1 INTRODUCTION

Weeds affect agricultural, natural, and urban ecosystems (Lawes & Wallace, 2008; Eiswerth & van Kooten, 2002; Parsons & Cuthbertson, 2001) and are a recognised global economic problem. Pimentel *et al*. (2000) estimated that exotic plants cost the United States over US$34 billion p.a. For American ranchers, weeds are estimated to cost US$6 billion annually due to control costs, losses to pasture production and damages to capital. Many of the weeds that exist in Australia were deliberately imported for food, fodder, fibre or ornamental purposes (Scott, 2000; Panetta, 1993; Parsons, 1981). Australian farming systems operate on extensive areas with relatively low yields per unit area, compared to other developed countries (Vere *et al.*, 1997). Weeds are an important problem within these farming systems and have an estimated annual cost of $3.4 and $4.4 billion to the Australian agricultural industry (Sinden *et al.*, 2005; Sinden *et al.*, 2004). The costs of weeds to rangeland graziers have yet to be determined but weeds pose a significant threat to this 12.3 billion dollar industry p.a. This is over one-third of Australia's total agricultural production (ABS, 2008; Grice, 2006).

Woody weed encroachment is a common phenomenon within rangeland systems. Rangelands are important systems that represent over 70 % of Australia and 20 % of the world's land mass (Grice, 2006; Sankaran *et al.*, 2005; Donohue *et al.*, 2005; Foran *et al.*, 1990). Graziers are particularly affected by woody weeds as they are the primary users of these landscapes (Range.Aust, 2007; Scanlan *et al.*, 1991; Harrington *et al.*, 1984b). Graziers are struggling to deal with woody weeds globally in temperate and tropical rangelands (Wilson *et al.*, 2001; Brown & Archer, 1999). In Australia, woody weed infestations can be found on 68 % of northern Australia and 96 % of northern Queensland grazing properties (Bortolussi *et al.*, 2005a). Despite the magnitude of these rangeland systems and the importance of woody weed encroachment, no research has attempted to derive an economically optimal integrated weed management (IWM) strategy to help weed managers (graziers) in these systems. Woody weeds are continuing to expand due to factors such as the accidental or deliberate introduction of alien plants, climate change, high stocking rates, changes in livestock types,

suppression of fires and disturbances within these systems (Grice, 2004; Kriticos *et al.*, 2003b; Noble *et al.*, 1997; Harrington *et al.*, 1984a).

Woody weeds decrease grazing production through decreased land carrying capacity and land condition, along with increased mustering effort, impediment of cattle accessing waterways, and loss of livestock (Parsons & Cuthbertson, 2001; Grice, 2000; Mackey, 1997). Woody weed infestations are particularly problematic to graziers as the productive capacity of rangelands is inherently low, with grazing gross margins being < \$14 ha^{-1} per annum and the cost of a single weed-control event often being many times greater (N.D. Macleod 208, pers. comm. Ash & Stafford Smith, 2003; Brown *et al.*, 1999; Anon, 1970). This results in low economic returns and long payback periods for investing in weed management (Ash & Stafford Smith, 2003; Brown *et al.*, 1999; Anon, 1970). It is therefore important to establish efficient weed management strategies to reduce costs (Taylor & Hastings, 2004; Byers *et al.*, 2002; Grice, 1998).

In controlling woody weeds, rangeland weed management must consider a number of factors including the vastness, remoteness, and heterogeneity of these systems, along with the modest labour supplies and resources available (Grice *et al.*, 2004a). Periodic and episodic recruitment of weeds due to climatic events complicates weed management decisions (Lawes & Grice, 2007; Grice, 2000). Unlike weeds in cropping systems, which are managed throughout the year, it may be many years between woody-weed controls. A number of woody-weed control methods are available, including chemical, mechanical, fire, and biological control. Methods vary in their efficacy and cost depending on the state of the infestation and the species being controlled. These control methods can be combined to derive an IWM strategy to optimise returns. Although it may appear that rangeland weed management offers little financial benefit, these properties are vast and a low return from weed management per hectare can still generate large returns at the property scale (Noble *et al.*, 2005).

Little is known about the population dynamics of long-lived woody weed infestations and the temporal effects of both damage to pasture production and weed control, as there are few long-term studies of these populations (Grice, 2000; Downey & Brown, 2000; Grice *et al.*, 1994). Whilst cropping systems can rely on the quantity of weed seeds or plants per unit area to estimate weed density, woody-weed management must consider both the number and size of individual plants in assessing weed density, as this factor affects competition between weeds, carrying capacity for livestock, and the cost and efficacy of different weed controls strategies. In many cases, rangeland graziers must manage a range of weed species with different biological characteristics, impacts on pasture production, and weed management efficacies (Lawes *et al.*, 2006). Moreover, many of these weed species often coexist. The impact of the infestation on pasture production and weed management decisions must be assessed for many years into the future for long-lived weeds. In the absence of long-term ecological studies, population models can be used as surrogates (Hester & Cacho, 2003).

Many factors influence optimal decisions for woody-weed control at the farm level (Auld *et al.*, 1979a). Before attempting optimisation, it is prudent to first establish a robust economic framework on the cost and benefits of different combinations of weed management strategies. With this information, a manager can assess whether a given decision is appropriate under the current situation. Similar to other investment decisions such as purchasing capital equipment, graziers should weigh up all future costs and benefits of weed control.

The aim of this research is not to establish the optimal weed management strategies for a woody weed. Rather, it is to establish an economic framework that can be used for a suite of different species of woody weeds within rangelands systems. In this study, a decision tool is developed that can be applied not only to an individual infestation but also generalised to a region. Benefits are measured as the difference in returns between management and non-management of a woody-weed infestation. Returns are measured as grazing gross-margins per hectare and other aspects of the production function at the farm level are not considered.

1.1 STUDY AREA

Rangelands are vast areas that include natural grasslands, shrublands, woodlands, and savannas (Donohue *et al.*, 2005). They represent between 70 and 75 % of the Australian continent (Figure 1-1), equating to approximately 6 million km^2 (Grice, 2006; Donohue *et al.*, 2005; Foran *et al.*, 1990). These systems are used by graziers for beef or wool production and are unsuitable for cropping or improved pasture systems as the rainfall is too low or variable (Range.Aust, 2007; Donohue *et al.*, 2005; ASEC, 2001; Scanlan *et al.*, 1991; Harrington *et al.*, 1984b). Approximately 13 % of the Australian population lives in the rangelands and there are over 20,000 specialised grazing properties (Donohue *et al.*, 2005). The northern Australian rangelands have a monsoonal climate, with hot-wet summers and warm-dry winters. Southern Australian rangelands tend to have hot-dry summers and cold-wet winters (Cook & Heerdegen, 2001; Suppiah, 1992). Whilst the far northern regions of Australia are typically associated with monsoonal rains, even central Australia ($20^oS–25^oS$) receives 50 to 80 % of its rain during the summer, with average January rainfall exceeding 100mm (Aust.Gov, 2002; Suppiah, 1992). This study primarily explores issues of relevance to the northern rangelands, with its seasonal and episodic recruitment of woody weeds. Whilst the model is designed for the northern Australian rangelands, as a case study this research uses rainfall data from a smaller region (Charter Towers – Figure 1-1).

