokal and Michener 1958; Kent and Coker 1996). The resulting output was then examined for groupings of sites based on the relative cover of each species present.

The number of exotic species and number of native species per site were correlated with soil chemical and physical data.

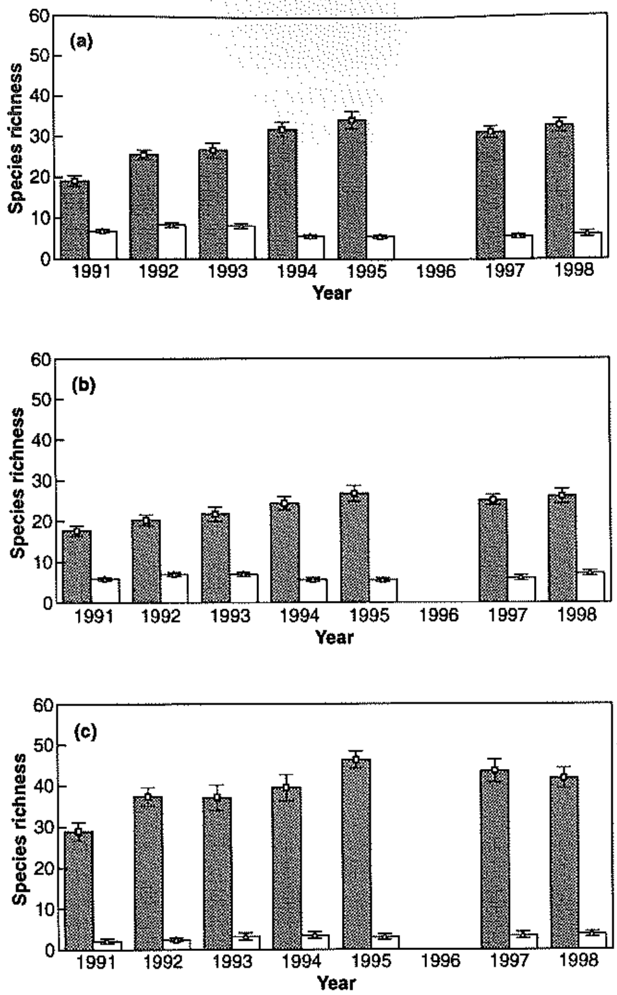

Figure 4.1

Species richness in grazing excluded (a), grazed (b) and never grazed control sites (c) in jarrah-dominant remnant vegetation, in the Wellington catchment, from 1991 to 1998. Mean numbers of native species are shown as closed bars and exotic species as open bars. P ≤ 0.01 for ANOVA's comparing native species richness in grazed and grazing excluded sites. P values for ANOVA's comparing native and exotic species richness in previously grazed sites and never grazed sites are P ≤ 0.00 and P ≤ 0.05. Vertical bars indicate standard errors of the means.

a: $F_{\text{native spp. 91-98}} = 6.8$ (d. f. = 6, P = 0.000)

b: $F_{\text{native spp. 91-98}} = 3.1$ (d. f. = 6, P = 0.006)

c: $F_{\text{native spp. 91-98}} = 4.4$ (d. f. = 6, P = 0.001)

4.3 Results

4.3.1 Wellington Reservoir catchment surveys

From 1992 to 1998, the number of native species in quadrats where grazing had been excluded, had increased significantly relative to grazed quadrats (Figure 4.1a and b). In 1991, when quadrats were fenced, there were 19.1 and 17.6 mean numbers of native species in grazing-excluded and grazed quadrats respectively. By 1998, this had increased to 32.8 and 26.0 mean number of native species. Over the same time period, there was no significant difference in number of exotic species between treatments. Grazing-excluded and grazed quadrats had significantly less native species and significantly greater number of exotic species, than control quadrats in remnant vegetation that had never been grazed (Figure 4.1c).

When sites were separated into groups based on their prior condition from livestock grazing, sites which had previously been subjected to only light or moderate grazing showed no changes in numbers of native species seven years after grazing was excluded (Table 4.3). Sites that had been subjected to heavy grazing prior to exclusion had an increased number of native species after three years (1994). There was no significant increase in number of native species in sites that had been previously subjected to severe grazing.

From the means and standard errors, sites that had been lightly grazed appeared to have greater numbers of native species than other grazing classes for each year. Although significance levels from 1993 onwards do not support this statement, power analysis of the ANOVA indicated that there was only a 30–60% confidence of detecting a difference between means for each grazing class in those years. Further calculation showed that a sample size of eleven sites in each graze class would be required to be 80% confident of detecting a difference between grazing groups at the 0.05% level of significance.

When the relative cover of each species was ordinated, sites where grazing was excluded and sites which were grazed did not group separately. However, sites that had never been grazed, least grazed and moderately grazed grouped together, as did sites that had been heavily and severely grazed (Figure 4.2). These two groups formed a continuum along axis 1 and did not change between 1991 and 1998.

Table 4.3 Mean number of native species recorded in the Wellington Reservoir catchment, in fenced quadrats in four classes of prior grazing intensity, between 1991 when quadrats were first fenced and 1998. Standard errors are shown in brackets.

Year	**Grazing Class** (Pettit and Froend, 1994)				
	Light grazing	**Moderate grazing**	**Heavy grazing**	**Severe grazing**	**$F_{grazing\ class}$ (d.f., P)**
1991	30.7 (3.8)	15.7 (3.4)	15.7 (1.4)	10.0 (1.6)	8.7 (3, 0.00)
1992	30.7 (1.2)	20.3 (3.5)	18.7 (2.1)	19.7 (2.7)	5.1 (3, 0.01)
1993	38.0 (5.1)	24.0 (4.5)	21.3 (2.0)	10.7 (2.0)	2.4 (3, 0.10)
1994	41.7 (2.6)	27.3 (2.7)	30.0 (1.6)	15.7 (1.9)	3.0 (3, 0.06)
1995	47.0 (4.6)	29.7 (2.8)	25.3 (2.4)	17.0 (1.7)	4.0 (3, 0.02)
1997	32.3 (1.8)	32.0 (2.6)	27.7 (1.5)	18.0 (3.9)	0.6 (3, 0.60)
1998	39.3 (3.8)	30.0 (2.6)	29.3 (2.2)	16.0 (2.9)	1.3 (3, 0.30)
F_{year} (d.f., P)	2.0 (6, 0.13)	2.2 (6, 0.10)	5.5 (6, 0.00)	1.4 (6, 0.27)	

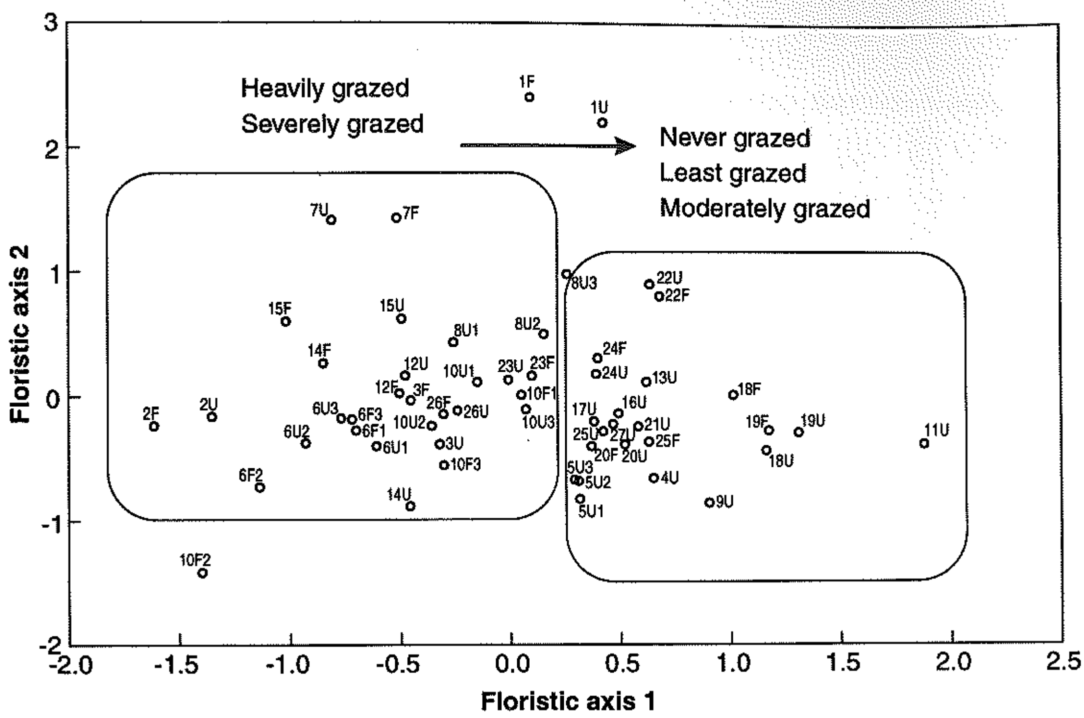

Figure 4.2 1998 ordination analysis using multidimensional scaling of relative cover of each species present in sites in jarrah-dominated remnant vegetation in the Wellington catchment. Heavily and severely grazed sites were separated along floristic axis 1 from sites that were moderately grazed, least grazed and never grazed.

4.3.2 Analysis of years since grazing and site size data— Kent–Denmark surveys

Species richness in jarrah-marri sites was not significantly different for sites currently grazed, sites grazed four to five years ago and sites grazed ten years ago (Figure 4.3). However, sites that had been grazed had a significantly lower number of native species than sites that had never been grazed (5.3 to 15.6). Conversely, there was a decrease in the number of exotic species per site from 6.2 to 2.6, as time since grazing increased.

Ordination of sites based on the relative cover of each species present, showed a separation of plots between those currently grazed and grazed five years ago, and those grazed ten years ago and never grazed (Figure 4.4).

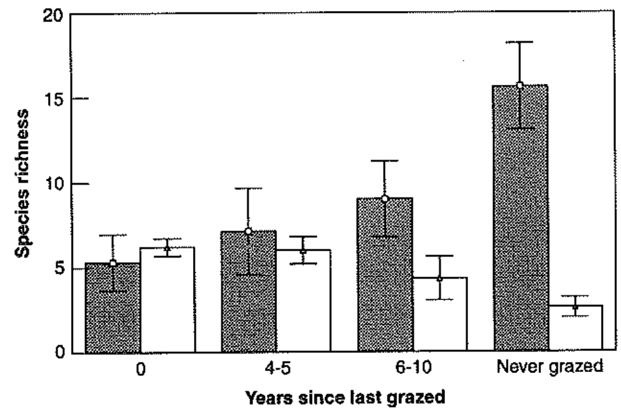

Figure 4.3

Species richness in jarrah-marri remnants in response to years since last grazed by livestock. Mean number of native species are shown as solid bars and mean number of exotic species as open bars.

$F_{\text{native spp.}} = 4.3$ (d.f. = 3, P = 0.013);

$F_{\text{exotic spp.}} = 5.1$ (d.f. = 3, P = 0.006).

Vertical bars indicate standard errors of the means.

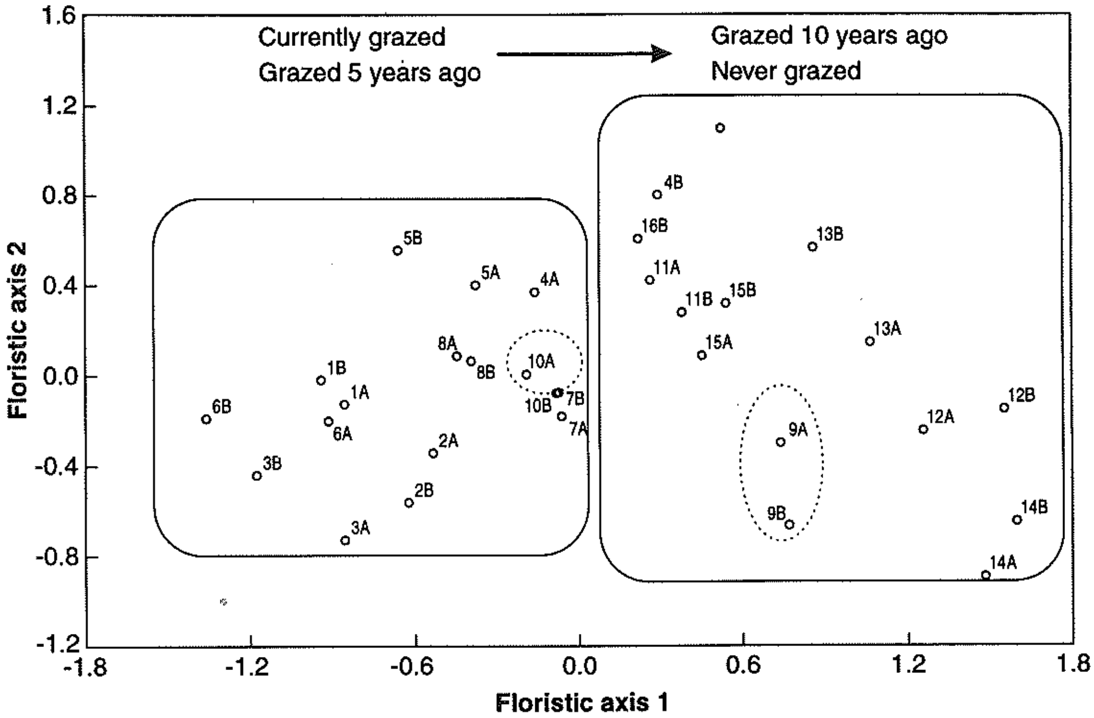

Figure 4.4 Ordination analysis using multidimensional scaling of relative cover of each species present in jarrah-marri sites in the Denmark catchment. Groups are separated along floristic axis 1 by increasing time since grazing. Sites enclosed by broken lines indicate areas of overlap.

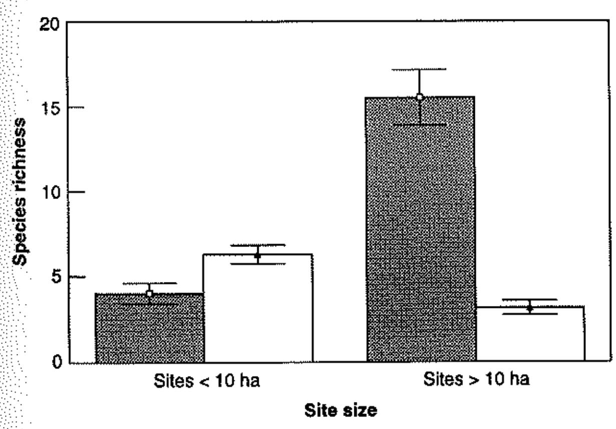

Figure 4.5

Species richness in jarrah-marri remnants in response to site size. Mean number of native species are shown as solid bars and mean number of exotic species as open bars.

$F_{native\ spp.} = 52.8$ (d.f. = 1, P = 0.000);

$F_{exotic\ spp.} = 19.1$ (d.f. = 1, P = 0.000).

Vertical bars indicate standard errors of the means.

Site size had a highly significant effect on species richness. Sites less than 10 ha had a mean of 4.0 native species per site and sites greater than 10 ha had a mean of 15.5 native species per site (Figure 4.5). Small sites had two times the number of exotic species as large sites (6.3 compared to 3.1). There was no significant interaction between age class of sites and site size on species richness.

Wandoo sites that had never been grazed were limited in number and a large standard error resulted from the statistical analysis (Figure 4.6). Because of this, sites with previous grazing history were also analysed separately from those never grazed. The number of native species was significantly lower in sites currently grazed (4.0) than in those where grazing had been excluded for five to ten years (5.4) and ten to twenty years (5.0). The number of exotic species present in sites with previous grazing history did not differ significantly.