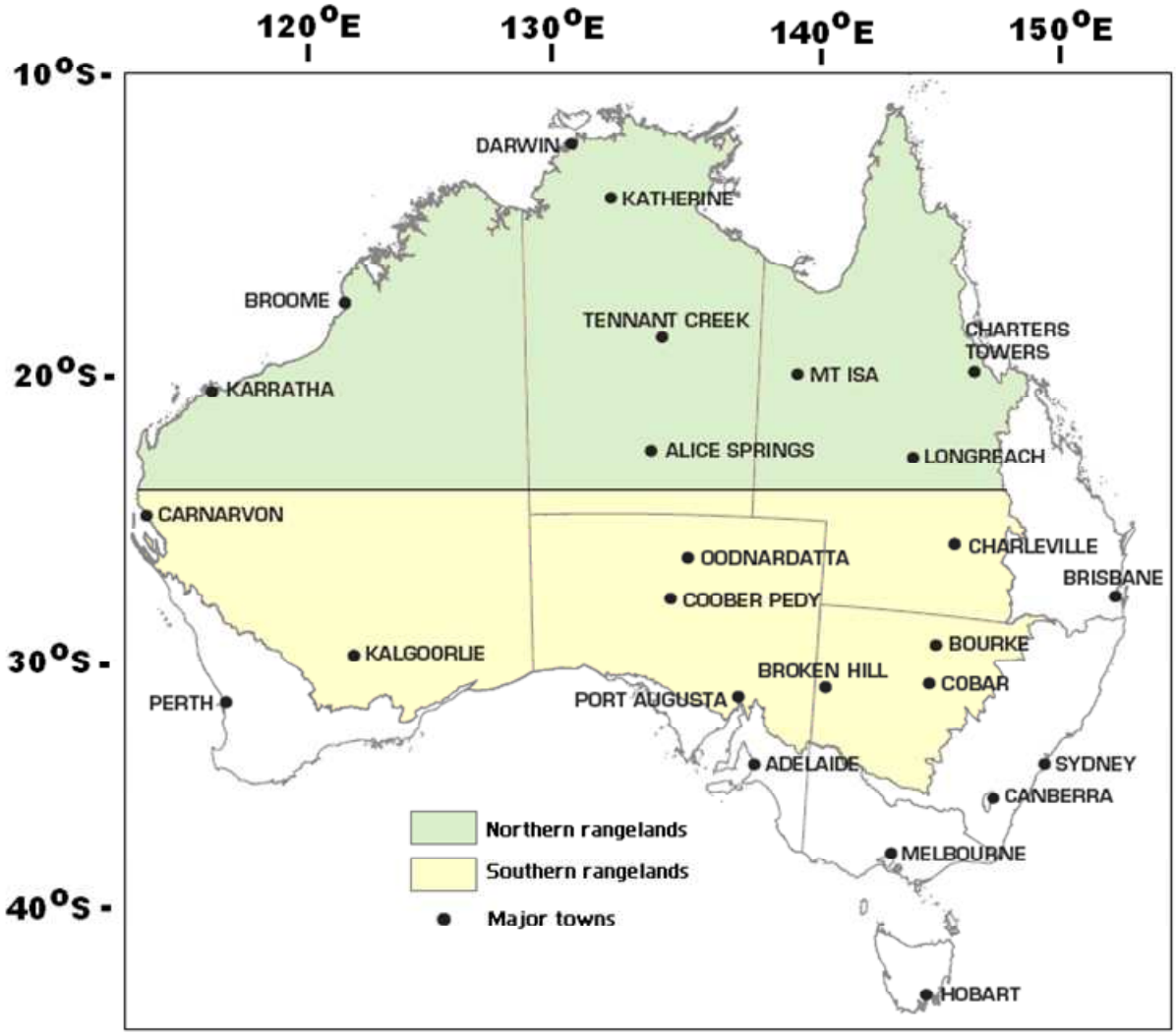


Figure 1-1: The Australian rangelands are divided into northern and southern rangeland systems based on rainfall events.
Credit: (Bastin G & the ACRIS management Committee, 2008; Aust.Gov, 2002; Suppiah, 1992).

1.2 PROBLEM STATEMENTS

A number of research problems exist in the economic management of woody weeds within the Australian rangeland systems:

1. It is unknown if woody weed control within the Australian rangelands is cost effective. If it is, then the optimal IWM strategy is yet to be determined.

2. Whilst the efficacies of different control methods are well understood in the year of control, it is yet to be determined how they translate into financial benefits over time.

3. The population dynamics of an infestation affect weed management decisions but there is poor quantitative knowledge of this process.

4. It is yet to be determined whether long-term benefits exist from the use of biocontrol on woody weeds, either by itself or as part of an IWM strategy.

5. There are many different woody weed species, with multi-species woody weed infestations being common within the rangelands. These infestations are problematic from a weed management perspective and it is important to determine whether management strategies should differ from those of single species infestation.

6. Episodic recruitment of woody weeds within northern Australian rangelands is a driver for many infestations but how this affects weed management decisions is unclear.

7. External stakeholders may need to assist graziers financially to intensify weed control but the required level of compensation depends on the unknown level of public benefit.

1.3 THESIS AIMS

This research has eight aims to address the research problems highlighted in Section 1.2:

1. To review both economic and ecological literature to identify current knowledge (and knowledge-gaps) to aid in establishing a bioeconomic framework for the optimisation of woody weed management.

2. To construct an ecological density-dependent population model to determine the most vulnerable life stages.

3. To derive a woody-weed damage function that measures damages to grazing production and to estimate the temporal benefits to pasture production from different weed control strategies.

4. To investigate whether biocontrol of woody weeds has the potential to return positive economic returns on investment.

5. To study the dynamics of multispecies infestations with a model and determine the effects of various weed management strategies.

6. To simulate the stochastic processes of woody-weed recruitment based on historical rainfalls.

7. To establish an optimising bioeconomic framework for the IWM of woody weeds within the northern Australian rangelands.

8. To calculate the level of compensation required by graziers from external stakeholders to implement IWM policies, when the cost of woody weed control exceeds private benefits.

1.4 THESIS STRUCTURE

The research aims are addressed in six of the eight chapters of this thesis (Figure 1-2) and a synthesis of research outcomes is presented in the final chapter.

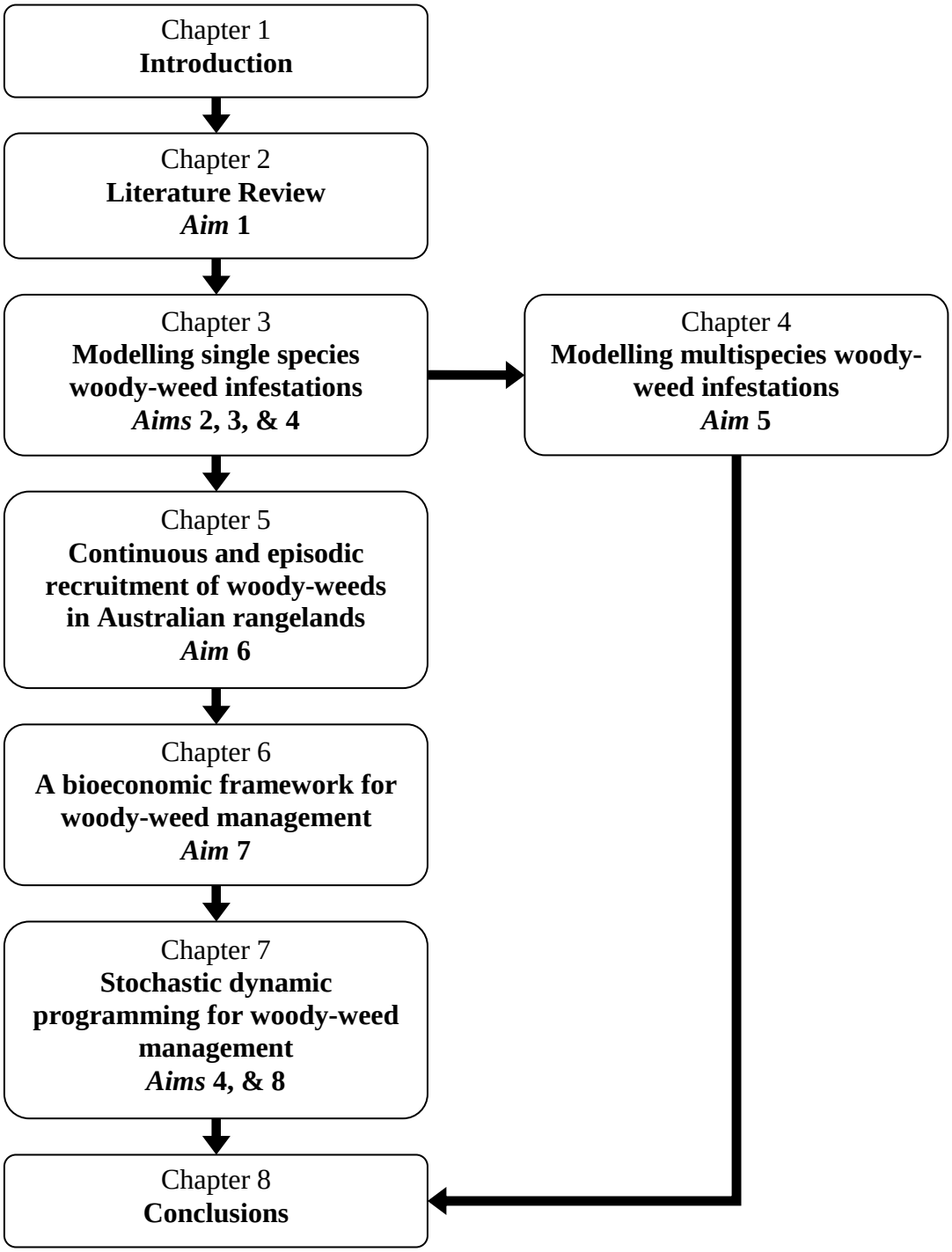


Figure 1-2: Flow chart of the thesis structure – linking the chapters with the research objectives.

Chapter 2 provides a review of weed management strategies, methods of control, and ecological and economic paradigms of woody-weed management. The chapter highlights the strengths and weaknesses of two paradigms to help establish a conceptual framework for the IWM of woody weeds. Chapter 3 establishes a density-dependent population model for woody weeds within rangeland systems. This chapter provides an understanding of the population dynamics of three different species of woody-weed infestations and the effect of weed control, including biocontrol. Chapter 4 extends the model and provides an understanding of multispecies infestations with respect to the effects of density dependence and weed control. Chapter 5 investigates the effects of climatic conditions on woody weed population dynamics and develops a stochastic model of episodic recruitment. Chapter 6 establishes a bioeconomic optimisation framework that captures the population dynamics of long-lived woody weed infestations within a stochastic dynamic programming model. Chapter 7 establishes the threshold-frontier, where the management of the woody weed infestation is deemed optimal with respect to the cost and efficacy of weed control and weed-free grazing gross-margins. In essence, this method provides multiple thresholds for different control actions with respect to the state of the infestation. The benefit of biocontrol as part of an IWM strategy is also investigated. Finally, Chapter 8 presents a synthesis of the outcomes of this study and identifies the opportunities for future research.

2 REVIEW – WOODY WEEDS, MODELLING, AND ECONOMIC & ECOLOGICAL PARADIGMS

This chapter reviews both ecological and economic literature to facilitate the development of an economic optimisation framework for woody-weed management in the northern Australian rangelands (Aim 1). This review defines woody weeds within the Australian rangelands and the demography of three case-study woody weeds: *Ziziphus mauritiana* (chinee apple), *Acacia nilotica* (prickly acacia) and *Parkinsonia aculeata* (parkinsonia). It also outlines the management options for these weeds, methods of control and the strengths and weaknesses of economic and ecological paradigms to establish an economic framework for woody-weed management.

2.1 DEFINING WEEDS

Any plant can be a weed (Fryer & Makepeace, 1977). There are many definitions of a weed, which in essence it is a plant growing in a place that is undesirable from a human perspective (Grice & Campbell, 2000; Akobundu, 1987; Fryer & Makepeace, 1977; Harlan & de Wet, 1965). The 'undesirability' of weeds means that it is more of a 'socio-economic' term than a biological concept and is both time and location dependent (Grice & Martin, 2006; Sindel, 2000c). For example, canola (*Brassica napus*) is a desirable plant used for human consumption but its status can change from a desirable plant to a weed when it grows in another location such as another crop, national park, or in the same field but in a different year when another crop is being grown. Likewise local native plants within a canola crop may be deemed weeds. As Harlan and de Wet (1965) stated "…one man's weed is often another man's crop." Although not officially declared as weeds, many Australian rangeland graziers consider endemic plants such as *Carissa ovata* (currant bush), *Cassini arcuata* (chinese shrub or sifton bush), and *Acacia harpophylla* (brigalow) to be woody weeds and would like to control them (Campbell, 1991; de Corte *et al.*, 1991; Combellack, 1987).

2.2 WOODY WEEDS IN NORTHERN AUSTRALIAN RANGELANDS

Woody weeds can form thickets that increase variable costs due to increased mustering effort, personal injuries (due to thorns), and damage to capital and decreased property values[1]. Woody weeds also increase stock losses, stock mortalities (including poisoning of cattle), impede cattle access to waterways, decrease land carrying capacity and reduce land condition (Parsons & Cuthbertson, 2001; Grice, 2000; Mackey, 1998; March, 1995d; Macleod, 1993). There are many exotic fodder, horticultural, and ornamental plants that are well established within the Australian rangelands (Martin *et al.*, 2006; Parsons & Cuthbertson, 2001; Lonsdale, 1994). The primary woody weeds in the northern Australian rangelands are listed in Table 2-1.

Woody weeds are thought to be more prominent in the northern Australian rangelands than in the south (Grice, 2000). In a survey conducted by Bortolussi *et al.*(2005a), over 68 % of producers in the northern Australian beef industry reported woody weeds on their properties and 96 % of producers in northern Queensland had woody-weed infestations. March (1995a; 1995b; 1995c; 1995d) also found that 77–93 % of grazing properties in north-west Queensland had woody-weed infestations, with 58–74 % of properties reporting financial costs, time, and labour as the primary limiting factors in controlling woody weeds. Assuming that both time and labour restrictions can be overcome with additional finance; then finance retards 71–93 % of the properties from undertaking woody-weed management. Despite these restrictions, between 88–100 % of properties continued to be involved in woody-weed management. Woody weeds may provide shade, dry-season fodder and fix nitrogen (Zhai *et al.*, 2004; Kriticos *et al.*, 1999a), but are still considered one of the greatest threats to Australian rangeland graziers (Landsberg, 2005; White, 2005). Only 4–12 % of graziers in the north-west Queensland thought prickly acacia and parkinsonia were of benefit (March, 1995b; 1995c).

[1] It appears however, that Australian rangeland property prices are less driven by pasture production and land condition than the prospects of higher future land prices, regardless of condition (Reichard & Hamilton, 1997; Miller, 1996).

Table 2-1: Common northern Australian rangeland woody weeds

Acacia catechu (cutch tree)	a, c, d
Acacia nilotica (prickly acacia) – (WoNS)	a, b, c, e
Azadirachta indica (neem tree)	b, c
Calotropis procera (calotrope)	b, c, d
Cascabela thevetia (captain cook tree, yellow oleander)	b, c
Cryptostegia grandiflora (rubber vine) – (WoNS)	a, b, c, d, e
Jatropha gossypifolia (bellyache bush)	a, b, c, d
Lantana camara (lantana) – (WoNS)	b, c, e
Leucaena leucocephala (leucaena)	b, c, d
Mimosa farnesiana (Mimosa bush, prickly moses)	b, d
Mimosa pigra (giant sensitive tree) – (WoNS)	a, b, c, e
Parkinsonia aculeata (parkinsonia)	a, b, c, d
Prosopis spp. (mesquite, algarroba, Cloncurry prickle bush) – (WoNS)	a, b, c, d, e
Ricinus communis (castor-oil plant)	a, d
Tamarix aphylla (Athel pine) – (WoNS)	c, d, e
Ziziphus mauritiana (chinee apple, chonky apple, Indian jujube)	a, b, c, d

Reference key: (a) Parsons and Cuthbertson (2001), (b) Grice (2003), (c) Martin *et al*. (2006), (d) Alexander *et al*., (2007), and (e) Thorpe and Lynch (2000) – Weeds of National Significance (WoNS).