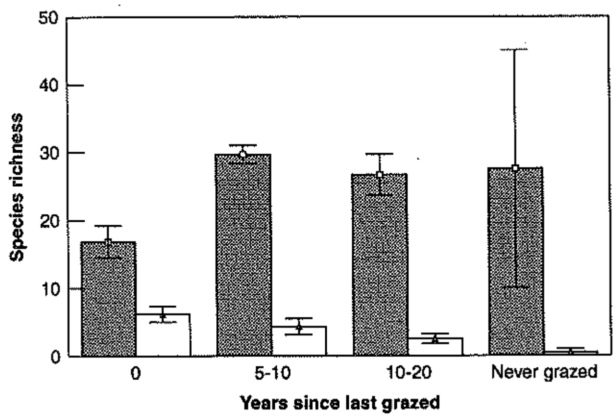

Figure 4.6

Species richness in wandoo remnants in response to years since last grazed by livestock. Mean number of native species are shown as solid bars and mean number of exotic species as open bars. Statistics are reported from analysis where sites that have never been grazed are excluded.

$F_{\text{native spp.}} = 8.9$ (d.f. = 2, P = 0.003);

$F_{\text{exotic spp.}} = 3.0$ (d.f. = 2, P = 0.078).

Vertical bars indicate standard errors of the means.

Ordination analysis of species cover in wandoo sites showed a continuum from sites currently grazed and grazed ten years ago, to those grazed twenty years ago and never grazed (Figure 4.7).

When species were ranked according to their importance value in each grazing class, four out of the top five species in currently grazed jarrah–marri remnants were exotic (Table 4.4). Five out of five and three out of five species were exotic in sites last grazed four to five years ago and six to ten years ago respectively. The annual grass *Vulpia myuros* (rats tail fescue) was the most important ranked species in all three of these grazing classes. The only native species to rank in the top five in currently grazed sites was *Pteridium esculentum* (bracken). Jarrah-marri sites that had never been grazed still contained the annual herbs *Hypochaeris glabra* (smooth cats ear) and *Oxalis purpurea* (purple wood sorrel) among the top five important species.

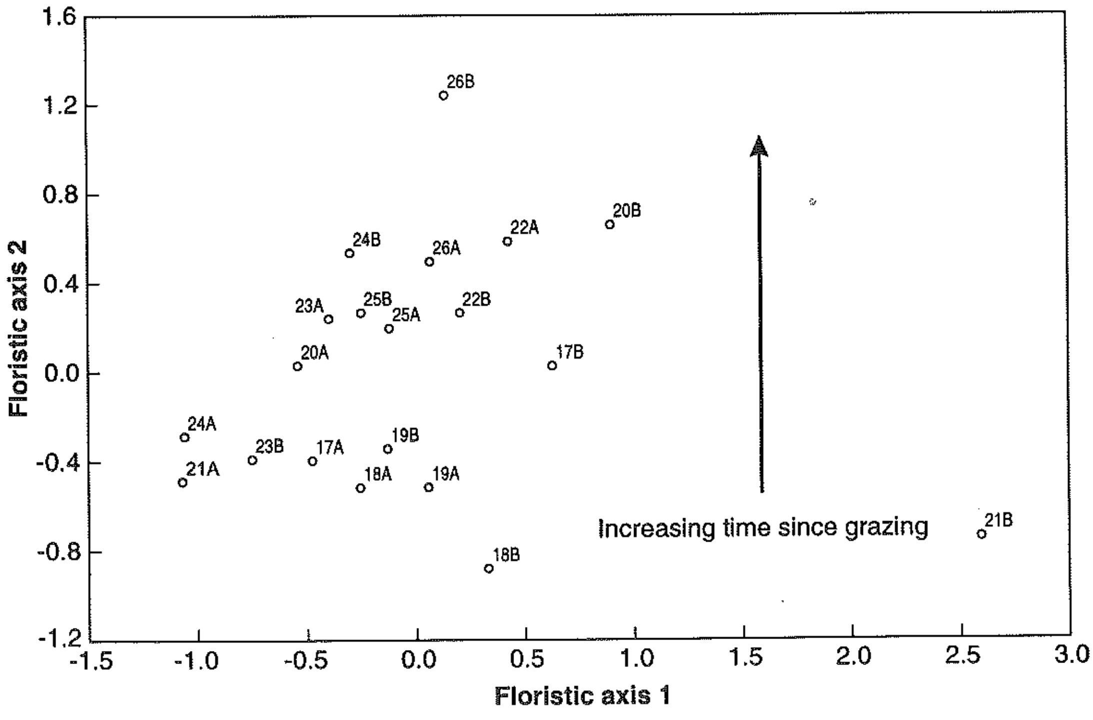

Figure 4.7 Ordination analysis using multidimensional scaling of relative cover of each species present in wandoo sites in the Kent catchment. There is a continuum of sites with increasing time since grazing along floristic axis 2.

Table 4.4 Abundance, cover and frequency data of the five highest ranked species according to their importance value for each grazing class in jarrah-marri sites. * Indicates exotic species.

Species	Abundance (# individuals)	Cover (%)	Frequency (% sites)	Importance value
Currently grazed				
*Vulpia myuros**	2483	39.1	82	79.3
*Trifolium subterraneum**	568	18.1	49	29.8
*Malva parviflora**	488	19.7	48	29.2
Pteridium esculentum	174	17.1	34	19.6
*Hordeum vulgarum**	655	0.6	15	18.2
Grazed 4-5 years ago				
*Vulpia myuros**	1442	32.0	59	66.5
*Oxalis purpurea**	242	12.5	41	22.8
*Hypochaeris glabra**	165	9.8	31	21.4
*Aira caerophyllea**	416	9.8	20	20.2
*Lolium rigidum**	471	8.4	16	19.8
Grazed 6-10 years				
*Vulpia myuros**	337	6.0	27	35.8
*Hypochaeris glabra**	366	16.8	40	33.5
Tremandra diffusa	150	14.8	32	17.8
*Lolium rigidum**	172	5.0	8	14.4
Lepidosperma sp.	89	19.2	35	14.1
Never grazed				
Lepidosperma sp.	441	20.2	39	24.8
*Hypochaeris glabra**	290	13.9	32	18.8
Bossiaea disticha	126	30.1	48	14.6
Tetratheca affinis	215	10.5	25	13.1
*Oxalis purpurea**	191	3.6	9	10.8

The top five important species in currently grazed wandoo sites included *Lolium ridgidum* (annual ryegrass), *Hypochaeris glabra* and *Trifolium subterraneum* (sub-clover) (Table 4.5). No exotic species ranked highly in wandoo sites grazed five to ten or ten to twenty years ago, or never grazed. Currently grazed wandoo sites included the native species *Hakea lissocarpha* and *Chaemaescilla corymbosa*. *C. corymbosa* occurred in all wandoo sites with decreasing importance value as time since grazing increased. It ranked ninth in sites which had never been grazed. Wandoo sites that had never been grazed contained more native seeder species in the top five than sites that had been previously grazed. Native species in these sites were predominantly resprouters.

Table 4.5 Abundance, cover and frequency data of the five highest ranked species according to their importance value for each grazing class in wandoo sites. * Indicates exotic species.

Species	Abundance (# individuals)	Cover (%)	Frequency (% sites)	Importance value
Currently grazed				
*Lolium ridgidum**	85	1.3	67	17.5
Hakea lissocarpha	7	1.3	17	15.0
Chaemaescilla corymbosa	88	0.8	100	13.7
*Hypochaeris glabra**	125	0.7	100	12.3
*Trifolium subterraneum**	25	0.7	83	11.9
Grazed 5-10 years ago				
Opercularia sp.	115	4.0	83	17.8
Bossiaea eriocarpa	40	3.6	50	15.3
Chaemacscilla corymbosa	88	2.2	100	11.2
Bossiaea linophylla	12	2.0	17	8.0
Hakea lissocarpha	10	1.7	50	7.8
Grazed 10-20 years ago				
Lepidosperma sp.	332	7.6	100	27.1
Pericalymma ellipticum	100	3.4	33	11.9
Loxocarya fasiculata	21	2.4	83	10.2
Leptocarpus sp.	20	2.1	17	7.1
Chaemacscilla corymbosa	61	0.9	67	5.1
Never grazed				
Bossiaea eriocarpa	175	30.7	100	112.8
Stylidium spathulata	49	3.8	50	34.2
Davesia hakeoides	14	2.6	50	23.0
Lepidosperma sp.	74	4.7	100	11.3
Acacia obovata	7	1.1	50	9.7

4.3.3 Analysis of soil variables—jarrah-marri sites

Soil profiles from jarrah-marri sites showed most sites to be duplex (texture contrast) soils with loamy sands to between 35 cm and 55 cm, over a clay soil. Two sites (6 and 12) had loamy sands to 1m. Five sites (9, 10, 14, 15 and 16) had B horizon's of gravel hard pans.

Grazed sites had a significantly greater nitrate content of 6.73 mg kg^{-1} compared to 2.8 mg kg^{-1} for sites which had not been grazed for five years (Table 4.6). Nitrate content did not decrease significantly with time past this point.

Table 4.6

Effect of years since grazing on significant soil chemical and physical properties. Standard errors are shown in brackets.

Years since grazing	NO_3^- (mg kg^{-1})	WR (M)
0	6.73 (1.75)	2.99 (0.07)
5	2.83 (0.64)	3.22 (0.09)
10	4.11 (0.77)	2.90 (0.08)
Never grazed	2.00 (0.17)	2.41 (0.09)
F; df; p	5.77; 3; 0.00	15.57; 3; 0.00

Table 4.7

Effect of site size on significant soil chemical and physical properties. Standard errors are shown in brackets.

Site Size	P (mg kg^{-1})	NO_3^- (mg kg^{-1})	K (mg kg^{-1})	BD (g cm^{-3})	WR (M)
Small	6.15 (0.42)	5.25 (1.03)	224.26(43.98)	0.87 (0.03)	3.16 (0.05)
Large	4.57 (1.02)	2.57 (0.46)	98.52 (22.03)	1.03 (0.04)	2.55 (0.06)
F; df; p	11.20; 1; 0.00	11.67; 1; 0.00	23.58; 1; 0.00	11.10; 1; 0.00	53.62; 1; 0.00

Water repellency was not significantly different between sites currently grazed (2.99 M), those not grazed for five years (2.99 M) and those not grazed for ten years (3.22 M). However, water repellency was significantly decreased at sites which had never been grazed (2.41 M). There was no significant effect of grazing history on phosphorous and potassium content in soils or on bulk density.

Small sites had significantly greater contents of nitrate, phosphorus and potassium than large sites (Table 4.7). For nitrate and potassium the difference was two fold (5.25 mg kg^{-1} compared to 2.57 mg kg^{-1}, 224.26 mg kg^{-1} compared to 98.52 mg kg^{-1}).

The mean water repellency of small sites was 3.16 M, which was significantly higher than that for large sites (2.55 M.) Bulk density was significantly greater for large sites (1.03 g cm^{-3} compared to 0.87 g cm^{-3}).

Water repellency was positively correlated with the number of exotic species per site (0.74) and negatively correlated with number of native species per site (-0.72) (Table 4.8).

Table 4.8

Significant correlations between soil physical and chemical properties of jarrah-marri sites and total number of species, number of exotic species and number of native species per site.

Soil property	No. exotic species per site (r, n, P)	No. native species per site (r, n, P)
Water Repellency (M)	0.74, 16, 0.000	-0.72, 16, 0.002
Phosphorus (mg kg^{-1})	0.65, 16, 0.007	-0.63, 16, 0.008
Nitrate (mg kg^{-1})	0.42, 16, 0.108	-0.56, 16, 0.025
Potassium (mg kg^{-1})	0.53, 16, 0.034	-0.52, 16, 0.037

A similar relationship existed for soil phosphorus and potassium content. Soil nitrate content was negatively correlated with the number of native species (-0.56) but was not significantly correlated with the number of exotic species per site (0.43).

4.3.4 Analysis of soil variables—wandoo sites

Soil profiles from wandoo sites showed most sites to be duplex soils with coarse, loamy or clayey sands over a clay soil. For all sites, the A Horizon varied between 30 to 75 cm. One site (5) had clay nodules in the A2 horizon and four sites had gravel throughout the A horizon (3, 4, 7 and 9).

There was no significant effect of grazing history on nitrate, phosphorus and potassium content in soils. The bulk density of surface soils from sites which were currently grazed was 1.21 g cm^{-3}, which was significantly higher than for sites which had increasing time since grazing (0.95, 1.06 and 0.95 g cm^{-3} respectively) (Table 4.9). These sites were not significantly different from each other.

Sites which were currently grazed, had not been grazed for five to ten years and had never been grazed, all had similar water repellence on surface soils (2.47, 3.11 and 3.12 M) (Table 4.9). Sites that had not been grazed for ten to twenty years showed significantly less water repellence (1.13 M).

Nitrate content was negatively correlated with the number of exotic species per site (r=-0.58, n=20 and P=0.008). There was no significant correlation with the number of native species per site.

Table 4.9

Effect of years since grazing on significant soil chemical and physical properties of wandoo sites. Standard errors are shown in brackets.

Years since grazing	BD (g cm^{-3})	WR (M)
0	1.21 (0.03)	2.47 (0.23)
5 - 10	0.95 (0.04)	3.11 (0.15)
10 - 20	1.06 (0.05)	1.13 (0.19)
Never Grazed	0.95 (0.01)	3.12 (0.30)
F; df; p	10.16; 3; 0.00	13.64; 3; 0.00

4.4 Discussion

Analysis after eight years, of jarrah-marri and jarrah-wandoo sites in the Wellington Reservoir Catchment, showed that grazing exclusion ameliorated sites that had limited grazing history. Under these conditions, the number of native species and cover of native perennial species increased. The number of exotic species remained the same indicating that once established, only active management can remove these species. Remnants that had been subjected to intense livestock grazing before exclusion did not deteriorate any further, but showed no improvement.

Similarly, in wandoo remnants in the Kent River catchment, both number and cover of native species increased after ten years of grazing exclusion. In contrast, grazing exclusion in jarrah–marri remnants in the Denmark River catchment, did not show increased numbers of native species after ten years. However, the cover of native perennial species did increase. This indicates that over this time period, livestock exclusion may appear to restore jarrah–marri remnants as perennial species are not continually grazed back. However, once removed from the community, there was very limited recruitment of species. Similarly to those remnants surveyed in the Wellington Reservoir Catchment, numbers of exotic species in both the wandoo and jarrah–marri remnants in the Kent and Denmark Catchments, did not decrease after livestock were excluded.

The lack of recruitment of both understorey and overstorey species seen in jarrah-marri remnants in this study is most likely explained by a depleted soil seed bank. The amount of native perennial seed of understorey species in the soil seed bank in undisturbed jarrah forest has been shown to be comparatively low (Vlahos and Bell 1986; Pettit and Froend 1994), and herbivory of native annual and perennial species would mean that there is little recruitment of these species into the soil seed bank (Pettit and Froend 1994). Gradual depletion of the seed bank store would then take place as seed germination, seed predation and decay reduces the number of viable seed in the soil (Pettit and Froend 1994). These factors have also been shown to decrease the soil seed bank of the jarrah overstorey (Majer 1980; Stoneman and Dell 1994; Stoneman *et al.* 1994; Yates *et al.* 1995; Ward *et al.* 1997). Pettit and Froend (1994) found that native annual species have the greatest proportion of seed in the soil of jarrah-dominant remnants, however these species form only a small component of the understorey.

In south Western Australia, jarrah forest generally has a high percentage of perennial species which regenerate from resprouts (Bell and Heddle 1989). However, resprouters have been found to be susceptible to sustained grazing pressure (Bowen and Pate 1993). The ability for species to regenerate from grazing by resprouting will be a function of intensity and length of grazing history.

When species were ranked according to their importance value, exotic species were dominant in sites with most recent grazing history. In jarrah-marri remnants, the only native species to rank in the top five in currently grazed sites was *Pteridium esculentum* (bracken) which can itself become a weed in unmanaged paddocks and remnants in higher rainfall zones (Hussey *et al.* 1997).