Few data are currently available on the location and distribution of most rangeland weeds and little is known about their biology or control efficacies (Martin *et al.*, 2006; Campbell *et al.*, 2004). Despite all of these species continuing to expand their footprint in the Australian rangeland and the high possibility of more being introduced, few studies have attempted to quantify their economic impact within the Australian rangelands (Grice, 2006; Reichard & Hamilton, 1997). Ecological population models have been constructed for some rangeland weeds (Raghu *et al.*, 2006; Kriticos *et al.*, 2003a; Kriticos *et al.*, 2003b; Watson *et al.*, 1997a), but these have not been coupled with economic analyses to develop an economically optimal weed management strategy.

Graziers need to know if the benefits of weed management outweigh the costs over different time horizons. Weed costs are generally incurred up front, whilst the benefits are accrued over many years in the future and the benefits are therefore difficult to assess. The economic benefits and costs of managing woody-weed invasions can be synthesised with models. These models can provide new insights about weed management to assist graziers with complex weed management decisions. To achieve this requires an understanding of weed population dynamics, impacts on production and the efficacy of the management actions (Jones, 2003).

Three woody weeds: prickly acacia, parkinsonia, and chinee apple, were selected as case-study weeds for this research to establish a generic weed-management framework for riparian and upland zones of rangeland habitats. These weeds were selected based on available data and because they encompass a good range of population structures, population dynamics, longevity (short to very long lived), age to maturity (months to years), and methods of control. Additionally these are among the more widely spread weeds and have the potential to spread much further within the northern Australian rangelands (Grice, 2002; Grice *et al.*, 2000; Kriticos *et al.*, 2000; March, 1995d). Prickly acacia and parkinsonia have been declared Weeds of National Significance (WoNS) (Thorpe & Lynch, 2000). Chinee apple is a declared weed in Queensland, the Northern Territory and Western Australia under the Land Protection (Pest and Stock Route Management) Act 2002, the Weeds Management Act and the Agriculture and Related Resources Protection Act 1976. However, none of these weeds can poison

livestock. The framework developed here has the potential to be adapted to other species if the relevant biological information becomes available.

All three species have been found in riparian[2] and upland zones within the Australian rangelands (Lawes *et al.*, 2006; Grice *et al.*, 2004a; Lawes *et al.*, 2003; Radford *et al.*, 2002). Although riparian zones represent less than one percent of grazing areas, they are a major contributor of weed and seed production (Lawes *et al.*, 2003; Radford *et al.*, 2001c; Brown & Carter, 1998). Riparian zones also have higher nutrient and moisture levels, therefore they support denser plant populations than upland regions (Lawes *et al.*, 2006; Grice *et al.*, 2000). Because of these differences the two areas are modelled separately.

2.2.1 Chinee apple

Chinee apple was first introduced into Australia during the mid–1800s for informal horticultural purposes (Grice, 2002; Parsons & Cuthbertson, 2001). Dense infestations are located around Townsville, Ravenswood and Charters Towers (Qld.). There are also isolated infestations across Queensland, the Northern Territory, and Western Australia (Grice, 2002; 1998; Grice & Brown, 1996; Carter, 1994). Chinee apple can be found in both upland and riparian zones (Grice *et al.*, 2000; Grice, 1996). Compared with other weeds, chinee apple has a slow spread rate and takes decades to establish and fully infest new sites (Grice *et al.*, 2000). Despite its slow spread rate, it has the potential to populate vast areas in northern Australia and is highly competitive (Grice & Maywald, 2003; Grice, 2002).

[2] "Riparian" zones are those areas within 50-100m of bore drains, and primary and secondary streams with either annual or perennial flows (Lawes *et al.*, 2003; Kriticos *et al.*, 2003a; Grice *et al.*, 2000), all non-riparian zones are referred to as "uplands".

Chinee apple is a spiny tree that grows to a height of 6–8m, with a canopy of up to 10m (Grice, 2002; Grice & Brown, 1996). It reproduces through fruit production and each drupe has a single seed (Grice, 1996). Chinee apple is long lived and can survive for several decades within the Australian rangelands. It takes around 5–6 years to mature (Grice, 2002; Campbell & Grice, 2000). Chinee apple seedlings are very vulnerable to dry conditions and most will perish during their first dry season. They are also unable to establish under the canopy of other trees (Grice, 1998; James, 1995). Once established though, chinee apple is drought and fire tolerant (Grice *et al.*, 2000). Most chinee apple seeds remain near or under the parent's canopy (Grice, 1996). Cattle, feral pigs and emus are the primary dispersal vectors (Grice, 2002; Grice *et al.*, 2000; Grice, 1996). The passing of the seed through the gut of animals can increase germination rates through scarification, being deposited in manure pats (providing moisture and fertiliser) and reduced competition when deposited away from other established woody plants (Grice *et al.*, 2000; Campbell & Grice, 2000; Grice, 1996).

2.2.2 Prickly acacia

Prickly acacia was introduced into Australia in the early 1900s to provide shade and fodder for stock (Parsons & Cuthbertson, 2001; Radford *et al.*, 2001c; Carter & Cowan, 1993). The main infestations are the within the Australian Mitchell Grass Plains and northern Brigalow Belt in Queensland. Infestations are also present in the Northern Territory, South Australia, and New South Wales, with the potential to become a serious weed over vast areas within the Australian rangeland systems (Palmer & Julien, 2005; Kriticos *et al.*, 2000; Mackey, 1998; Mooy *et al.*, 1992). Prickly acacia is now one of Australia's worst woody weeds as it has the potential to change vast areas of open savannah into woodlands (Kriticos *et al.*, 2003b; Radford *et al.*, 2001b). The transition of these landscapes will reduce grazing production, increase soil erosion, and threaten biodiversity (Johnson, 2004; Mackey, 1998; 1997).

Prickly acacia is a thorny tree that grows to a height of about 10m and reproduces via seed pods (Kriticos *et al.*, 1999a; Jeffrey, 1995). It is capable of producing thickets in both uplands and riparian

zones (Radford *et al.*, 2001c). Prickly acacia can live for 30–60 years and matures, to start producing seeds, within eight months to 12 years. More typically, seed production starts in 5–7 years in upland areas (Campbell & Grice, 2000; Kriticos *et al.*, 1999a; Mackey, 1998; Carter, 1994). Prickly acacia tends to grow faster, denser, and have higher fecundity within riparian zones (Radford *et al.*, 2001c). Many prickly acacia seedlings will perish in their first dry season but as they mature they become more drought resistant (Tiver *et al.*, 2001; Kriticos *et al.*, 1999a). Cattle are the main dispersal vector of prickly acacia through the consumption of seed pods (Tiver *et al.*, 2001; Kriticos *et al.*, 1999a; Mackey, 1998; Harvey, 1981). The number of seeds deposited decreases as the distance from the riparian zone increases (Radford *et al.*, 2001c). Scarification by age, fire, or passing through the gut of an animal also increases the germination rate of prickly acacia (Radford *et al.*, 2001a; Mackey, 1998; Bolton *et al.*, 1987). The micro-environment within dung pats increases the germination and survival rates of prickly acacia seedlings (Radford *et al.*, 2001c; Mackey, 1998). Water flows can also disperse prickly acacia seeds many kilometres away, as pods can float for up to 30 hours (Kriticos *et al.*, 1999a; Carter, 1990). Strong winds can also disperse prickly acacia seeds a short distance from the parent tree (<25m) (Mackey, 1998; Carter & Cowan, 1993). Episodic climatic conditions are believed to result in invasion pulses of prickly acacia recruitment (Kriticos *et al.*, 1999a; Brown & Carter, 1998; Carter & Cowan, 1993).

2.2.3 Parkinsonia

Parkinsonia was first introduced to Australia in the late 19th century as a shade and ornamental shrub (Grice *et al.*, 2002). It is one of the most widely distributed woody weeds in Australia, having spread into Queensland, New South Wales, the Northern Territory, South Australia, and Western Australia (McKenzie *et al.*, 2004; Grice *et al.*, 2004b; Parsons & Cuthbertson, 2001). While parkinsonia grows in upland zones, it is more prevalent in riparian zones (Lawes *et al.*, 2003). Its potential distribution includes most of the northern and southern Australian rangelands (Deveze, 2004; Parsons & Cuthbertson, 2001).