In currently grazed wandoo sites, *Hakea lissocarpha* and *Chaemaescilla corymbosa* occur in the top five important species. These species are resprouters which have the ability to grow back after grazing to ground level. Conversely, Wandoo sites that had never been grazed contained a higher proportion of seeder species as lack of herbivory meant recruitment of seeds into the soil seed bank. *C. corymbosa* occurred in all wandoo sites with decreasing importance value as time since grazing increased. This species has become more dominant in relation to other native species in the currently and recently grazed sites which may be more affected by grazing.

Exotic species ranked as important in jarrah-marri sites that had never been grazed but did not rank in similar sites in wandoo remnants. This may indicate that jarrah-marri remnants are more susceptible to weed invasion, however the small sample size of ungrazed wandoo remnants precluded any firm conclusions. Studies in wheatbelt wandoo communities found that number of weed species present increased from undisturbed to grazed sites (Abensperg-Traun *et al.* 1996). Weed cover also increased from less than 5% in undisturbed patches compared to 25% in grazed areas. These studies also found that the diversity and number of native species decreased with increasing percentage cover of weeds.

Sites greater than ten hectares contained more native species and less exotic species than smaller sites. They also showed lower soil nutrient levels and surface soil water repellency. These findings agree with a study by Prober and Thiele (1995) who found that in *Eucalyptus albens* (white box) remants, species richness generally increased with remnant size and smaller remnants were more vulnerable to weed invasion. However, native species richness and composition were more severely affected by livestock grazing than by remnant size.

Edge effects have a greater impact on smaller remnants as larger remnants have a greater internal area which is much less affected by the environmental changes occurring at the edge (Williamson 1975; Janzen 1983; Harris 1988; Yahner 1988). This is important, given the small size of many of the remnants in south Western Australia with high edge to area ratios. However, long thin remnants may lie along environmental gradients and thus contain more vegetation types and habitats than a square reserve of a similar area (Saunders *et al.* 1991).

Jarrah-marri remnants had increased nitrate levels in soils that were currently grazed. Other studies have shown that livestock grazing induced changes in soil chemical and physical characteristics of sites, which generally promoted the establishment of exotic species (Heddle and Specht 1975; Hobbs 1989; Cale and Hobbs 1991; Hester and Hobbs 1992; Scougall *et al.* 1993; Pettit and Froend 1994). Soils under native vegetation are generally low in nutrients (Bettenay and Hingston 1961; Bettenay, 1984), and plants have evolved under this environment. The nutrient cycles pre-European settlement were fairly closed, with redistribution of nutrients across the landscape occurring from fire and normal erosional processes (Hobbs 1993a). In many native heath species, increased nutrient levels result in increased shoot growth (Specht 1963). In the dry season, the less developed root system is unable to maintain water to the shoots and severe water stress develops leading to plant death. Increase in nutrient levels in woodlands has lead to the increase in the abundance of defoliating insects, which has contributed to canopy decline and tree dieback (Landsberg *et al.* 1990).

Bulk density of soil was greater in currently grazed wandoo sites. However after five years this returned to a level similar to sites which had never been grazed. Soil compaction adversely affects the processes that ordinarily maintain structure (e.g. invertebrate burrowing, root exploration by plants, carbon cycling in upper soil horizons) (Scougall and Majer 1991; Main 1993). It may cause reduced water infiltration in the dry season and provide a physical barrier to seedling emergence and root growth (Hunt and Gilkes 1992). Decreased water infiltration through out the dry season may also increase the risk of erosion from high intensity thunderstorms (Pettit et *al.* 1998). In temperate eucalypt woodlands, loss of soil structure and water infiltration are considered to have the greatest impact on species composition (Yates and Hobbs 1997). Annual exotic species utilise the top few centimetres of soil in the wet season so are not affected by decreased water infiltration in the dry season, as are native perennial species (Pettit *et al.* 1998).

In the jarrah-marri remnants studies, sites subject to previous livestock grazing had increased levels of soil surface water repellency compared to ungrazed sites. This was not ameliorated by increasing time since grazing ceased. This relationship between livestock grazing and water repellency has been previously reported in jarrah-marri remnants by Pettit *et al.* (1998).

Water repellence is caused by hydrophobic organic compounds produced by mycorrhizal fungi growing in the soil (Bond 1969; Hubble *et al.* 1983). The mechanism of the indirect effect of livestock grazing on water repellence is not evident in the published literature. Although soil compaction decreased water infiltration, it also reduced mycelial growth (Bowen 1983). Increased nutrients from animal waste may increase the growth of fungal hyphae as they readily penetrate decomposing organic matter and out compete other soil organisms and plant roots for nutrients (Bowen 1973). An increase in nutrients available to native species above optimal levels, from the increased mycorrhizal hyphae growth, would have no benefits for native vegetation, as discussed previously.

The increase in water repellency seen in ungrazed wandoo sites in this study, may be related to past fire events. Fire increases water repellency (Adams *et al.* 1970; De Bano *et al.* 1970; De Bano and Rice 1973; Scott 1997). However, the burning history of these sites is unknown. The observed surface soil water repellency in both the wandoo and jarrah-marri remnants are classed as severe (2.4 – 3.0 MED) and very severe (3.2 – >3.8 MED) (King 1981). Severe water repellency also leads to surface water flow (Scott 1997) which moves nutrients and weed seed into remnants and causes soil erosion with water moving off of remnants.

Grazing exclusion, as a single management practice, will only be useful where remnant vegetation has been infrequently grazed and impacts are low. More intense grazing will have resulted in a depleted soil seed bank and changed soil chemical and physical parameters that do not favour the establishment of native species. Recently, the concept of crossing thresholds in the degradation of Australian remnant woodlands has been developed (MacLeod *et al.* 1993; Hobbs and Norton 1996). A degrading remnant may cross a threshold such that any attempt at restoration will involve a much greater degree of management than simply removing the disturbance. Additional management is required to increase native species recruitment and decrease the presence of exotic species.

5. Fire as a rehabilitation tool

5.1 Introduction

Fire is a natural component of the forest and woodlands of south Western Australia and native vegetation has developed adaptations to regular occurrence to fire. A study of 300 jarrah forest understorey species showed that 70 to 74% of species resprout following fires and the remainder regenerate from seed stored either in the canopy or the soil (Burrows *et al.* 1995). Many native species (especially the families Mimosaceae and Fabaceae) require the heat of a fire to break the hard seed coat and stimulate germination (Christensen and Kimber 1975; Shea *et al.* 1979, McCaw 1988, Portlock *et al.* 1990). Increased regeneration of wandoo seedlings was mostly dependent on seed germination in ashbeds created by burning logs and limbs on the ground (Hatch 1960; Burrows *et al.* 1990; Mercer 1994). Smoke has been found to stimulate germination in fynbos species in families including Asteraceae and Restionaceae, and enhance germination in species of fynbos Proteaceae (Brown, 1993). It has also been used to increase germination rates of jarrah forest species (Dixon *et al.*, 1995).

The use of fire may be a valuable tool in enhancing natural regeneration of remnants once livestock is excluded. Pettit and Froend (1994) sampled soil in jarrah-marri woodlands from sites that had been heavily grazed, lightly grazed and ungrazed and were heated at 70°C for 20 minutes. At all sites heating significantly decreased the germination of exotic annual species. A slight increase in germination was seen for native species and this was most significant in samples from the ungrazed site. Although fire may also be useful in reducing competition from weeds long enough for native species to become established, if there is a large store of weed seed in the soil, high soil nutrient levels combined with soil disturbance may promote weed cover (Hobbs and Atkins, 1988).

Fire season may be important in determining the response of the vegetation to burning, particularly the vegetative regrowth and seedling establishment of native perennial species and the invasion and establishment of exotic species. Burrows et *al.* (1990) found that the number of wandoo seedlings increased and the number of exotic species decreased, in ashbeds following an autumn burn. High temperatures in the top soil beneath burning logs were thought to have killed the seeds and roots of grasses and other herbs. In the study by Pettit and Froend (1994), higher germination of exotic annual species in samples collected in the winter indicated that hotter autumn fires may be more effective in reducing weed seed numbers than fires at other times of the year. The season of fire did not significantly affect the floristic composition of seedling regeneration in a jarrah-marri forest, however dry soil fires in summer/autumn resulted in higher numbers of initial seedlings and higher survival rates in the first year (Burrows et *al.* 1995).

Burning will reduce the weed seed in the soil, however seed buried at depth will generally escape all but the hottest fires and as fires tend to be patchy, seed in unburnt areas will be unaffected (Burrows *et al.* 1995). The authors suggested that a hot autumn burn would be most effective in reducing weed seed in the soil. However, those seeds that survive are likely to have ideal conditions for growth with elevated levels of soil nutrients (especially N) and reduced competition from native perennial species. Spring burning may be effective in destroying the winter crop of weeds after they have germinated and prevent seed set. It may also allow the native regrowth to establish for six to nine months before experiencing competition from exotic annuals the following winter. However this would be over the period of summer drought which may affect survival of native seedlings and resprouts.

The objectives of this study where to:

1. Compare the effects of an autumn and a spring burn on the numbers and abundance of exotic species in a jarrah-marri remnant.

2. Compare the effects of an autumn burn on a jarrah-marri remnant containing sites with varying levels of livestock disturbance and weed invasion.

5.2 Materials and methods

5.2.1 Site description and burning treatment

The study was conducted in the Wellington reservoir catchment, in a 213 ha remnant that had experienced past grazing disturbance and consequent invasion of exotic species and loss of native species. The area was divided into two with one half burnt in October 1994 (Spring burn) and the other in April 1995 (Autumn burn). The Department of Conservation and Land Management carried out the fire management. On either side of the fire break three 10 m × 10 m quadrats were established. The area either side of the firebreak was considered to be in fair condition after limited disturbance from livestock grazing. Three 10 m × 10 m quadrats were also placed at two other sites in the autumn burn block in areas of moderate livestock disturbance and weed invasion and at a site with high livestock disturbance and weed invasion. Therefore in the spring burn area there was one site comprising three quadrats (1a, 1b and 1c) and in the autumn burn area there were a total of three sites, each with three 10 m × 10 m quadrats; fair (2a, 2b and 2c), moderate (3a, 3b and 3c) and poor (4a, 4b and 4c) condition sites.

5.2.2 Vegetation survey

Within each quadrat, species richness (number of species present) and relative cover of each species were recorded. Species were identified using local floras (Blackall and Grieve, 1980; Blackall and Grieve,1981a; Blackall and Grieve, 1981b; Blackall and Grieve, 1982; Marchant *et al.*, 1987a; Marchant *et al.*, 1987b; Blackall and Grieve, 1988; Hussey *et al.* 1997). Relative cover was estimated using the Domin-Krajina scale (Mueller-Dombois and Ellenberg, 1974). The presence and cover of the seedlings of overstorey species were recorded and analysed as part of the understorey. The pre-burn surveys were conducted in October 1994 immediately before the spring fire and six months prior to the autumn fire. The sites were then surveyed in October 1995 (six and twelve months after autumn and spring burning respectively) and in October 1996 (18 and 24 months after burning).

5.2.3 Site characterisation

Post fire soil samples were taken. At each site the soil profile was described to 1 m (McDonald *et al.*, 1990). Three soil samples were taken at each site (0–10 cm) and analysed for nitrate, phosphorus and potassium (Rayment and Higginson 1992). Ten surface soil samples (top 1cm) were taken at every site and analysed for water repellency using the molarity of ethanol drop method (MED)(King, 1981). Nine surface cores were also taken per site and the bulk density (g cm^{-3}) of the top 10 cm of soil determined (Loveday, 1974).

5.2.4 Statistical and non-parametric analysis

Statistical and non-parametric analyses of the data were completed using Statistica 5.1 software (Statsoft 1984–1996). Species richness and soil data were examined for normal distribution using the Shapiro-Wilks W test (Zar 1984). Analysis of Variance (ANOVA) was then used to examine the effects of burning season and livestock disturbance on species richness and soil chemical and physical characteristics. Bartlett's test was conducted to determine if the data met the assumption of homogeneity of variance (Zar 1984). Where they did not, species richness data were transformed to their log values and the soil data to log values. Transformed data were then analysed for variance. Multivariate analysis of variance (MANOVA) was used to determine which soil chemical and physical properties differed significantly between sites with different burning season and grazing histories. Post hoc comparison of means was undertaken with the Neuman-Keuls test and power analysis of each ANOVA was also undertaken (Zar 1984).

Relative cover data were ordinated using Multidimensional Scaling. The ordination matrix was prepared by using the coefficient of squared Euclidean distance measures (Kent and Coker 1996) with unweighted pair-group method using arithmetic averages (UPGMA) linkage rules (Sokal and Michener 1958; Kent and Coker 1996). The resulting output was then examined for groupings of sites based on the cover of each species present.

5.3 Results

5.3.1 Soil profile

Soil profiles showed all sites to be duplex (texture contrast) soils with yellow/red clay sands containing gravel to 1 m.

5.3.2 Autumn v spring burning

Pre-burn, there was no significant difference between the mean number of native species in autumn and spring burnt plots (48.7 compared to 42.7) (Figure 5.1a). However, there was a greater mean number of exotic species in the autumn burn plots (9.7 compared to 6.7). After 12 months, there was a significantly greater mean number of native species in the spring burnt plots (41.3) compared to the autumn burnt plots (36.7), but no significant difference between the number of exotic species present for each treatment (Figure 5.1b). After 24 months there was no significant effect of burning season on the number of native species or the number of exotic species (Figure 5.1c).

The number of native species declined over time, for both spring and autumn burning treatments (F_{year}=81.8 (d.f.=2, P=0.00)). In spring burnt plots, the number of exotic species significantly increased after 12 months, but then decreased to numbers less than pre-burn levels after 24 months (F=15.8(d.f.=2, P=0.00). In autumn burnt plots, there was no significant difference between pre-burn levels of exotic species and numbers found after 12 months. There was then a significant decrease after 24 months (F_{year}=72.3 (d.f.=2, P=0.00).

Autumn and spring burnt sites were not separated when the relative cover of each species was ordinated.

Soil water repellency was higher in the spring burn sites (2.06 M) than the autumn burn sites (0.64) (F=15.23, d.f.=1, P=0.001). Nitrate, Phosphorus and Potassium were not significantly different between spring and autumn burns.

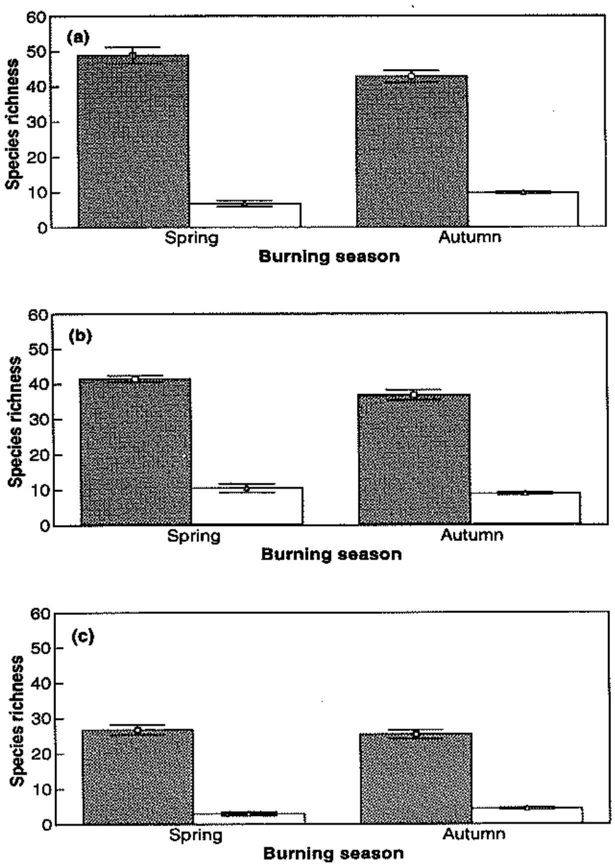

Figure 5.1

Species richness of jarrah-marri remnant sites in the Wellington catchment pre-burn (a); 6 and 12 months (b); and 18 and 24 months (c) after spring or autumn burning. Mean number of native species are shown as solid bars and mean number of exotic species as open bars. Vertical bars indicate standard errors of the means.

a: $F_{native\ spp.}$ = 4.4 (d.f.=1, P=0.10);

$F_{exotic\ spp.}$ = 10.1 (d.f.=1, P=0.03).

b: $F_{native\ spp.}$ = 7.5 (d.f.=1, P=0.05);

$F_{exotic\ spp.}$ = 1.8 (d.f.=1, P=0.25).

c: $F_{native\ spp.}$ = 5.3 (d.f.=1, P=0.52);

$F_{exotic\ spp.}$ = 0.6 (d.f.=1, P=0.11).

5.3.3 Autumn burning and livestock disturbance

Pre-burn, fair and moderate condition sites had similar mean numbers of native species (42.7 and 37.7), which were greater than that found in poor condition sites (20.7) (Figure 5.2a). Conversely, fair and moderate condition sites had similar mean numbers of exotic species (9.7 and 7.0) which were significantly lower than the number of exotic species in poor condition sites (15.0).

Six months after burning, the differences between sites were maintained and number of native species (36.7, 38.3, 21.0) and number of exotic species (8.7, 6.3, 13.7) were not significantly different from their pre-burn levels for fair, moderate and poor sites respectively (Figure 5.2b). Eighteen months after burning, species distribution between fair, moderate and poor sites still remained the same, but species numbers were significantly lower than in previous years. The fair site contained means of 25.3 and 4.3 native and exotic species respectively, the moderate site 27.7 and 3.3 species and the poor site 10.7 and 6.3 species (Figure 5.2c).

Pre-burn, there was no distinct separation of quadrats based on grazing condition, when relative cover data were ordinated (Figure 5.3a). Six months after burning, fair and moderate sites are clustered together and separated from poor condition sites (Figure 5.3b). Twelve months after burning, this separation based on species cover is more evident (Figure 5.3c).

Bulk density was significantly higher for the moderate site (1.12 g cm^{-3}) than for the fair and poor condition sites (0.69 and 0.89 g cm^{-3}) (Table 5.1). Potassium soil content was similar in the fair and poor sites (100.0 mg kg^{-1} and 115.3 mg kg^{-1}) and higher than in the moderate site (56.0 mg kg^{-1}). Phosphorus and Nitrogen soil content and surface soil water repellency were not significantly different between sites of different condition.

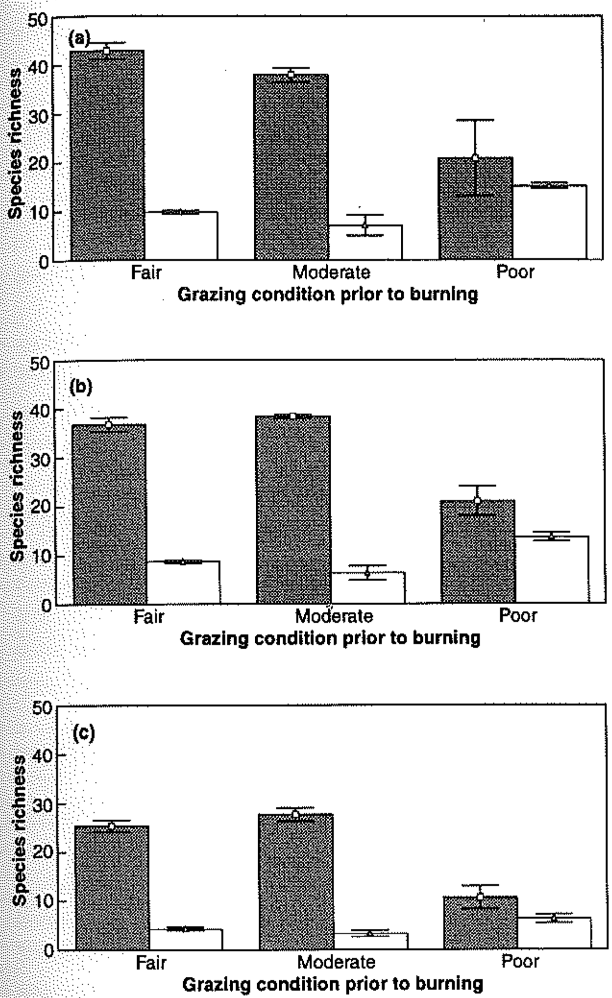

Figure 5.2

Species richness of jarrah-marri sites with different grazing histories pre-burn (a); 6 months (b); and 18 months (c) after autumn burning. Mean number of native species are shown as solid bars and mean number of exotic species as open bars. Vertical bars indicate standard errors of the means.

a: $F_{\text{native spp.}} = 6.2$ (d.f.=2, P=0.030);

$F_{\text{exotic spp.}} = 10.4$(d.f.=2, P=0.010).

b: $F_{\text{native spp.}} = 24.4$(d.f.=2, P=0.001);

$F_{\text{exotic spp.}} = 14.0$(d.f.=2, P=0.005).

c: $F_{\text{native spp.}} = 29.4$(d.f.=2, P=0.001);

$F_{\text{exotic spp.}} = 5.2$(d.f.=2, P=0.050).

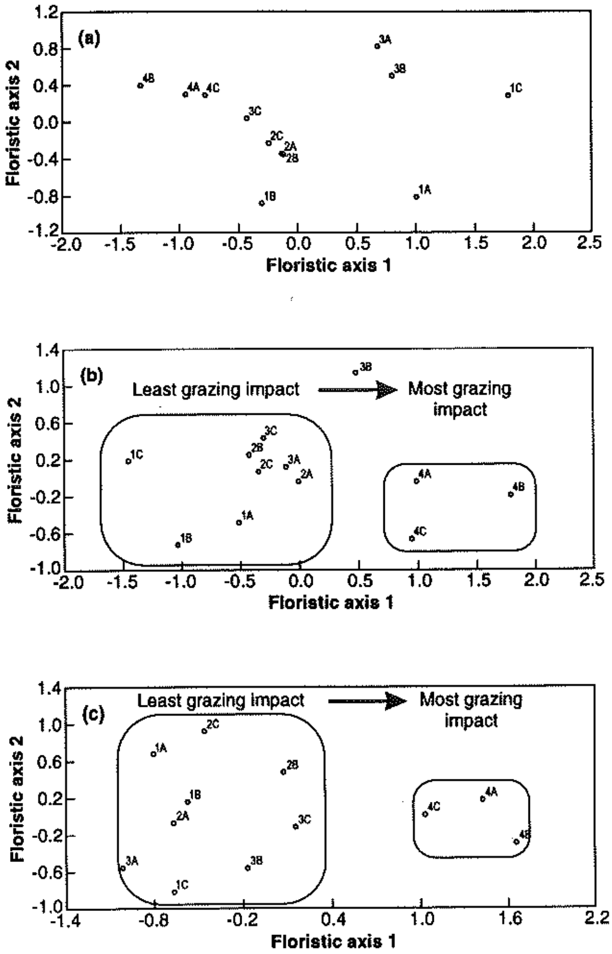

Figure 5.3

Ordination analysis using multidimensional scaling of relative cover of each species present in jarrah-marri sites pre-burn (a); 6 months (b); and 18 months (c) after autumn burning. After 6 months, groups are separated along floristic axis 1 by increasing grazing impact prior to burning.

5.4 Discussion

Burning in spring resulted in an increase in the number of native species in quadrats compared to autumn burning, however this difference was not maintained after 24 months. Spring or autumn burning did not affect the number of exotic species present in quadrats. Fire season also had no effect on relative cover of species.

Published reports of successful species regeneration in jarrah communities after autumn rather than spring burning have mostly described results obtained from intact forest burnt for fire control (Peet 1971; Floyd 1976; Shea *et al.*

Table 5.1

Effect of site condition on significant soil chemical and physical properties. Standard errors are shown in brackets.

Condition of site	BD (g cm^{-3})	K (mg kg^{-1})
Fair	0.69 (0.01)	100.0 (7.4)
Moderate	1.12 (0.02)	56.0 (7.6)
Poor	0.89 (0.01)	115.3 (11.3)
F, df, P	10.48, 2, 0.0005	7.92, 2, 0.02

1979; Monk *et al.* 1981; McCaw 1988). Cowling and Lamont (1987) attribute this effect to the increased time that seed remains on the ground after spring burning, prior to winter germination, thus increasing the likelihood of seed predation. However, Burrows *et al.* (1995) describe jarrah forest sites that had been burnt every five to eight years and which showed no significant difference in species richness between spring and autumn burns. As discussed previously in Chapter four, seed production may be much less in remnants that have been degraded by livestock grazing (Pettit and Froend 1994; Yates *et al.* 1994). Even following disturbances such as fire which promote germination and early establishment, there may not be enough seeds for recruitment.

In all quadrats, the numbers of native and exotic species declined after 24 months. This effect may be related to rainfall or other climate factors that have disproportionately affected annual species. Fewer native and exotic annual species were identified in the 1996 survey than in 1994 and 1995.

Autumn or spring burning did not differentially effect the nutrient status of the soil. However, water repellency was greater in soils burnt in spring than in those burnt in autumn. The dry conditions prevailing in autumn fires produce more heat than spring burning and degree of water repellency has been shown to be positively related to fire intensity (Scott and Van Wyk 1990). However, after a hot fire, the water-repellent layer is usually beneath the soil surface (De Bano and Rice 1973). Organic substances in the litter layer and O horizon are vaporised during hot fires. Some of this is lost to the atmosphere but part of it moves downward along steep temperature gradients and condenses in cooler soil layers, causing a water-repellent layer (De Bano *et al.* 1970). Cooler fires result in water repellency at or near the soil surface (De Bano and Rice 1973). Fire induced water repellency is greatly reduced after one year but may still be evident up to ten years after a hot fire event (De Bano and Rice 1973).

The extent of previous grazing and subsequent condition of the remnant before burning took place did not affect the numbers of native or exotic species present after burning. However, it did affect the successful regrowth and cover of species. Similarly to exclusion of livestock grazing in jarrah-marri remnants in the Denmark River catchment (Chapter four), sites least grazed prior to amelioration showed most increase in cover of species. Visual monitoring of remnants may give rise to the assumption that remnants are being restored after burning, when in fact no native species recruitment has occurred. Perennial species already present are increasing their cover and this may be due more to lack of current grazing than any effect of burning.

Water repellency was similar in all grazed sites indicating a uniform heating effect of the autumn fire. Although bulk density and Potassium levels differed between sites, this did not appear to be related to effect of burning or prior grazing condition.

Heat treatment has been shown to be effective in reducing germination of exotic annual species (Hobbs 1989; Burrows *et al.* 1990; Pettit and Froend 1994). In the study described here, fire had no effect on the numbers of exotic annual species present in quadrats. Studies in the Wheatbelt have shown that fire did not increase the degree of weed invasion in native remnants, providing other types of disturbance were absent (Hobbs 1991b). However, the promotion of exotic species from frequent fires in remnants has been observed in many studies (Baird 1977; Bridgewater and Backshall 1981; Wycherley 1984; Hopper and Burbidge 1989; Hopkins and Griffin 1989; Hobbs and Atkins 1990; Hester and Hobbs 1992; Hussey and Wallace 1993; Milberg and Lamont 1995). These increases in exotic species have been attributed to soil nutrient increases (Cale and Hobbs 1991) and the removal of native perennial vegetation by fire which opens up more establishment sites for exotic species (Milberg and Lamont 1995). The proliferation of exotic species then leads to an increase in fire from high fuel loadings, which in turn promotes further growth of exotic species (Wycherley 1984; D'Antonio and Vitousek 1992).

In the study reported here, neither spring nor autumn burning led to a significant increase in species recruitment. Fire resulted in increased cover of perennial native species, but only in areas that had limited, previous livestock grazing. Previous studies have found that acute disturbances such as fire are of less significance in rehabilitation of remnants compared to the chronic disturbance from livestock grazing and nutrient input (Hobbs 1991b; Hobbs and Atkins 1991; Hester and Hobbs 1992). This study and those reported in Chapter Four, support those findings.

6. Management strategies for remnant vegetation

6.1 Management objectives

The long-term sustainability of a remnant is dependent on many factors including size and shape; degree of isolation; position in the landscape and vegetation type. It is more acutely dependent upon the impact of disturbances such as salinity, waterlogging, livestock grazing, high nutrient inputs and erosion. This report and previously published studies, have demonstrated that the most extensive disturbance comes from the frequency, duration and rate of livestock grazing.

The majority of remnants on private land in the water resource recovery catchments have been subject to such disturbances, which have resulted in loss of species richness and abundance. This in turn has resulted in changes to the ecosystem processes and functions that existed previously in a sustainable vegetation community. The degradation of remnants and subsequent loss of biodiversity and ecosystem functions will continue if active management is not imposed. Management of remnant vegetation requires the management of disturbances and the restoration of ecosystem functions. It should focus on developing a vegetation community based on native perennial species that is similar to the original and fulfils a functional role in the agricultural system, such as maintaining the hydrologic balance, reducing the risk of erosion and contributing to nature conservation (Lambeck and Saunders 1993).

The objective of the Water and Rivers Commission is to facilitate the management of water resource recovery catchments to maintain or restore the water quality to potable levels. The Commission has engaged with landholders in each catchment and staff from other Government agencies to form Recovery Teams which will oversee the implementation of strategies designed to meet this objective. One such strategy is the protection of remnant vegetation from further degradation and to promote rehabilitation of previously degraded remnants to optimise water use and conserve biodiversity.

Fencing subsidies specifically for remnant vegetation in water resource recovery catchments have been made available through the Western Australian Salinity Action Plan (1996; 1998). The plan recognises the role of remnant vegetation in contributing to the control of salinity and protection of nature conservation values. Funding and management guidelines are available through the Commission and the Recovery Teams to ensure that remnants on private land in these catchments are fenced by 2010 and managed to ensure their long-term retention.

6.2 Management issues

6.2.1 Livestock grazing

The results of surveys carried out as part of this project and the existing literature on the impacts of livestock grazing on remnant vegetation have been discussed extensively in Chapter four. Briefly, this project has shown that sustained livestock grazing of remnants leads to loss of understorey species and overstorey recruitment, increase in exotic species and increase in soil compaction, soil nutrients and erosion. Grazing exclusion will only ameliorate sites which have a history of limited grazing (Figure 6.1a). Where grazing has been more intense, protection from grazing will prevent further deterioration and the cover of some native perennial species may increase (Figure 6.1b). However, the depleted soil seed bank and changed soil chemical and physical parameters does not promote recruitment of native species.

6.2.2 Control of exotic species (weed invasion)

Disturbance of remnant vegetation, particularly from livestock grazing and increased soil nutrient levels, has resulted in the invasion of exotic species from adjacent agricultural lands. Where this is most severe, the original vegetation community has moved to one dominated by annual exotic species.

Fire has often been used as a weed control tool, however it is not recommended. Although fire is an essential natural disturbance in remnants (section 6.2.3), frequent burning has been shown to lead to increased invasion of exotic species. Burning of road side vegetation in southern Western Australia caused a significant increase in weed

Figure 6.1 **Grazing exclusion may ameliorate sites which have a history of limited livestock grazing (a). Regeneration may not result after protection of remnants where grazing has been more intense (b). Revegetation and weed control will be required to restore the understorey and promote recruitment of overstorey species.**

Figure 6.2 Burning for weed control did not result in regeneration of native species in this remnant in the Wellington Reservoir catchment. Instead, an increased cover of exotic species resulted which will increase the fuel load and inhibit the establishment of native species.

species numbers and abundance (Milberg and Lamont 1995). This increase was still evident after seven years, while unburnt sites showed no change in numbers of weed species over this time.