Parkinsonia is a spiny tree that grows to a height of 2–8m, with a canopy >7m, and reproduces from seed (Lawes & Wallace, 2008; Grice *et al.*, 2002; Parsons & Cuthbertson, 2001). Relative to other woody weeds, parkinsonia is shorter lived (<20 year lifespan) (van Klinken, 2006) and matures in 1–2 years (D. M. Nicholas 2008, pers. comm.). Most seeds remain close to the parent tree and seed is distributed primarily by water and cattle (Cochard & Jackes, 2005; Parsons & Cuthbertson, 2001). Water distribution of seeds results in invasion fronts tending to move downstream (Lawes & Grice, 2007). Scarification by fire or passing through the gut of the animal increases germination rates (Parsons & Cuthbertson, 2001).

2.3 TREATING WOODY-WEED INFESTATIONS

There is a clear distinction between controlling and managing weeds. Control involves an action (including burning, poisoning, and the mechanical removal of plants). Management is the goal that is sought, using control actions to achieve these goals (Buhler *et al.*, 2000). Goals often move through prevention, eradication, containment (slowing the spread), maintaining[3], and in some cases ignoring an infestation. These goals move as the state of an infestation moves from pre-introduction, introduction, establishment, spread, and saturation (Grice, 2009; Hulme, 2006; Sharov, 2004; Masters & Sheley, 2001). At a landscape scale, an individual species of woody weed can be in all five stages. At a paddock (field) level, they are most likely to be in only one of these invasion stages. Whilst prevention, eradication, and ignoring are mutually exclusive weed management strategies, containment and maintaining are not, as both strategies can be combined to achieve a desired outcome (Grice, 2009).

[3] There is a common, yet incorrect synonymous uses of the terms ‘maintain’ and ‘control’. Whilst both have been used interchangeably as management objectives (Grice, 2009), ‘control’ usually refers to the method in which a management objective is achieved. To avoid confusion the term ‘maintain’ is reserved for a management objective, and ‘control’ (i.e. the chemical or mechanical removal of plants) is the method of achieving this objective.

2.3.1 Strategic goals of woody-weed management

Prevention

Quarantining cattle and border controls are the two main strategies in preventing weed incursion (Hulme, 2006). Quarantining cattle requires them to be held in small holding pens for 6–12 days which allows all seeds to pass through their digestive systems prior to them entering weed-free fields (Grice, 2002; Tiver *et al.*, 2001; Radford *et al.*, 2001c; Grice, 1998). Quarantining cattle is arguably the most successful and cost effective strategy in weed management (Vitelli & Pitt, 2006; Tiver *et al.*, 2001). Cattle that are moved from an infested area to non-infested fields, need only be quarantined when the target weed is producing fruit (Grice, 1998). Border control utilises both natural landscape characteristics (soil type, hills, lakes, etc.) and man-made structures (fences, roads, etc.) to protect non-infested areas, or areas that have lighter infestations (Grice, 1998; Sharov & Liebhold, 1998). Cattle, for example, may be placed in a small (fenced) quarantine field, before being moved to uninfested areas. For another example, fences can be erected to stop cattle spreading seed into uninfested areas of the field. However, man-made structures such as fences are of limited use for preventing seed dispersal by feral pigs, emus, and wallabies (Grice, 2002; 1996).

Eradication

Within the Australian rangelands, eradication is unlikely unless it is sought early in the invasion process (Groves & Panetta, 2002). Weed eradication is considered successful when the target weed has not been detected in the area for a time period greater than the weed's seed longevity (Panetta, 2007). Eradication has three distinct phases (1) finding the weeds, (2) removing weeds and (3) monitoring the area.

Eradication has the appeal of ongoing economic and ecological benefits, with little ongoing weed management costs (Panetta, 2007). In order for an eradication program to be adopted, the expected net benefits should exceed the net costs. The likelihood of eradication having a net benefit decreases as the

infestation becomes more established (Cacho *et al.*, 2008; Grice, 2000; Myers *et al.*, 2000). Whilst it is perceived that eradication has solved the problem 'once and for all', there is often a risk of reinfestation from other areas (Wilman, 1996). Weed eradication is extremely difficult, due to the increasing marginal costs of finding and removing surviving weeds as their density decreases (Auld *et al.*, 1979b).

The cost of finding the last few weeds will most certainly exceed the short-term opportunity loss of reduced pasture production. One strategy to reduce detection costs is to decrease the frequency of monitoring within the increased time-elapses since the last sighting (Panetta, 2007). However, once a single weed is detected in an area (confirming non-eradication) the frequency of monitoring is again increased. There are a number of inherent problems with detecting weed populations:

(1) Detection is dependent on the searchers' attentiveness.

(2) Few people are skilled in plant identification.

(3) Landscape heterogeneity can result in inaccessible areas.

(4) The longevity of seeds is often unknown.

(5) Seeds cannot be detected until they have germinated (Panetta, 2007; Cacho *et al.*, 2006; Dewey & Andersen, 2004).

These issues often result in weed eradication being sub-optimal (Odom *et al.*, 2003). Cacho *et al*. (2008) established an economic framework to investigate the 'switching point' between eradication to control. They found that seed longevity is the main constraint in eradicating a woody-weed infestation. Unfortunately, in the Australian rangelands, discovery and intervention of weed infestations seldom occurs early enough to achieve eradication (Grice, 2000). Early control measures, however, may help reduce local impacts.

Containing and maintaining infestations

When an infestation is too large to eradicate and it is still expanding within the landscape, the optimal strategy may be to contain and maintain the infestation (Grice, 2009). Sharov & Liebhold (1998) showed economic benefits when the population spread rate is slowed, even when the uninvaded area is relatively small.

The spread rate of an infestation can be slowed by maintaining the infestation at a lower density, or by containing the infestation within its current field with both natural and man-made structures to reduce the rate of spread into other fields (Grice, 2009; Sharov & Liebhold, 1998). Barriers need not be fixed and can expand with the infestation. Biological control can also be used to retard weed expansion and will expand with the spread of the infestation. When using containment, the methods of weed dispersal and the potential range must be considered. The use of fences, for example, is ineffective in retaining wind-blown seed.

An infestation may be maintained at a lower density to increase pasture production through periodic management. Mechanical, chemical, fire, and biological control may be required to ensure a population remains at a low level (Grice, 2009; Odom *et al.*, 2003). Therefore, an infestation can be both maintained and contained by using different methods of control simultaneously to maximise economic outcomes.

Ignoring

When an area is fully infested by woody weeds or when the cost of controlling exceeds the benefits, the optimal decision may be to ignore it. Weed control is expensive and it may be financially prohibitive to manage an infestation (Grice, 2009; Odom *et al.*, 2003).

2.3.2 Methods for controlling woody weeds

There are four main methods of woody-weed control: fire, chemical, mechanical, and biological control (biocontrol), each with advantages and disadvantages (Vitelli & Pitt, 2006; Vitelli, 2000). There is no 'silver bullet' for weed control and follow-up control is always required (Grice, 2002; Vitelli, 2000). Combinations of different weed controls are needed to successfully manage weed infestations (Vitelli, 2000). It is common to apply only one type of woody-weed control in any one year, in a single application, although biocontrol may persist indefinitely. Integrated weed management (IWM) for woody weeds involves the application of different control methods in different years to achieve a management goal. This is discussed later in more detail.

2.3.2.1 Fire

Woody vegetation, including noxious weeds, has increased within the Australian rangelands as a result of overgrazing and the suppression of wildfire frequency over the last 150 years by pastoralists (Dyer & Smith, 2003; Noble *et al.*, 1997; Macleod, 1993; Carter *et al.*, 1991). Fires have been suppressed through management practices, such as fire trails, hazard reduction burns, and back-burning (Fensham *et al.*, 2005; Costello *et al.*, 2000; Noble *et al.*, 1997; Macleod & Noble, 1991).