Invasion of eucalypt woodlands by the exotic grass species *Ehrharta calycina*, has increased the frequency and intensity of fires and established a cycle which promotes fire and further invasion and causes native species decline and extinction of fire sensitive species (Wycherly 1984; Panetta and Hopkins 1991; Milberg and Lamont 1995). In intact jarrah forest, the exotic species *Aira caryophylla* had built up a significant soil seed store (27 seeds m^{-2}) (Vlahos and Bell 1986). The authors hypothesise that the regime of fuel reduction burns may have provided a habitat more conducive to the invasion of introduced annuals. The promotion of weed invasion by frequent burning has also been observed in many other studies (Baird 1977; Bridgewater and Backshall 1981; Hopper and Burbidge 1989; Hopkins and Griffin 1989; Hobbs and Atkins 1990; Hester and Hobbs 1992; Hussey and Wallace 1993) (Figure 6.2).

Most recommended herbicides for weed control in remnant vegetation are not specifically registered 'bushland' herbicides, but are widely used in agriculture. As a general rule, it is best to use those herbicides that do not persist in the soil or are easily leached into adjacent land or streams. The effects of many of these herbicides on native plants and their effectiveness in bushland settings are not known. Records should be kept of what herbicides are used on sites with application methods and rates. This information can then be used to determine what control methods have been most successful.

A successful weed control programme has three phases (Dixon and Keighery 1995a).

- Primary weeding—first removal
- Secondary weeding—removing weeds which germinate after primary weeding and associated soil disturbance. This phase may last a few months or up to a year depending on the weed species present. It is especially important for the survival or regenerating native plants.

- Long term maintenance—site may only need maintenance annually to remove any scattered weeds that have re-invaded.

For general blanket spraying, Fusilade 212®, Assure ®, Targa®, Sertin® and Verdict® have been used in bushland trials since 1985 (Dixon and Keighery 1995a). These herbicides are suitable for a wide range of native plants for the control of grass species such as perennial and annual veldt grass, couch and kikuyu. Glyphosate-based herbicides such as Round-up® do not persist in the soil so do not affect native seedling germination or replanting. However, they are non-selective and will kill all plants. They are better suited to direct application or spot spraying methods. Sprayseed 200® and Tryquat® are also non-selective contact herbicides which are quickly inactivated in the soil allowing replanting the following day.

A comprehensive list of herbicides for specific weed species and notes for application are tabled in Dixon and Keighery (1995b). Further information on the application, mixing, calibration and toxicity of herbicides is found in Dodd *et al.* (1993). Information of the safe use of herbicides can be obtained from Agriculture Western Australia or the Western Australian Health Department.

6.2.1 Fire

Successful seedling regeneration after fire has mostly been reported in remnants that have not been subjected to significant disturbance such as livestock grazing (Chapter five). An exception to this was a study by Mercer (1994) that described increases in wandoo regeneration in ashbeds. In most cases, where remnants had been heavily grazed, fire only served to increase weed invasion and did not increase the recruitment of native species. As discussed above, due to the many studies that have reported increased weed invasion that often follows burning, fire is not encouraged as a method of weed control.

Fire is a useful tool in the long term management of remnants once species have re-established. Plant communities have evolved to survive fire regimes as opposed to single fire events (McArthur and Cheney 1966; Gill 1975). Resprouting after fire is the most common method of regeneration in the drier jarrah forests. Seventy percent of plants in this community have the capacity to regenerate from subterranean organs after the shoot system has been destroyed or severely damaged (Christensen *et al.* 1981; Gill 1981). This use of starch resources allows rapid refoliation and growth after fire which in turn allows the plant to exploit water and nutrients at a greater initial rate than reseeder species (Pate *et al.* 1990). Resprouter species must have enough time to build up adequate starch resources before the next fire event.

Reseeder species, typically legumes, produce large quantities of hard seeds which accumulate and survive in the soil for long periods and germinate after being heated (Christensen and Kimber 1975; Christensen *et al.* 1981). These species have fast shoot growth and are able to reproduce relatively quickly (Pate *et al.* 1990). This allows them to maximise seed reproduction and establish an effective seed bank before the next fire event occurs which will kill the mature plant.

Resprouter species also produce seed which directly competes with the seedlings of obligate seeders following fire. As seedlings from the latter frequently outnumber those from the former by 20:1 (Pate *et al.* 1990), and shoot growth rate of reseeders is faster than that of resprouters, resprouter seedling survival is very low. These seedlings are also not sufficiently mature to carry sufficient below ground starch reserves to survive a short interval before the next fire (Pate *et al.* 1990).

Frequent, low intensity fuel reduction fires will remove those plant species which require longer, fire-free periods to produce seeds (Adamson and Fox 1982; Main 1987) or rely on hot fires for germination (Shea *et al.* 1979; McCaw 1988; Portlock *et al.* 1990). This type of fire regime will favour annual and short lived plants and remove fire sensitive species. Loss or reduction of species under new fire regimes may have effects which do not become evident for a long time (Christensen *et al.* 1981). For example, changes in the relative abundance of legumes may alter the nitrogen cycle of the ecosystem.

Much of the southern forest is prescribed burned with low intensity fires, by CALM, for fuel reduction purposes. Most of the fires occur in spring and early summer. Five to 15% occur in autumn. Fire frequency is five to seven years in jarrah forest and seven to nine years in karri forest (Christensen and Annels, 1985). In special conservation areas, burning is carried out on a longer rotation, between 15 to 20 years, during summer and autumn. High intensity wild fires occur in the region and should be considered as part of the fire regime. Approximately 10% of the southern forests have been affected by wild fires over the last 30 years. However, because of improved prevention, detection and suppression of wild fires, their contribution to the fire regime appears to be decreasing.

Studies in jarrah forests have shown that is difficult to maintain a three to four year cycle of burning (Christensen *et al.* 1981). Short fire intervals have been shown to cause the reduction of a multiple layered understorey to a single layer (Gilbert 1959) and a 66% reduction in shrub density (Christensen *et al.* 1981). Estimates based on the proportion of resprouting species in the floristic composition of jarrah forest indicate natural fire intervals of less than 25 years (Bell 1985). Species diversity declines significantly in jarrah forest sites which have had greater than 30 years since the last fire (Bell and Koch 1980). A fire free period of about six years in intermediate and high rainfall jarrah forests was required and about eight years for low rainfall forests, for the restoration of seed banks for all seeder species (Burrows *et al.* 1995). Species richness was lowest in plots which had not been burnt for 12 years.

Cowling *et al.* (1990) advised the following management strategies for fire management of south western Australian *Banksia spp.*. Fire intervals should not be less than ten years. Non-sprouting species take three to five years to set seed and in many non-sprouting species, it takes more than ten years for substantial follicle production to occur (Cowling *et al.* 1987; Lamont and Barker 1988). Even more extreme, seedlings of *B. tricuspis*, a resprouter, require more than 20 years to mature and develop fire resistance, after which mature plants can tolerate short fire intervals (Lamont and van Leeuwen 1988). Upper limits of fire regimes have been harder to determine due to lack of older stands (Cowling *et al.* 1990). However, senescence of *B. prionotes* has been recorded at around 30 years and at 15 years for *B. coccinea.*

Karri tends to be fire sensitive and does not produce lignotubers; rather it relies on bark protection of buds. Under natural fire regimes, fires are infrequent , large masses of fuel accumulate and mortality is high after fire (Gill 1981). The even aged stands of these forests which were evident at first settlement are thought to have arisen this way (Ashton 1976). Although seeds fall each year, regeneration is greatest following fire when seed fall can be rapid and heavy, fungal infection is reduced, shade is removed and increased nutrients are available in the seed bed (Gill 1981). Successful regeneration of logged Karri forest may result from the decreased shade and increased soil nutrients following disturbance, which partly mimics the effects of fire.

When the vegetation of south Western Australia is considered as a whole, fire tolerance is closely related to environmental variables (Bell *et al.* 1989). Tree species characteristic of lateritic uplands and fertile valley sites are most tolerant of fire (*E. marginata, E. calophylla, E. patens*). These sites show rapid fuel accumulation, regular seasonal drought and often a steeper topography which render them prone to periodic fires with a potential for a wide range of intensities. Lower fire tolerance is characteristic of species growing in consistently moist environments where fire is naturally restricted (*E. rudis, E. megacarpa*), where there is slow fuel accumulation (*E. wandoo*), or where physical barriers such as rocky outcrops impede fires (*E. laeliae*).

6.2.2 Size and edge to area ratio

Large remnants generally contain a wider diversity of habitats than smaller ones, however a group of smaller reserves may ultimately cover a greater diversity of habitats because one large reserve is not likely to contain all of the habitats which occur in a region (Saunders *et al.* 1991). Larger remnants are more likely to have larger populations of species and more likely to resist extinction (Soule 1987). However, it has been suggested that a population size of 500 was required to maintain genetic diversity in *Eucalyptus albens* (white box) remnants (Prober and Brown 1994). Many *Eucalyptus* species have similar breeding systems (Moran 1992; Prober & Brown 1994), which indicates that many remnant populations in south western Australia may be below the level of individuals required to maintain genetic diversity (Yates and Hobbs 1997).

Edge effects have a greater impact on smaller remnants as larger remnants have a greater internal area which is much less affected by the environmental changes occurring at the edge (Williamson 1975; Janzen 1983; Harris 1988; Yahner 1988). This is important given the small size of many of the remnants in south western Australia with high edge to area ratios. Many road verge corridors in the Western Australian Wheatbelt are less than five metres wide (Griffin and Hopkins 1991). However, long thin remnants may lie along environmental gradients and thus contain more vegetation types and habitats than a square reserve of a similar area (Saunders *et al.* 1991).

Buffer strips extend the effective edge of remnants and protect against in-blown nutrient and particulate inputs, wind damage and weed invasion (Hester and Hobbs 1992; Hobbs 1993b; Scougall *et al.* 1993). Buffer strips can be established using commercial tree species with both high water use and commercial timber attributes. The required width for buffer strips depends upon the type of disturbance which is trying to be minimised. A narrow buffer strip can stop wind-borne nutrients and particulate matter while weed seeds may be transported considerable distances requiring a wider buffer strip.

Riparian remnants have specific problems which are not shared by remnant vegetation in other parts of the landscape. By their nature, riparian remnants often have a large edge to area ratio which results in high costs associated with vermin control and protection from fire and disease (Watson 1991). However, these ecosystems are important in maintaining the water quality of rivers and wetlands, the integrity of river and wetland banks and contain species unique to riparian environments. They often form important natural corridors that link separate preserved areas (Watson 1991; Recher 1993).

6.2.3 Changed hydrology

As recharge in catchments has increased from wide scale clearing of native vegetation for agriculture, remnants which occur down slope from large areas of cleared land and in riparian zones, receive increased water, nutrients, sediment and salt loads (George *et al.* 1995). These disturbances may eventually cause loss of indigenous species in these remnants. Surface water moving from cleared farmland also facilitates the introduction of exotic species into remnants. Water movement and increased soil moisture is also a more favourable environment for *Phytophthora cinnamomi*, the fungus which causes jarrah die back (Shearer and Tippett 1989).

It has been estimated that if current recharge rates remain unchanged, up to 25% of many landscapes, and as much as 40% to 50% of areas such as lower slopes and valley floors, will become salt affected within the next century (George *et al.* 1995). Many woodlands are now saline or waterlogged for most of the year and the vegetation has disappeared or has only dead trees remaining. Some specialist plant communities, such as swamp yate (*Eucalyptus occidentalis*), salt salmon gum (*E. salicola*) and many *Casuarina* and *Melaleuca* populations could disappear at local and regional levels. Riparian vegetation fringing previously fresh water courses and lakes is also being adversely affected by increasing saline inputs (Froend *et al.*, 1987; Bell and Froend, 1990; Froend and McComb, 1991; Saunders *et al.*, 1991). In the Warren River catchment, saline water from the Unicup lakes and surrounding farmland are threatening Kodjinup Lake and its reserve, and the Buranganup Plain wetlands (George and McFarlane 1993). Some saline lakes within the catchment are beginning to fill with groundwater and overflow into adjoining reserves.

Protecting areas of remnant vegetation from the consequences of changed hydrological conditions will be achieved through whole catchment management to stabilise or reduce rising groundwater and salinisation. In the short term however, it is advisable that management costs and efforts are mainly expended in those remnants which are not under immediate threat of degradation from waterlogging and salinity. An individual remnant which has been reduced to a structure of canopy and grass and occupies a strategic place in the landscape in regard to high water use may be a higher priority to restore and protect than a more intact remnant in immediate threat of salinisation.

6.2.6 Vermin management

Control of pest species is an important part of management and rehabilitation of remnant vegetation. The impacts of rabbits, foxes and feral cats have been well documented. Rabbits are particularly destructive to regeneration as they selectively graze seedlings and resprouting shoots. If grazing pressure becomes too great or drought occurs they will also ring bark trees and eat roots and less palatable plants. Feral pigs are also destructive to vegetation and may be a problem in larger remnants or those bordering State Forest in south Western Australia. Kangaroos can damage regenerating plants where the carrying capacity of the remnant had been exceeded. Removal of foxes from reserves have led to significant increases in endangered species such as Numbats (*Myrmecobius fasciatus*), Woylies (*Bettongia penicillata*) and black footed rock wallabies (*Petrogale lateralis*) (Kinnear *et al.* 1988; Friend 1990).

Feral cats, foxes and rabbits need to be controlled at the same time to avoid either increases in rabbits or increased predation on native fauna by cats and foxes. Poison control must be on-going to prevent reinvasion of vermin species. Advice on the most effective vermin control methods can be obtained from Agriculture Western Australia. A detailed discussion of vermin control is also found in Hussey and Wallace (1993).

6.2.7 Revegetation

As regeneration has been shown to be low in eucalypt woodlands, revegetation is required to re-introduce species that might otherwise never or only slowly recolonise a site. While there are no published studies demonstrating revegetation within woodland remnants (Yates and Hobbs 1997), there are many publications describing the revegetation of cleared land and mine site rehabilitation in Western Australia, using direct seeding and seedling establishment (Edminston 1985; Greening Australia 1990; Loney 1990; Lefroy *et al.* 1991; Hussey and Wallace 1993; Scheltema 1993).

Direct seeding does not carry the costs associated with purchasing seedlings and large areas can be sown in a short time. A mix of trees and understorey species can be sown at the same time and varied to suit different soil types and different positions in the landscape. In addition, plants from seed more quickly adapt themselves to the natural environment. The main disadvantage of direct seeding is that it can result in complete failure, particularly if the season following seeding has below average rainfall, or uneven establishment (Scheltma 1993). In addition, many *Eucalyptus* and grass species are subject to granivory by ants (Mott and McKeon 1977). Seed may be pelletised, then treated with pesticide or drilled into the soil. Alternatively the amount of seed sown may need to take into account the proportion lost to ants (Majer 1990). Timing of seeding will also influence the amount of ant predation. In south Western Australia, seeding in May coincides with the time that ant activity is declining, due to the colder weather.

Detailed planning before seeding increased establishment and survival rates of seedlings (Loney 1990).

Direct seeding activities need to fit into the annual farm plan. The best season for germination of rehabilitation species needs to be known. In general, May and June sowing times have been optimal (Lefroy *et al.* 1991; Pigott *et al.* 1994), after the first winter rains, to allow time for adequate root establishment so that summer watering is not necessary. However, direct seeding on the south coast has been most successful in August and September (Scheltema 1993). Where other farm activities and optimal time for direct seeding conflict, it may be better to employ a specialist contractor for the purpose. This may also be an option where the amount of site amelioration required before seeding or replanting, or the pre-treatment of seeds before sowing, requires machinery or expertise not held by the individual landholder. On-going management of the site is then taken over by the landholder. Grewar (1986) used both a standard combine and an air seeder with good results in the Esperance area. Depending on the size of the area to be revegetated, hand broadcasting of seed on prepared sites is an effective technique.