There are mixed views on the effectiveness of fire in controlling woody weeds. A "prescribed burn" involves burning a defined area, in climatic conditions that will maintain a controlled grass and/or bush fire, using available fuel loads, primarily plant litter and standing or dry plant material to retard, kill, or remove woody vegetation (Schumann *et al.*, 2001). Most graziers and researchers agree that fire has little effect on most adult woody vegetation. However, it is unclear how the effects of fire on young plants and seeds impact the population dynamics of woody weeds.

The efficacy of burning woody weeds is dependent on climatic conditions, fuel loads, species, size of plant (age), ground slope and terrain (Vitelli & Pitt, 2006; Noble *et al.*, 2005; Dyer & Smith, 2003; Radford *et al.*, 2001b; Trollope, 1984). Fire has little effect on chinee apple or prickly acacia adults

(Radford *et al.*, 2001b; Grice, 1997). However, adult parkinsonia plants are adversely affected by fire (Grice *et al.*, 2004b). Moreover, fire can kill 100 % of prickly acacia seeds in cow dung, which represents a large proportion of the seedbank (Radford *et al.*, 2001c; Carter, 1994; Harvey, 1981). Because fire primarily affects juveniles in most species, the constant removal of juveniles can result in a decline of the population (Craig, 1999). As a result, burning is now generally used as part of an integrated weed management (IWM) strategy (Vitelli & Pitt, 2006; Buckley *et al.*, 2004; Bebawi & Campbell, 2002; Carter *et al.*, 1991). To control mesquite, Wilson *et al*. (2001) suggests using mechanical or chemical control, followed by burning in later years to control seedlings.

Although burning is inexpensive relative to other methods of weed control, it is still costly relative to the returns offered from grazing. Burning grazing land is not a cheap undertaking due to the cost of spelling, agisting, erecting/repairing fences, and creating fire-breaks (Schumann *et al.*, 2001; Landsberg, 2001; Grice, 1997; Ash *et al.*, 1997). Additionally, burning has associated risks, due to the possibility of losing control of the fire, resulting in damage to other fields, farm capital, and/or human life (Dyer & Smith, 2003).

2.3.2.2 Chemical control

Chemical control is a primary method for controlling weeds in the Australian rangelands and is often used for light to medium infestations with small to medium (<2 m) height woody weeds (DPI&F, 2007; Vitelli & Pitt, 2006; Mackey, 1997). Herbicides have been extensively used for weed management since the 1940s and can be applied to the foliage, stems or soil base by using foliar spraying, basal barking, cut and stumping, and soil applied herbicide (Grice *et al.*, 2008; Mackey, 1997).

Foliar spraying

Foliar spraying has been successfully used to control many woody-weed species and is generally reserved for controlling regrowth (Deveze, 2004; Spies & March, 2004; Grice, 2002). Foliar sprays are in liquid form and can be delivered by hand (usually with a backpack/knapsack), from a vehicle (i.e. a quad-bike with a mounted tank and pump), or by helicopter (for larger and denser infestations) (Vitelli & Pitt, 2006; McKenzie *et al.*, 2004).

Basal barking

Basal barking is a method of applying the herbicide to the lower stems (basal) of shrubs or trees, via a carrier being either water or diesel (Jeffrey, 1995). Basal barking has been successfully used to control chinee apple, prickly acacia, and parkinsonia (Deveze, 2004; Spies & March, 2004; Grice, 2002). Basal barking is recommended for small stemmed plants, as the cost of control increases with the size of the plants and the associated stem surface area (Grice, 2002; Kriticos *et al.*, 1999a; Mackey, 1997; Jeffrey, 1995). A major disadvantages of this method is the difficulty in gaining access to the lower branches, especially on spiny plants, and the time required to treat to larger plants (Grice, 2002).

Cut and stumping

Cut and stumping involves cutting the plant down at near ground level (<150 mm high) and applying a poison to the freshly cut sum within 15second of the cut being made (DPI&F, 2007; Grice, 2002; James, 1995; Jeffrey, 1995). This method of weed control is successful in controlling chinee apple, prickly acacia, and parkinsonia (Deveze, 2004; Spies & March, 2004; Grice, 2002). However, access to the base of spiny plants is often a problem and can result in personal injury, especially if limbs fall on to the operator (Grice, 2002).

Soil applied herbicides

Soil applied herbicides have been successfully used in controlling chinee apple, prickly acacia, and parkinsonia (Deveze, 2004; Spies & March, 2004; Grice, 2002). The herbicides can be in solid (pellets) or soluble form and can be administered by hand, vehicle, or aircraft (McKenzie *et al.*, 2004; March, 2004). This control process is triggered by rainfall, and the chemical is taken up by the roots of the plant. The main benefits of this control method is the ease and speed of application (DPI&F, 2007; Jeffrey, 1995). However, its major disadvantage is the collateral damage to non-target trees (Grice, 2002).

2.3.2.3 Mechanical control

There are five main categories of mechanical controls for woody weeds: hand pulling, chain pulling, bulldozing, stick raking, and blade ploughing (grubbing) (Vitelli & Pitt, 2006). Hand pulling weeds is unlikely to be undertaken for large areas and is usually reserved for individual plants on an ad hoc basis (Vitelli & Pitt, 2006). The other four methods of mechanical control usually involve a large track-type tractor. Depending on the weed species, the mechanically removed plants may be used as cattle fodder, pushed up into heaps and left to decay or burnt.

Chain pulling

Chain pulling involves dragging a large heavy chain between two large track-type tractors (Kasasian & Walker, 1971). This method of control can help control woody weeds without damaging grasses. Chaining rapidly removes trees and shrubs (Kasasian & Walker, 1971). Whilst it is effective on prickly acacia, it is ineffective on chinee apple (and smaller plants) (Vitelli & Pitt, 2006; Grice, 2002; Jeffrey, 1995). Chaining will often result in a mass recruitment of seedling after good rainfall, due to soil disturbance and decreased competition (Campbell *et al.*, 2004).

Bulldozing

Bulldozers can be used to control woody weeds by clearing whole areas or individual plants. Although clearing above-ground biomass might not result in a high kill rate it can allow access to plants, and regrowth can be treated by other means of control in the following years (Grice, 2002; James, 1995).

Stick raking

A stick rake is an attachment that fits on the front-blade of a large tractor (bulldozer) to increase the front-blade size (up to 6m), and often has a cutter bar just above the ground to shear the base of the targeted weeds (Mackey, 1997; Jeffrey, 1995). Whilst it can be used to remove the above ground weed vegetation, it is often ineffective in killing weeds such as chinee apple, as they require subterranean cutting (Jeffrey, 1995).

Blade ploughing

Blade ploughing involves using two blades (2.5–6m wide) simultaneously: one up high to push the tree and the lower blade (that sits 20–25cm below the ground) to cut the stems from the root stock (Bortolussi *et al.*, 2005b; Scanlan *et al.*, 1991; Kasasian & Walker, 1971). There are very few plants that can survive this method of control (Scanlan *et al.*, 1991; Scanlan & Anderson, 1981). Whilst blade ploughing is expensive per unit area, it is very effective. Blade ploughing is particularly effective in controlling parkinsonia and chinee apple (McKenzie *et al.*, 2004; Grice, 2002; Scanlan *et al.*, 1991).

2.3.2.4 Biocontrol

Biocontrol of weeds has been used for over 200 years. In the Australian rangelands it was first used in 1941 on lantana (a woody weed) (Julien & Griffiths, 1998; Swarbrick *et al.*, 1995). Biocontrol research is yet to be undertaken or is in its infancy for many Australian rangeland weeds. The two main types of biocontrol are classical-biocontrol and browser-biocontrol.

Classical biocontrol

Classical biocontrol is the introduction of a control agent (insect or micro-organism) from the host plant's place of origin, to permanently suppress the infestation to a lower steady-state and/or reduce its spread rate (Vitelli & Pitt, 2006; Masters & Sheley, 2001; Vitelli, 2000; Briese, 2000; Swarbrick *et al.*, 1995). Biocontrol is theoretically well suited for perennial-rangeland weeds, as both the host-plant and the biocontrol agent remain present throughout the year (Vitelli & Pitt, 2006; Vitelli, 2000). Moreover, biocontrol can operate over vast, remote, and inaccessible areas where other control methods fail or are financially prohibitive (Julien, 2006; Kasasian & Walker, 1971). One of the main benefits of biocontrol is that once established, it can help control the infestation indefinitely with little or no additional costs. However, it is expensive and can take many years to find a biocontrol agent (often >10 years) (Julien, 2006; Andres, 1977). *Phytophora* (dieback), although not strictly 'classical biocontrol' as it is found within the Australian rangelands, is being evaluated as a possible biocontrol agent for parkinsonia (Diplock *et al.*, 2006; Tomley & Evans, 2004; Tomley, 1995; Akobundu, 1987).

The primary risk with biocontrol is the required introduction of new organisms into complex ecosystems, which may result in unwanted outcomes (Julien, 2006; Masters & Sheley, 2001; Evans, 1998; Howarth, 1991). Biocontrol agents are rigorously tested prior to release, yet there are cases of attacks on non-target species (Mack *et al.*, 2000).

Browser biocontrol

The use of browsing animals such as goats and camels can be used as another form of biocontrol (Vitelli & Pitt, 2006; Campbell, 1991; Scanlan *et al.*, 1991; Kasasian & Walker, 1971). Browser biocontrol agents have the advantage of being able to target multiple woody-weed species in the same area (Harrington *et al.*, 1984a). However, unlike classical biocontrol which is often released into a region (which goes beyond man-made boundaries), browsing biocontrol agents are restricted to a field (Scanlan *et al.*, 1991; Harrington *et al.*, 1984a; Kasasian & Walker, 1971). Heavy stocking rates are required by browsing biocontrol agents to effectively kill woody weeds. This can reduce pasture

condition and increase soil erosion (Vitelli & Pitt, 2006; Harrington *et al.*, 1984a). There is also the ongoing cost of adding and maintaining fences. Moreover, if the biocontrol animals cannot be sold for meat, then the area offers no financial returns during the control period.

Often biocontrol agents are only deemed successful if the abundance of the infestation is reduced to a level where it no longer constitutes a problem or causes economic losses (Vitelli & Pitt, 2006). Van Klinken (2006) states that from a grazier's perspective, biocontrol is deemed successful when the size and density of infestation is reduced to a level where it has little impact on pasture production and/or requires little control effort. However, these definitions are incorrect from an economic perspective. As with all methods of weed control, the measure of success is the degree to which the benefits of control exceeds the costs (Auld *et al.*, 1987). Biocontrol will be deemed successful if it results in a positive net present value (*NPV*).

2.3.2.5 Integrated weed management

No one method of weed control can efficiently control rangeland woody weeds, therefore integrated weed management (IWM) strategies are required (Monjardino *et al.*, 2005; Masters & Sheley, 2001; Grice, 2000; Pimentel, 1982). The concept of integrating controls was first conceived in the 1950s in entomology (Vitelli & Pitt, 2006; Norris *et al.*, 2003; Sindel, 2000b; Thill *et al.*, 1991). Since then, it has evolved into integrated pest management (IPM) for managing single-species pests (animal, plant, insect etc.), groups of pests, or all the pests within a location (Buhler *et al.*, 2000; Thill *et al.*, 1991; Akobundu, 1987). As IPM includes all pests, then IWM is a subset of IPM (Jones *et al.*, 2006). IWM typically combines chemical, mechanical, fire and biological control methods (Vitelli & Pitt, 2006; Grice, 2000). The primary benefit of IWM is the increased effectiveness and efficiency of weed management, through complementary interactions. See Figure 2-1 for the theoretical benefits of combining mechanical control and biocontrol to manage a woody weed infestation. Four control options are considered. The population is initially at a steady-state (old) where the population is fluctuating due to external factors such as climate, but on average the population (size) damage is neither tending to increase or decrease. Option 1 is to ignore the infestation, where it remains at a

steady-state. Option 2 is to mechanically control the infestation to reduce the level of damage. The infestation is no longer in steady-state but over time it will return to the old steady-state of damage. Option 3 is to introduce a biocontrol agent. Initially the infestation is no longer in steady-state. However, over time it will trend to a new steady-state, where the infestation and the biocontrol agent are in equilibrium. This, however, may take many years. Option 4 is to use biocontrol and mechanical control as part of an IWM strategy. The mechanical control will dramatically reduce the damage of the infestation and the biocontrol can bring the infestation to the new steady-state, in a shorter time period than if no mechanical control is used. Although IWM is promoted in principle, it is seldom used in practice, due to lack of knowledge of the system and the temporal effects of different controls.

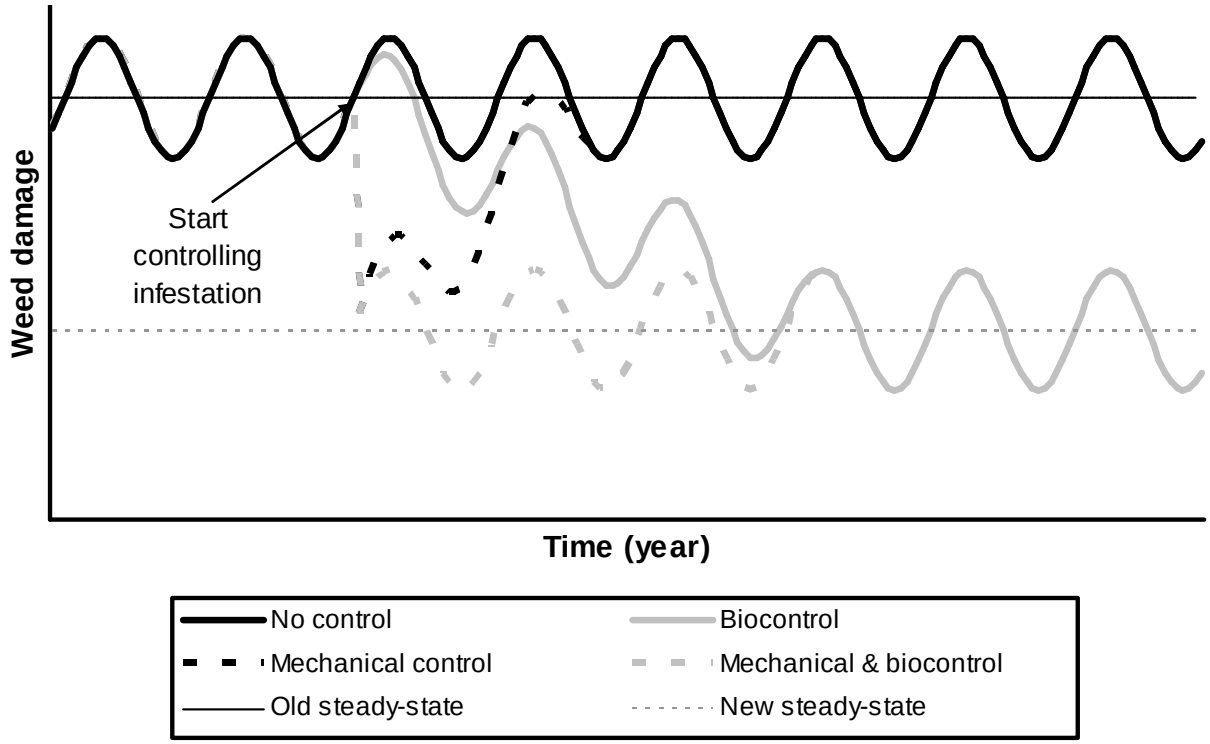


Figure 2-1: A schematic representation of different control strategies. Fluctuations are due to climatic conditions. Option 1: No control ——— (ignore the infestation), Option 2: mechanical control – – – (rapidly decreasing damage but returns to the old steady-state), Option 3: biocontrol ——— (decreases to a new steady-state), and Option 4: IWM of mechanical and biocontrol – – – (rapidly decreases to new steady-state).
Credit: (Norris *et al.*, 2003; Briese, 2000).