Local species should be used for direct seeding purposes. Species used should also match the correct soil type and landscape position of the area to be revegetated (Lefroy *et al.* 1991). It is also important to ensure that the existing remnant vegetation will not be compromised i.e. species used must be compatible with those in the remnant and species which are liable to become invasive should be avoided (Hobbs 1993b). Seed collection and storage is time consuming and requires a degree of specialist knowledge to ensure that viability and genetic diversity are maintained. Local seed merchants are a source of seed or catchment groups may consider establishing seed orchards to provide a ready source of local seed. Some basic outlines on seed collection are covered in Hussey and Wallace (1993) and Scheltema (1993).

Seed quality needs to be determined before sowing in order that the correct seeding rate can be determined. Ideally, seed purchased from seed merchants would come fully certified for percentage germination, viability and purity. However, under the Seeds Act 1981, only species named in the Act require certification, and these are mostly agricultural and horticultural plants. For some species, seed testing can be relatively straight forward (Scheltema 1993; Strawbridge and Barrett-Lennard 1993). However, many native species require pre-treatment before they will germinate. Commonly, this may involve periods of cold stratification, scarification, heat treatment or smoke treatment (Loney 1990; Scheltema 1993; Dixon *et al.* 1995; Roche *et al.* 1997; Roche *et al.* 1998). Some native species do not store for more than a few months without losing viability.

Scarification of the seed bed has been shown to increase the germination of a range of eucalypt seedlings (Abbott 1984; Loney 1990; Stoneman *et al.* 1994). Soil cultivation may increase the amount and availability of water stored in the soil, reduce penetration resistance and thereby increase the soil volume explored by the roots and rooting density, and reduce competition from other plants. Deep ripping before seeding breaks up any hard pan layers in the soil and produces an aerated soil that assists root establishment and moisture infiltration. Most clay, loam and duplex soils that have been compacted require ripping 40 to 100 cm (Scheltema 1993).

Figure 6.3 Photopoints are recommended for landholders as a visual record of any changes to vegetation.

Almost all native plant species in undisturbed vegetation form symbiotic associations with mycorrhizal fungi. These associations enhance nutrient uptake of plants and lead to increased growth and improved soil structure and stability (Jasper 1990). Agricultural land to be revegetated and areas of degraded native vegetation will most likely contain sufficient numbers of some mycorrhizal fungi for successful revegetation. However, some species require specific mycorrhizal associations and pre-inoculation with particular fungi may give important benefits (Jasper 1990).

6.2.8 Monitoring

Monitoring changes in vegetation after management activities such as grazing exclusion, direct seeding, weed control and fire provides essential feedback into the review of management guidelines. Trends in condition, recruitment and abundance of species can be established. Where revegetation aims to restore or maintain the hydrological balance, as in the water resource recovery catchments, then depth to water tables etc. need to be monitored. A monitoring program should also include measurements in undisturbed areas of vegetation. This will determine if observed changes in community structure and species abundance are due to treatment effects or wider changes occurring across the landscape (Westman 1991; Lambeck and Saunders 1993; Hobbs and Norton 1996).

Successful outcomes from many of the current recommendations have not been extensively demonstrated. They are based on the best available knowledge and understanding to date and feed back from effective monitoring can only increase this. Results should also be used to identify those areas that are in need of further research, which will in turn lead to the improvement of management prescriptions. In addition, early recognition of failure of any treatments will reduce the risk of further investment into projects with low chances of success. This means that criteria need to be determined to measure whether any given objective has been achieved and the success or failure of particular treatments (Lambeck and Saunders 1993; Hobbs and Norton 1996).

A simple form of monitoring the rehabilitation of remnants is the use of photo points (Figure 6.3). Photographs taken seasonally or annually from the same position, provide a visual record of any changes in an area. However, as discussed in Chapters four and five, photo points give misleading feedback where species cover is improving but not species recruitment. Annual vegetation surveys of permanent transects in a remnant provide a more accurate picture of the success or failure of rehabilitation treatments. Because of the expertise and commitment of resources required for this, it is not a practical option to monitor all remnants in a catchment in this way. It is suggested that a few representative remnants be selected for this more intensive monitoring, while others are monitored with photo points.

6.2.9 Covenanting options

In the United States, United Kingdom, Canada and the Eastern States of Australia, voluntary conservation agreements on areas of remnant vegetation are an important complement to the direct conservation role of the State. These agreements provide a flexible and adaptable approach to remnant management on private land. When combined with Government regulation such as clearing control legislation, this approach balances private property rights with the public interest in nature conservation.

Within Western Australia, three possibilities exist for placing covenants on areas of remnant bush. The Remnant Vegetation Protection Scheme, the National Trust Conservation Covenanting Program and the Department of Conservation and Land Management Voluntary Nature Conservation Covenant. Land for Wildlife is a voluntary scheme administered by CALM, which involves the registration of a remnant for wildlife habitat with no binding covenant placed on the land title.

Remnant Vegetation Protection Scheme (RVPS)

The RVPS has operated since 1988 and up until 1995 had provided fencing for 835 remnants totalling 38 647 hectares (ANZECC 1996). However, as at March 1998, only four remnants (all in the Warren River catchment) totalling 85 ha, had been protected in the Water Resource Recovery Catchments (G. Beeston, Agriculture WA, unpublished data). This probably reflects the view that remnant vegetation is these catchments is protected from clearing by legislation. However, they are not necessarily protected from livestock grazing impacts and other forms of disturbance. The scheme provides a subsidy for 50% of the cost of fencing remnant vegetation with funding allocated on the basis of nature conservation value. In return, landholders enter into a 30 year management agreement to maintain the remnants and this is attached to the land title to retain agreed actions with future landholders. Land subject to this covenant is registered with the Valuer General and is eligible for council rate rebates (Young *et al.* 1996).

The RVPS is administered by Agriculture Western Australia. The Department of Conservation and Land Management assess the remnants for nature conservation value and pass their rankings back to Agriculture WA. Remnants which rank highly are then funded and protected under the scheme.

National Trust Conservation Covenanting Programme

Commencing in 1999, The National Trust of Western Australia (NTWA) has been funded by the Lotteries Commission and NHT to establish a covenanting programme. At the landholders request, the covenant is registered on the land title and requires all future owners to comply with the terms of the covenant. The land holder retains ownership and control over the land subject only to compliance with the covenant. The contents of the covenant are prepared in consultation with the land holder giving due regard to their requirements whilst protecting the conservation values of the area. The NTWA meets the costs of putting the covenant on the title and provides management advice and contacts with other organisations and groups to assist landholders.

The land holder has the responsibility for management of the area under covenant and bears the cost of implementing management requirements. NTWA does advise land owners of grant schemes, such as Save the Bush, which can provide financial assistance for management activities. A representative from the NTWA keeps in touch with the land holder and makes periodic visits to the property by prior arrangement with the owner. Special visits are arranged if any management problems arise or if the property changes hands.

The covenants are individually designed to recognise the requirements of each land holder. As covenants seek to protect bushland and wildlife habitat with high conservation value, typically they stop the destruction of native plants and animals, prohibit livestock grazing and the planting of exotic species. In some areas, the introduction of Australian plants from other areas is also prohibited to protect the balance of indigenous species.

Department of Conservation and Land Management Voluntary Nature Conservation Covenant

Also commencing in 1999, the Department of Conservation and Land Management (CALM) has the ability to enter into covenanting agreements between a landowner and the Executive Officer of CALM or the Minister for the Environment. These agreements are registered on the land title to bind future owners to the agreement. The Covenants are individually designed and supported by practical management plans developed by the landowner and CALM. CALM will continue to provide management advice and may also provide financial assistance for legal costs, fencing and initial management costs, on application. It is also possible to have rates adjusted if the land value varies as a result of the Covenant. Land qualifies if it has high nature conservation values and is not seriously threatened by salinity, mining or Government development proposals.

Land for Wildlife

Land for Wildlife is a voluntary scheme that aims to assist landholders to protect wildlife habitat on their properties or to integrate nature conservation with other land management objectives. Registration of a piece of land does not constitute a covenant being placed on the land and there are no financial incentives offered. The scheme provides support from regional officers who visit individual properties, distribution of advice and materials which contain practical information and field days. Land holders may also receive a sign for their property to indicate their participation in the scheme. No council rate rebates can be claimed as registration does not mean that the land is protected from disturbance from future managers or land owners.

Platt and Ahern (1995) reviewed the Land for Wildlife scheme in Victoria and commented that schemes which do not involve covenants have a high level of acceptance and promoted an ethic of ownership and responsibility. Landholders were not discouraged from participating by a concern that they may lose some degree of control over their land and were involved in a network that provided advice and a nature conservation ethic. However, Gilfedder and Kirkpatrick (1995), from the Australian Nature Conservation Agency, state that voluntary schemes may not provide long term protection of areas if land changes ownership.

Voluntary schemes have low administrative costs and high community acceptability (Young *et al.* 1996). However, they are only effective if they result in conservation of the biodiversity and functioning ecosystem concerned. Financial considerations are the main reasons cited for landholders being reluctant or unable to protect native vegetation on their land. Lack of information about the importance of protecting native vegetation, the potential and demonstrated benefits to landholders and management techniques are also important factors in lack of success in voluntary schemes.

6.3 Recommended management practice for remnant vegetation in water resource recovery catchments

It is appropriate here to re-state the management objectives of the Water and Rivers Commission for remnant vegetation in the water resource recovery catchments. The Commission aims to protect remnant vegetation from further degradation and to promote rehabilitation of previously degraded remnants to optimise water use and conserve biodiversity. The Commission coordinates activities in the various catchments through Recovery Teams. Recovery Teams are now operational for four of the five water resource recovery catchment; the Helena River catchment being the exception.

Management guidelines should recommend practices that can be readily incorporated into routine farm management practices. Recommending complicated procedures will not encourage landholders to view remnant vegetation management as an important part of their farm and catchment planning activities. Some areas of remnant vegetation which have low conservation value ie. they are reduced to two structural layers; tree canopy and exotic grass species, may be strategically important in their position in the landscape for water control. These remnants will require revegetation to ensure their sustainability. In these cases, it may be more appropriate that revegetation activities be undertaken by consultants or others with the relevant expertise required. Revegetation or plantation buffers should be considered for small, strategic remnants or riparian zones.

In either remnants with a relatively intact understorey or those revegetated, the most important management practice is to exclude livestock grazing. Under the remnant vegetation fencing scheme the Recovery Teams are

Table 6.1 Recommended fire intervals for maximising regeneration of common remnant vegetation types in southern Western Australia.

Remnant type	Recommended minimum fire intervals (years)	Comments
Intermediate and high rainfall jarrah-marri	6-8	Maximum fire intervals should be less than 25-30 yrs for jarrah forest species
Low rainfall jarrah-marri	8-12	
Wandoo woodland	No published information available	Note that where there is slow fuel accumulation, fire tolerance is low. Infrequent burning is recommended.
Karri	7-9	CALM have fire intervals of 15-20 years in special conservation areas.
Banksia woodland	10	Certain species require longer intervals between burning (see section 6.2.3 of this report).
Melaleuca	No published information available	Note that low fire tolerance is characteristic of species in moist environments. Infrequent burning is recommended.
Flooded gum	No published information available	As above

currently considering allowing, as acceptable management practice, limited grazing access in times of extreme need, such as drought or for stock shelter where sheep weather warnings may be issued after shearing. The findings from this project show that intact remnants will support this form of limited grazing, however further research is required to determine what stocking rates (Dry Sheep Equivalents) equate to 'limited' grazing. This may vary between vegetation types. Grazing is not recommended in remnants where the understorey is sparse or where remnants have been revegetated.

Weed control is an essential part of remnant vegetation management. Use of grazing and burning for weed control are not recommended. Chemical control is the preferred option in non-riparian remnants. Selective herbicides such as Fusilade® for blanket spraying in remnant bush or spot application of Round-up® are recommended. However, detailed information on suitable herbicides is available from the cited references and services in section 6.2.2 of this report.

Fire is required for the continued regeneration of remnant vegetation communities. Based on the current literature (section 6.2.3), the minimum times between burning are recommended for the most common vegetation communities found in the water resource recovery catchments (Table 6.1).

Where remnants are large enough, it is recommended that only a portion or not more than half of the remnant be burnt at one time. This will provide an additional seed source for regeneration of the burnt area. As well as evolving with fire regimes, vegetation communities have been part of continuous forest or woodlands from which seed sources were available, particularly after hot wild fires where some species may be killed out right. Hot fires are necessary for breaking seed dormancy and for the germination of some hard-seeded species. In special forest conservation areas, CALM carry out hot burns in Summer and Autumn every 15-20 years (Christensen and Annels 1985). In the current situation where remnants are often small and isolated, species may not have the potential to regenerate from hot fires and farm safety is an issue.

Vermin control is required to effectively manage areas of remnant vegetation. Methods used should be the most appropriate for the species involved as well as for each individual landholder.

Photopoints are recommended for landholders as an effective visual record of any changes to vegetation. It is recommended that more intense monitoring involving vegetation surveys and site characterisation is carried out in the demonstration sub-catchments which are already being intensively studied under the Salinity Action Plan. Information on ground water levels etc. in areas surrounding and possibly within remnants can then be readily obtained.

Entering into the remnant vegetation fencing scheme which exits under the Western Australian Salinity Action Plan, does not constitute a conservation covenant on the land. Young *et al.* (1996) reviewed similar schemes elsewhere in Australia and found that they have high community acceptability. The main impediments to continued protection and management of remnants registered in such schemes were lack of information and advice on management issues, poorly demonstrated benefits of conserving remnant vegetation and lack of assistance with the costs of management activities such as fencing.

Field days and other useful sources of management information need to be provided to landholders to address any lack of knowledge which may be restricting participation in the fencing scheme. Effective monitoring techniques must be adopted in order to demonstrate the outcomes of remnant vegetation protection and regeneration on water quality in these catchments. The fencing scheme under the Salinity Action Plan provides assistance with the cost of fencing remnants. Information should also be provided on additional funding sources available for revegetation projects.

7. References

Abbott, I. (1984). Emergence, early survival, and growth of seedlings of six tree species in mediterranean forest of Western Australia. *Forest Ecology and Management* 9: 51–66.

Abensperg-Traun, M., Atkins, L., Hobbs, R. and Steven, D. (1996). Weeds a curse for native plants in farm woodlands. *Western Australian Journal of Agriculture* 37:124–129.

Adams, S., Strain, B.R. and Adams, M.S. (1970). Water-repellent soils, fire and annual plant cover in a desert scrub community of south eastern California. *Ecology* 51: 696–700.

Adamson, D.A. and Fox, M.D. (1982). Change in Australasian vegetation since European settlement. In *A History of Australasian Vegetation*. Smith, J.M.B (ed.) pp 109–146. McGraw Hill: Sydney.

ANZECC Working Group on Nature Conservation on Private Land. (1996). *Nature Conservation on Private Land: Commonwealth, State and Territory Legislation and Programmes.* A report of the Working Group on Nature Conservation on Private Land prepared for the Australian and New Zealand Environment and Conservation Council Standing Committee on Conservation, Australian Nature Conservation Agency, Canberra.

Ashton, D.H. (1976). The development of even-aged stands of *Eucalyptus regnans* F. Muell. In Central Victoria. *Australian Journal of Botany* 24: 397–414.

Baird, A.M. (1977). Regeneration after fire in Kings Park, Western Australia. *Journal of the Royal Society of Western Australia* 60: 1–22.

Beard, J.S. (1981). *Vegetation Survey of Western Australia: The Vegetation of the Swan Area.* University of Western Australia Press, Perth, Western Australia.

Beard, J.S. (1990). *Plant Life of Western Australia.* Kangaroo Press, Kenthurst, NSW.

Beard, J.S. and Sprenger, B.S. (1984). Geographical data from the vegetation survey of Western Australia. *Vegetation Survey of Western Australia Occasional Paper* 2. Vegmap Publications: Perth.

Bell, D.T. (1985). Aspects of response tofire in the Northern Sandplain heathlands. In *Fire Ecology and Management of Western Australian Ecosystems.* J. Ford (ed.). pp. 33–40. Western Australian Institute of Technology Environmental Studies Group Report 14. Bentley, Western Australia.

Bell, D.T. McCaw, W.L. and Burrows, N.D. (1989). Influence of fire on jarrah forest vegetation. In *The Jarrah Forest. A Complex Mediterranean Ecosystem.* Dell, B., Havel, J.J. and Malajczuk, N (Eds.). pp 203–215. Kluwer: Dordrecht.

Bell, D.T. and Froend, R.H. (1990). Mortality and growth of tree species under stress at Lake Toolibin in the Western Australian wheatbelt. *Journal of the Royal Society of Western Australia* 72:63–66.

Bell, D.T. and Heddle, E.M. (1989). Floristic, morphologic and vegetational diversity. In *The Jarrah Forest. A Complex Mediterranean Ecosystem.* Dell, B., Havel, J.J. and Malajczuk, N (Eds.). pp 53–66. Kluwer: Dordrecht.

Bell, D.T. and Koch, J.M. (1980). Post fire succession in the northern jarrah forest of Western Australia. *Australian Journal of Ecology* 5: 9–14.

Belsky, A.J. (1992). Effects of grazing, competition, disturbance and fire on species competition and diversity in grassland communities. *Journal of Vegetation Science.* 3:187–200.

Bettenay, E. (1984). Origin and nature of the sandplains. In *Kwongan: Plant Life of the Sand Plain.* J.S. Pate and J.S. Beard (eds.) pp. 57–68. University of Western Australia, Perth, Western Australia.

Bettenay, E. and Hingston, F.J. (1961). The soils and land use of the Merriden Area, Western Australia. CSIRO Australia Division of Soils Land Use Series 41.

Blackall, W.E. and Grieve, B.J. (1980) *How to Know Western Austalian Wildflowers Part 3A.* University of Western Australia, Perth.

Blackall, W.E. and Grieve, B.J. (1981a) *How to Kknow Western Austalian Wildflowers Part 3B.* University of Western Australia, Perth.

Blackall, W.E. and Grieve, B.J. (1981b) *How to Know Western Austalian Wildflowers Parts 1 & 2*. University of Western Australia, Perth.

Blackall, W.E. and Grieve, B.J. (1982) *How to Know Western Austalian Wildflowers Part 4, 2nd Ed.* University of Western Australia, Perth.

Blackall, W.E. and Grieve, B.J. (1988) *How to Know Western Austalian Wildflowers Part 1 2nd Ed.* University of Western Australia, Perth.

Bond, R.D. (1969). The occurrence of water-repellent soils in Australia. In *Water Repellent Soils.* L.F. De Bano and J Letey (eds.). pp. 1–7. Proceedings of the Symposium on Water Repellent Soils. University of California, Riverside, USA.

Bowen, B.J. and Pate, J.S. (1993). The significance of root starch in post fire shoot recovery of the resprouter *Stirlingia latifolia* R.Br. (Proteaceae). *Annals of Botany (London)* 72:7–16.

Bowen, G.D. (1973). Mineral nutrient relations of ectomycorrhizae. In *Ectomycorrhizae–Their Ecology and Physiology.* (Eds. Marks, G.C. and Kozlowski, T.T.). pp 151–205. Academic Press, New York.

Bowen, G.D. (1983). Microbial determinants of plant nutrient uptake. In *Soils: an Australian Viewpoint.* Division of Soils, CSIRO. Pp. 693–710. CSIRO, Melbourne/Academic Press, London.

Bridgewater, P.B. and Backshall, D.J. (1981). Dynamics of some Western Australian ligneous formations with special reference to the invasion of exotic species. *Vegetatio* 46: 141–148.

Brown, N.A.C. (1993). Promotion of germination of fynbos seeds by plant-derived smoke. *New Phytologist* 123: 575–583.

Burrows, N., Gardiner, G., Ward, B. and Robinson, A. (1990). Regeneration of *Eucalyptus wandoo* following fire. *Australian Forestry* 53: 248–258.

Burrows, N., Ward, B. and Robinson, A. (1995) Regeneration and flowering responsed of plant species in jarrah forest plant communities. In *Burning Our Bushland. Proceedings of a Conference About Fire and Urban Bushland.* Harris, J. (ed.) pp. 30–33. Urban Bushland Council (WA), Perth, Western Australia.

Burrows, W.H., Carter, J.O., Ward, B. and Robinson, A. (1990).Regeneration of Eucalyptus wandoo following fire. *Australian Forestry* 53:248–258.

Cale, P. and Hobbs, R.J. (1991). Condition of roadside vegetation in relationto nutrient status. In D.A. Saunders and R.J. Hobbs (eds.). *Nature Conservation 2: The Role of Corridors.* Surrey Beatty and Sons, Chipping Norton, NSW. pp 353–362.

Christensen, P. and Annels, A. (1985). Fire in Southern Tall Forests. In *Fire Ecology and Mangement in Western Australian Ecosystems.* Ford, J. (ed.). pp. 67–82. Western Australian Institute of Technology Environmental Studies Group Report 14, Perth.

Christensen, P. and Kimber, P. (1975). Effect of prescribed burning on the flora and fauna of south-west Australian forests. *Proceedings of the Ecological Societiy of Australia* 9: 85–107.

Christensen, P., Recher, H. and Hoare, J. (1981). Responses of open forests (dry sclerophyll forests) to fire regimes. In *Fire and the Australian Biota.* Gill, A.M., Groves, R.H. and Noble, I.R. (eds.) pp. 243–272. Australian Academy of Science, Canberra.

Cowling, R. and Lamont, B.B. (1987). Post-fire recruitment of four co-occurring *Banksia* species. *Journal of Applied Ecology* 10:345– 500.

Cowling, R.M., Lamont, B.B. and Enright, N.J. (1990). Fire and management of south-western Australian banksias. *Proceedings of the Ecological Society of Australia* 16: 177–183.

Cowling, R.M., Lamont, B.B. and Pierce, S.M. (1987). Seed bank dynamics of four co-occurring *Banksia* species. *Journal of Ecology* 75: 289–302.

D'Antonio, C.M. and Vitousek, P.M. (1992). Biological invasions by exotic grasses, the grass/fire cycle and global change. *Annual Review of Ecology and Systematics* 23: 63–87.

Davies, J.R. and Bari, M.A. (1995). *Streamflow and Salinity Review of the Warren River Basin, Western Australia.* Water Authority of Western Australia Report WS150, WAWA, Perth, WA.

De Bano, L.F. and Rice, R.M. (1973). Water-repellent soils: their implications in forestry. *Journal of Forestry* 71: 220–223.

De Bano, L.F., Mann, L.D. and Hamilton, D.A. (1970). Translocation of hydrophobic substances into soil by burning organic litter. *Soil Science Society of America Proceedings* 34: 130–133.

De Bano, L.F. and Rice, R.M. (1973). Water-repellent soils: their implications in forestry. Journal of Forestry 71:220–223.

Dixon, B. and Keighery, G. (1995a) Suggested methods to control weeds. In *Managing Perth's Bushlands*. Sheltema, M and Harris, J. (Eds.). Greening Western Australia, Perth.

Dixon, B. and Keighery, G. (1995b) Recommended methods to control specific weed species. In *Managing Perth's Bushlands*. Sheltema, M and Harris, J. (Eds.). Greening Western Australia, Perth.

Dixon, K.W., Roche, S. and Pate, J.S. (1995). The promotive effect of smoke derived from burnt native vegetation on seed germination of Western Australian plants. *Oecologia* 101: 185–192.

Dixon, R.N.M., Mauger, G.W. and Rogers, A. (1998). Trees on farms to reduce salinity in the clearing control catchments. *Water Resources Technical Series* WRT20 Water and Rivers Commission, Perth, Western Australia.

Dodd, J., Martin, R. and Howes, K.M. (1993). *Management of Agricultural Weeds in Western Australia.* Bulletin 4243, Western Australian Department of Agriculture, Perth.

Edminston, R. (1985). Direct seeding in the wheatbelt areas of Western Australia. In *Direct Seeding and Natural Regeneration Techniques.* Revegetation Workshop Proceedings. Venning, J. (ed.). pp. 36–40. Greening Australia, Canberra.

Floyd, A.G. (1976). Effect of burning of regeneration from seeds in wet sclerophyll forest. *Australian Forestry* 39: 210–220.

Friend, J.A. (1990). The numbat Myrmecobius fasciatus (Myrmecobiidae): history of decline and potential for recovery. *Proceedings of the Ecological Society of Australia* 16:369–377.

Froend, R.H. and McComb, A.J. (1991). An account of the decline of Lake Towerrinning, a wheatbelt wetland. *Journal of the Royal Society of Western Australia* 73:123–128.

Froend, R.H., Heddle, E.M., Bell, D.T. and McComb, A.J. (1987). Effects of salinity and waterlogging on the vegetation of Lake Toolibin, Western Australia. *Australian Journal of Ecology* 12:281–298.

Gardener, C.A. (1957). The fire factor in relation to the vegetation of Western Australia. *Western Australian Naturalist* 5: 166–173.

George, R.J. (1992). Estimating and modifying the effects of agricultural development on the groundwater balance of large wheatbelt catchments. Applied Hydrogeology 1:41–54.

George, R.J. and McFarlane, D.J. (1993). The effect of changing hydrological processes on remnant vegetation and wetlands. In K.J. Wallace (ed.). *Remnant Native Vegetation Ten Years On.* Proceedings of the Dryandra Workshop. Department of Conservation and Land Management, Perth, WA.

George, R.J., McFarlane, D.J. and Speed, R.J. (1995) The consequences of a changing hydrologic environment for native vegetation in southwestern Australia. In *Nature Conservation 4: The Role of Networks.* D.A. Saunders, J.L. Craig and E.M. Mattiske (eds.). Surrey Beatty and Sons, Chipping Norton, NSW.

George, R.J., McFarlane, D.J. and Speed, R.J. (1996). Degradation of remnant vegetation. *Western Australian Journal of Agriculture* 37:3–9.

Gilbert, J.M. (1959). Forest succession in the Florentine Valley, Tasmania. *Papers of the Proceedings of the Royal Society of Tasmania.* 93: 129–151.

Gilfedder, L. and Kirkpatrick, J.B. (1995). *A survey of landholder attitudes and intentions towards the long term conservation of native lowland grasslands.* A Report to the Grasslands Ecology Unit, Australian Nature Conservation Agency.

Gill, A.M. (1975). Fire and the Australian flora: a review. *Australian Forestry* 38: 4–25.

Gill, A.M. (1981). Adaptive responses of Australian vascular plant species to fires. In *Fire and the Australian Biota.* Gill, A.M., Groves, R.H. and Noble, I.R. (eds.) pp. 243–272. Australian Academy of Science, Canberra.

Greening Australia (1990). *Sowing the Seeds. Direct Seeding and Natural Regeneration Conference Proceedings.* Greening Australia, Canberra.

Grewar, G. (1986). Direct sowing of trees on the south coast of Western Australia. In *Proceedings of Tree Establishment Workshop.* Greening Australia, Esperence, Western Australia.

Griffin, E.A. and Hopkins, A.J.M. (1991). Fire and management of road verge vegetation in the Western Australian wheatbelt. Western Australian Department of Conservation and Land Management Technical Report, Perth, WA.

Hacker, R.B. (1984). Vegetation dynamics in a grazed mulga community. 1. The mid-storey shrubs. *Australian Journal of Botany* 32: 239–249.

Harris, L.D. (1988). Edge effects and conservation of biotic diversity. *Conservation Biology* 2:330–332.

Hatch, A.B. (1960). Ash bed effects in Western Australian forest soils. *Forest Department of Western Australia Bulletin* 86, Perth, Western Australia.

Hawkins, B. J. (1997). Kent river flows and salt loads. In *National Dryland Salinity Programme Conference.* Mt Barker, Western Australia, October 13–15. Department of Conservation and Land Management, Albany, Western Australia.

Heddle, E.M. and Specht, R.L. (1975). Dark Island Heath (Ninety Mile Plain, South Australia) VIII. The effect of fertilisers on composition and growth, 1950–1972. *Australian Journal of Botany* 23:151–164.

Hester, A.J. and Hobbs, R.J. (1992). Influence of fire and soil nutrients on native and non-native annuals at remnant vegetation edges in the Western Australian wheatbelt. *Journal of vegetation Science* 3:101–108.

Hobbs, R.J. (1989). The nature and effects of disturbance relative to invasions. In J. Drake, F. di Castri, R. Groves, F. Kruger, H. Mooney, M. Rejmanek and M. Williamson (eds.). *Biological Invasions: A Global Perspective.* John Wiley and Sons, Chichester. pp. 389–405.

Hobbs, R.J. (1991a). Re-intergrating fragmented landscapes — a preliminary framework for the Western Australia wheatbelt. Journal of Environmental Management 33: 161–167.

Hobbs, R.J. (1991b). Disturbance a precursor to weed invasion in native vegetation. *Plant Protection Quarterly* 6(3):99–104.

Hobbs, R.J. (1993a). Effects of landscape fragmentation on ecosystem processes in the Western Australian wheatbelt. *Biological Conservation* 64:193–201.

Hobbs, R.J. (1993b). Can revegetation assist in the conservation of biodiversity in agricultural areas? *Pacific Conservation Biology* 1: 29–38.

Hobbs, R.J. and Atkins, L. (1988). Spatial variability of experimental fires in the south-west of Western Australia. *Australian Journal of Ecology* 13:295–299.

Hobbs, R.J. and Atkins, L. (1990) Fire-related dynamics of a Banksia woodland in south-western Western Australia. *Australian Journal of Botany* 38:97–110.

Hobbs, R.J. and Atkins, L. (1991). Interactions between annual and perennial vegetation components in a Western Australian wheatbelt reserve. *Journal of Vegetation Science* 2:643–654.

Hobbs, R.J. and Norton, D.A. (1996). Towards a conceptual framework for restoration ecology. *Restoration Ecology* 4: 93–110.

Hopkins, A.J.M. and Griffin, E.A. (1989). Fire in *Banksia* woodlands of the Swan Coastal Plain. *Journal of the Royal Society of Western Australia* 71:93–94.

Hopper, S.D. and Burbidge, A.H. (1989). Conservation status of *Banksia* woodlands on the Swan Coastal Plain. *Journal of the Royal Society of Western Australia* 71: 115–116.

Hubble, G.D., Isbell, R.F. and Northcote, K.H. (1983). Features of Australian soils. In *Soils: an Australian Viewpoint.* Division of Soils, CSIRO. Pp. 693–710. CSIRO, Melbourne/Academic Press, London.

Hunt, N. and Gilkes, R. (1992). *Farm Monitoring Handbook.* University of Western Australia, Perth, Western Australia.

Hussey, B.M.J. and Wallace, K.J. (1993). *Managing Your Bushland.* Department of Conservation and Land Management, Perth, Western Australia.

Hussey, B.M.J., Keighery, G.K., Cousens, R.D., Dodd, J. and Lloyd, S.G. (1997). *Western Weeds. A Guide to the Weeds of Western Australia.* Plant Protection Society of Western Australia, Perth, Western Australia.

Intergraph Corporation (1991). Microstation PC. 4.0. Huntsville, U.S.A.

Janzen, D.H. (1983). No park is an island: increase in interference from outside as park size decreases. *Oikos* 41:402–410.

Jasper, D. (1990). The role of vesicular-arbuscular (VA) mycorrhizal fungi in regeneration. In *Sowing the Seeds. Direct Seeding and Natural Regeneration Conference.* pp. 95–98. Greening Australia, Canberra.

Kelly, S. (1995). *Land Managment Report for the Kent River Catchment.* Agriculture Western Australia, Perth, WA.

Kent, M. and Coker, P. (1996). *Vegetation Description and Analysis. A Practical Approach.* John Wiley and Sons, Chichester, UK.

King, P.M. (1981) Comparison of methods for measuring severity of water repellence of sandy soils and assessment of some factors that affect its measurement. *Australian Journal of Soil Research* 19:275–285.

Kinnear, J.E., Onus, M.L. and Bromilow, R.N. (1988). Fox control and rock wallaby population dynamics. *Australian Wildlife Research* 15:435–450.

Lambeck, R.J. and Saunders, D.A. (1993). The role of patchiness inreconstructed wheatbelt landscapes. In *Nature Conservation 3. Reconstruction of Fragmented Ecosystems, Global and Regional Perspectives.* D.A. Saunders, R.J. Hobbs and P.R. Erlich (eds.). pp 153–161. Surrey Beatty, Chipping Norton.

Lamont, B.B. and Barker, M.J. (1988). Seed bank dynamics of a serotinous, fire-sensitive *Banksia* species. *Australian Journal of Botany* 36: 193–203.

Lamont, B.B. and van Leeuwen, S.J. (1988). Seed production and mortality in a rare *Banksia* species. *Journal of Applied Ecology* 25: 551–559.

Landsberg, J. Morse, J. and Khanna, P. (1990). Tree dieback and insect dynamics in remnants of native woodlands on farms. *Proceedings of the Ecological Society of Australia* 16: 149–165.

Lefroy, E.C., Hobbs, R.J. and Atkins, L. (1991). *Revegetation Guide to the Central Wheatbelt.* Western Australian Department of Agriculture, CSIRO and Greening Australia, Perth, Western Australia.

Loney, B. (1990). Direct seeding in Western Australia — Its status and future. In *Sowing the Seeds. Direct seeding and Natural Regeneration Conference Proceedings.* Greening Australia, Deakin, Australian Capital Teritory. pp 103–112.

Loveday, J. (1974) Methods of analysis of irrigated soils. *Technical Communication No.54*, Commonwealth Agricultural Bureaux, U.K.

McDonald, R.C., Isbell, R.F., Speight, J.G., Walker, J and Hopkins, M.S. (1990) *Australian Soil and Land Survey Field Handbook 2nd Ed.* Inkata, Melbourne.

MacLeod, N.D., Brown, J.R. and Noble, J.C. (1993). Ecological and economic considerations for the management of shrub encroachment in Australian rangelands. *10th Australian and 14th Asian-Pacific Weeds Conference.* Weed Society of Queensland, Brisbane, QLD.

Main, A.R. (1987). Management of remnants of native vegetation — a review of the problems and the development of an approach with reference to the wheatbelt of Western Australia. In *Nature Conservation: the role of remnants of native vegetation.* D.A. Saunders, G.W. Arnold, A.A. Burbidge and A.J.M. Hopkins (eds.) pp. 195–200. Surrey Beatty and Sons, Chipping Norton, Australia.

Main, A.R. (1993). Landscape reintegration : Problem definition. In R.J. Hobbs and D.A. Saunders (eds.). *Reintegrating Fragmented Landscapes: Towards Sustainable Production and Nature Conservation.* pp 189–208. Springer-Verlag, New York.

Majer, J.D. (1980). The influence of ants on broadcast and naturally spread seeds in rehabilitated bauxite mined areas. *Reclamation Review* 3: 3–9.

Majer, J.D. (1990). The greening of Australia: taking the animals into account. In *Sowing the Seeds. Direct Seeding and Natural Regeneration Conference.* pp. 17–25. Greening Australia, Canberra.

Marchant, N.G., Wheeler, J.R., Rye, B.L., Bennett, E.M., Lander, N.S., Macfarlane, T.D. (1987a) *Flora of the Perth Region Part 1.* Western Australian Herbarium, Perth.

Marchant, N.G., Wheeler, J.R., Rye, B.L., Bennett, E.M., Lander, N.S., Macfarlane, T.D. (1987b) *Flora of the Perth Region Part 2.* Western Australian Herbarium, Perth.

McArthur, A.G. and Cheney, N.P. (1966). The characterisation of fires in relation to ecological studies. *Australian Forest Research* 2:36–45.

McCaw, W.L. (1988). Regeneration of *Acacia* and *Kennedia* from soil stored seed following an autumn fire in jarrah (*Eucalyptus marginata*) forest. *Journal of the Royal Society of Western Australia* 71:1–6.

Mercer, J. (1994). Fire regeneration of Eucalyptus wandoo in farmland remnants. An unpublished report for the Save The Bush Programme and the South Stirlings Land Conservation District.

Merriam, G. and Saunders, D.A. (1993). Corridors in restoration of fragmented landscapes. In *Nature Conservation 3. Reconstruction of Fragmented Ecosystems, Global and Regional Perspectives.* D.A. Saunders, R.J. Hobbs and P.R. Erlich (eds.). pp 163–78. Surrey Beatty, Chipping Norton.

Milberg, P. and Lamont, B.B. (1995). Fire enhances weed invasion of roadside vegetation in southwestern Australia. *Biological Conservation* 73: 45–49.

Milchunas, D.G. and Lauenroth, W.K. (1993). Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs* 63: 327–366.

Monk, D., Pate, J.S. and Loneragan, W.A. (1981). Biology of *Acacia pulchella* R.Br. with special reference to symbiotic nitrogen fixation. *Australian Journal of Botany* 29: 579– 592.

Moran, G.F. (1992). Patterns of genetic diversity in Australian tree species. *New Forests* 6: 49–66.

Mott, J.J. and McKeon, G.M. (1977). A note on the selection of seed types by harvester ants in northern Australia. *Australian Journal of Ecology* 2: 231–235.

Moulds, B.D. and Bari, M.A. (1995a). *Denmark River Streamflow and Salinity Review.* Water Authority of Western Australia Report WS152, WAWA, Perth, WA.

Moulds, B.D. and Bari, M.A. (1995b). *Wellington Reservoir Catchment Streamflow and Salinity Review.* Water Authority of Western Australia Report WS151, WAWA, Perth, WA.

Mueller-Dombois, D. and Ellenberg, H. (1974). *Aims and Methods of Vegetation Ecology*. John Wiley, New York.

Panetta, F.D. and Hopkins, A.J.M. (1991). Weeds in corridors: Invasion and management. In D.A. Saunders and R.J. Hobbs (eds.). *Nature Conservation 2: The Role of Corridors*. Surrey Beatty and Sons, Chipping Norton, NSW. pp 341–351.

Pate, J.S., Froend, R.H., Bowen, B.J., Hansen, A. and Kuo, J. (1990). Seedling growth and sotrage characteristics of seeder and resprouter species of mediterranean-type ecosystems of S.W. Australia. *Annals of Botany* 65: 585–601.

Peet, G.B. (1971). A study of scrub fuels in the harrah forest of Western Australia. *Forest Department of Western Australia Bulletin* 80.

Pettit, N.E. and Froend, R.H. (1994). *Livestock grazing on native vegetation in the Wellington Reservoir catchment.* WS 143 Water Authority of Western Australia.

Pettit, N.E., Froend, R.H. and Ladd, P.G. (1995). Grazing in remnant woodland vegetation: changes in species composition and life form groups. *Journal of Vegetation Science* 6:121–130.

Pettit, N.E., Ladd, P.G. and Froend, R.H. (1998). Passive clearing of native vegetation: livestock damage to remnant jarrah (*Eucalyptus marginata*) woodlands in Western Australia. Journal of the Royal Society of Western Australia 81:95–106.

Pigott, J.P., Brown, P.H. and Williams, M.R. (1994). Direct seeding trees on farmland in the Western Australian wheatbelt. *CALMScience* 1: 131–158.

Platt, S. and Ahern, L. (1995)Voluntary nature conservation on private land in Victoria– evaluating the Land for Wildlife programme. In Bennett, A., Backhouse, G. and Clark, T. (eds.). *People and Nature Conservation, Perspectives on Private Land Use and Endangered Species Recovery.* Transactions of the Royal Zoological Society of New South Wales.

Portlock, C.C., Shea, S.R., Majer, J.D. and Bell, D.T. (1990). Stimulation of germination of *Acacia pulchella*: Laboratory basis for forest management options. *Journal of Applied Ecology* 27:319–324.

Prober, S.M.and Brown, A.H.D. (1994). Conservation of grassy white box woodlands: population genetics and fragmentation of *Eucalyptus albens. Conservation Biology* 8:1003–1013.

Prober, S.M. and Thiele, K.R. (1995). Conservation of the grassy white box woodlands: relative contributions of size and disturbance to floristic composition and diversity of remnants. *Australian Journal of Botany* 43:349–366.

Rayment G.E. and Higginson F.R. (1992) *Australian Laboratory Handbook of Soil and Water Chemical Methods.* Inkata, Melbourne.

Recher, H.F. (1993). The loss of biodiversity and landscape restoration: conservation, management, survival. An Australian perspective. In D.A. Saunders, R.J. Hobbs and P.R. Erlich (eds.). *Nature Conservation 3. Reconstruction of Fragmented Ecosystems, Global and Regional Perspectives.* pp 141–151. Surrey Beatty, Chipping Norton.

Roche, S., Dixon, K.W. and Pate, J.S. (1998). Fore everything a season: smoke induced seed germination and seedling recruitment in a Western Australian *Banksia* woodland. *Australian Journal of Ecology* 23: 111–120.

Roche, S., Koch, J.M. and Dixon, K.W. (1997). Smoke enhanced seed germination for mine rehabilitation in the south west of Western Australia. *Restoration Ecology* 5: 191–203.

Ruprecht, J.K., Stokes, R.A. and Pickett, R.B. (1985). *Denmark River Yield and Salinity Study.* Water Authority of Western Australia Report WH8, WAWA, Perth, WA.

Saunders, D.A. and Hobbs, R.J. (1989). Corridors for conservation. *New Scientist 28 January*: 63–68.

Saunders, D.A., Hobbs, R.J. and Margules, C.R. (1991). Biological consequences of ecosystem fragmentation: a review. *Conservation Biology* 5:18–32.

Scheltema, M.A. (1993). *Direct Seeding of Trees and Shrubs.* Greening Western Australia, Perth.

Scott, D.F. (1997). The contrasting effects of wildfire and clearfelling on the hydrology of a small catchment. *Hydrological Processes* 11: 543–555.

Scott, D.F. and Van Wyk, D.B. (1990). The effects of wildfire on soil wettability and hydrological behaviour of an afforested catchment. *Journal of Hydrology* 121:239–256.

Scougall, A. and Majer, J. (1991). The impact of fencing on soil, flora and soil fauna in remnants of open Jam–York Gum woodlands in the Western Australian wheatbelt. In *Soil Science and the Environment.* Proceedings of a Conference, Albany, Western Australia. Australian Society of Soil Science Inc (WA):Perth, Westrn Australia.

Scougall, S.A. Hobbs, R.J. and Majer, J.D. (1993). Edge effects in grazed and ungrazed Western Australian wheatbelt remnants in relation to ecosystem reconstruction. In D.A. Saunders, R.J. Hobbs and P.R. Erlich (eds.). *Nature Conservation 3. Reconstruction of Fragmented Ecosystems, Global and Regional Perspectives.* pp 163–78. Surrey Beatty, Chipping Norton

Shea, S.R., McCormick, J. and Portlock, C.C. (1979). The effect of fires on regeneration of leguminous species in the northern jarrah (*Eucalyptus marginata* Sm.) forest of Western Australia. *Australian Journal of Ecology* 4: 195–205.

Shearer, B.*L.* and Tippett, J.T. (1989). *Jarrah dieback: the dynamics and management of Phytophthora* cinnamomi in the Jarrah (*E. marginata*) forest of south western Western Australia. *Research Bulletin* 3. Department of Conservation and Land Management, Perth, Western Australia.

Sokal, R.R. and Michener, C.D. (1958) A statistical method for evaluating systematic relationships. *University of Kansas Science Bulletin* 38: 1409–1438.

Soule, M.E. (1987). Introduction. In M. E. Soule (ed.). *Viable Populations for Conservation.* pp 1–10. Cambridge University Press, Cambridge, UK.

Specht, R.L. (1963). Dark Island heath (Ninety Mile Plain, South Australia). VII. The effect of fertilisers on composition and growth, 1950–1960. *Australian Journal of Botany* 11:67–94.

Statsoft (1984– 1996). *STATISTICA for Windows.* StatSoft Inc., Tulsa, USA.

Stoneman, G.L. and Dell, B. (1994). Emergence of *Eucalyptus marginata* (jarrah) from seed in Mediterranean-climate forest in response to overstorey, site, seedbed and ant harvesting. *Australian Journal of Ecology* 19:96–102.

Stoneman, G.L., Dell, B. and Turner, N.C. (1994). Mortality of Eucalyptus marginata (jarrah) seedlings in Mediterranean-climate forest in response to overstorey, site, seedbed, fertiliser application and grazing. *Australian Journal of Ecology* 18:103–109.

Strawbridge, M. & Barrett-lennard, E.G. (1993) Growing Saltbush seedlings and cuttings. *WA Department of Agriculture Farmnote* 7/93.

True, D.E., Kikiros, G. and Froend, R.H. (1992). Preliminary assessment of the effects of grazing on the remnant vegetation in the Kent River water reserve. Water Authority of Western Australia. WS 106. August 1992 14pp.

Vlahos, S. and Bell, D.T. (1986). Soil seed-bank components of the northern jarrah forest of Western Australia. *Australian Journal of Ecology* 11:171–179.

Ward, S.C., Koch, J.M. and Grant, C.D. (1997). Ecological aspects of soil seed-banks in relation to bauxite mining. 1. Unmined jarrah forest. *Australian Journal of Ecology* 22: 169–176.

Watson, J.R. (1991). The identification of river foreshore corridors for nature conservation in the south coast region of Western Australia. In D.A. Saunders and R.J. Hobbs (eds.). *Nature Conservation 2: The Role of Corridors*. pp 63–68. Surrey Beatty and Sons, Chipping Norton, NSW.

Western Australian Salinity Action Plan (1996). Government of Western Australia.

Western Australian Salinity Action Plan (1998). Draft Update. Government of Western Australia.

Westman, W.E. (1991). Ecological restoration projects: measuring their performance. *Environmental Professional* 13: 207–215.

Williams, O.B. (1969). Studies in the ecology of the riverine plain. V. Plant density response of species in a *Danthonia caespitosa* grassland to 16 years grazing by merino sheep. *Australian Journal of Botany* 16:565–578.

Williamson, M. (1975). The design of nature preserves. *Nature (London)*. 256:519.

Wycherley, P. (1984). People, fire and weeds: can the vicious spiral be broken? In *The Management of Small Bushland Areas in the Perth Metropolitan Region*. S.A. Moore (ed.). pp. 11–17. Western Australian Department of Fisheries and Wildlife, Perth, Western Australia.

Yahner, R.H. (1988). Changes in wildlife communities near edges. *Conservation Biology* 2:33–339.

Yates, C.J. and Hobbs, R.J. (1997). Temperate eucalypt woodlands: a review of their status, processes threatening their persistence and techniques for restoration. *Australian Journal of Botany* 45:949–973.

Yates, C.J., Hobbs, R.J. and Bell, R.W. (1994). Landscape-scale disturbances and regeneration in semi-arid woodlands of south-western Australia. *Pacific Conservation Biology* 1: 214–221.

Yates, C.J., Taplin, R., Hobbs, R.J. and Bell, R.W. (1995). Factors limiting the recruitment of *Eucalyptus salmonophloia* in remnant woodlands. 2. Post-dispersal seed predation and soil seed reserves. *Australian Journal of Botany*. 43: 145–155.

Young, , M.D., Gunningham, N., Elix, J., Lambert, J., Howard, B., Grabosky, P. and McCrone, E. (1996). Reimbursing the future; an evaluation of motivational, voluntary, price-based, property-right and regulatory incentives for the conservation of biodiversity. *Biodiversity Series No. 9.* Biodiversity Unit, Department of the Environment, Sport and Territories, Canberra.

Zar, J.H. (1984). *Biostatistical Analysis*. Prentice-Hall, New Jersey.