2.4 MODELLING PRINCIPLES FOR WOODY WEEDS

Models are simplifications or abstractions of the important elements of the real and complex world, built to gain conceptual clarity. Models are utilised for many reasons, including (Hester & Cacho, 2003; Higgins & Richardson, 1998; Bellamy & Brown, 1994): (1) communicating ideas and concepts, (2) showing temporal processes/behaviours of systems, (3) predicting the outcomes of different scenarios, (4) identifying knowledge gaps, (5) setting research priorities, (6) identifying a system's sensitivity to currently unmeasured variables, (7) evaluating current assumptions, and (8) as a surrogate for large-scale and/or long-run experiments. Conceptual models are typically more general frameworks, often constructed using optimal control theory in economics which provide general insight into a given problem. Empirical models are more specific and highlight the best actions in a given situation, usually using insights from conceptual modelling to interpret model results.

Modellers often have to weigh up the benefits and costs of model complexity to solve specific problems. As Cousens (1995) highlights, there is a marginal diminishing return to model complexity. It can be argued that weed invasions in natural ecosystems are so complex they preclude predictions through modelling. One strategy is to abandon model-simulations and rely solely on statistical analysis (which only describes historical events). However, this provides limited forecasting information, especially when there is little historical data or future conditions are changing. Alternatively, modelling frameworks can be developed to capture more of the system's complexity, to achieve more 'realistic' predictions (Higgins & Richardson, 1998). Even a truly realistic model can only estimate what 'might' happen (Cousens, 1995). Additionally, the more complex models become, the more obscure they are to stakeholders, who in turn may become more sceptical of the results (McCown *et al.*, 2009; Mccown, 2002; Smith & Foran, 1990). The key aim of a model is to capture key processes and dynamics of a system, whilst minimising the errors of a solution (Peters, 2004). The two broad classes of model errors (as with statistical analysis) are: (1) error of omission – excluding important processes and parameters, and (2) error of inclusion – including unnecessary or incorrect processes and parameters.

Another consideration for modellers lies in weighing up the accuracy of results and inference space, which is also the case for statistical analysis. For example, an analysis can be conducted for a single infestation within a field in northern Queensland. Whilst the analysis at this scale may reduce model errors (a randomly-constructed sample space), its inference space is also reduced to that one infestation (Hogg & Tanis, 1993). If the analysis is generalised for a geographical region, modelling errors (random sample space) will increase but so too will the inference space of the results. The answer to solving rangeland problems can be found in stepping back from fine-scale interactions (Higgins *et al.*, 2000). Therefore, it must be remembered that decreased sample space does not directly translate to increased accuracy in model solutions.

Whilst the generalisation of the solution to a problem may be required within a geographical region, this does not mean that the solution needs to be solved at a regional scale. As with many conservation issues, the scope of the problem is often greater than that of an individual landholder (Bathgate & Pannell, 2002; Pannell, 1998). This does not imply that farm level analysis is redundant (Pannell, 1998). On the contrary, many of the decisions that affect regional conservation issues are undertaken at the farm level. Therefore, farm based solutions are beneficial as (1) the final implementation of the solution depends on the actions of individual landholders, (2) it internalises the problem and solution, (3) it is not dependent on the participation of neighbouring landholders (which is often difficult to arrange), and (4) these models can assist and/or educate individual landholders in their decision making (Pannell, 1998).

2.5 ECONOMIC ANALYSES OF WOODY-WEED CONTROL

Since the 1980s, research on the economics of weeds has evolved considerably to include damage functions, threshold theories, benefit-cost analysis, linear and non-linear programming, optimal control, simulated annealing, multiple-phase optimal control, dynamic programming, stochastic dynamic programming, and genetic algorithms (Doole & Pannell, 2008; Doole, 2008; Jones *et al.*, 2006; Shoemaker, 1981). These analyses were used to generate weed management decisions that varied from simplistic single applications of control for a single species infestation, through to using optimisation techniques with IWM strategies for multispecies infestations and the temporal carryover effects. Nevertheless, woody-weed management has received considerably less research attention than cropping weed management (Zhai *et al.*, 2004).

Economic analyses of rangeland weeds had also started in the 1980s (Macleod & Johnston, 1990; Denne, 1988; Pannell, 1988b; Vere & Campbell, 1984). These studies investigated the management of both grass weeds and woody weeds and in turn helped develop various modular software programs for the management of the whole farm. The overarching software program from this research is called RANGEPACK, along with other software modules including SHRUBKILL (for controlling woody weeds with fire), which subsequently included BURNECON (as a decision tool for optimal burning) (Macleod & Ludwig, 1991; Stafford Smith & Foran, 1988; Ludwig, 1988). The management of woody weeds was integrated into the farm-level decision models but weed control decisions were not the primary focus of these studies. Additionally, this research only considered one method of control (burning), and relied on a simplistic benefit-cost analysis. Macleod and Johnston (1990) expanded this research to include other control methods and some combinations. Whilst this research moved towards the IWM of rangeland woody weeds, it oversimplified weed population structures and dynamics. Economic research for rangeland woody weeds is still dominated by benefit-cost analysis, whereas other weed-economic research in non-rangeland systems (for example crop-weeds) is far more advanced.

2.5.1 Contrasting cropping and rangeland weed problems

There are many reasons that economic analysis of rangeland weeds has lagged behind cropping weeds. These include:

- The intensity and financial returns per unit area are higher in cropping systems than in rangeland grazing systems.
- The cost of managing woody weeds is high, relative to the annual returns offered from grazing (Paynter & Flanagan, 2004; Grice, 2000). Therefore, many researchers deem it uneconomical to control woody weeds in rangelands systems (Grice, 2000; Kriticos *et al.*, 1999c; Mooy *et al.*, 1992).
- Relative to cropping systems (which tend to be more homogenous cultivations), rangeland systems are complex, vast, heterogeneous, and remote (Lawes & Grice, 2007; Grice, 2000).
- Farm cropping systems tend to operate on 100's of hectares, whereas farm grazing systems operate on 1000's of hectares.
- Cropping systems tend to have smaller annual weeds, whereas rangeland weeds include large perennial woody weeds and their associated population dynamics.
- Weed control within cropping systems tends to occur every year (with multiple applications of different controls). Whereas with rangeland systems only one method of weed control is used in any year and it is often many years between controls.
- Application rates of control for cropping systems can vary in response to weed density, whereas the application rate of woody weeds is fixed–control or non-control.
- It is difficult in capture the carryover effect of woody-weed management, where the effects of a single control can last for many decades (Pannell, 1988b).
- The episodic recruitment of woody weeds, due to climatic conditions, can change the population dynamics for many years and therefore affects the temporal effectiveness of control (Higgins *et al.*, 2000; Kriticos *et al.*, 1999a; Watson *et al.*, 1997a).
- There is difficulty in capturing the relationships between the number and size of woody weeds and their effect on pasture production and beef production (Pannell, 1988b).

Despite these complications, many graziers continue to manage these weeds without the insight of economic analysis.

2.5.2 Economic optimisation techniques

Dynamic optimisation rules return greater insight to weed research than static rules, as they capture the temporal benefits of weed management (Wu, 2001; Kennedy, 1988). Dynamic optimisation has its mathematical basis on optimal control (OC) theory, which is traditionally based on Pontryagin's maximum principle (Conrad & Clark, 1995). Direct numerical solution of OC problems in weed management is normally difficult. The two primary restrictions of traditional use of OC are: (1) the equation of motion must be continuously differentiable, which is problematic for the control of long-lived woody weeds where the growth of the population varies with different population structures, and (2) the inability to optimise problems with stochastic processes, such as episodic weed recruitment. Additionally, woody weed control decisions tend to have dichotomous application rates (control or non-control). These restrictions can be overcome in various ways by using numerical techniques such as mathematical programming, simulated annealing, dynamic programming, and stochastic dynamic programming (Doole & Pannell, 2008; Doole, 2008; Jones *et al.*, 2006; Shoemaker, 1981). There are also more complex algorithms available to solve stochastic versions of OC and to deal with multiple phases in cropping (Doole, 2008). One problem with using mathematical programming with non-linear functions is the problem of reaching local optima instead of a global solution. Simulated annealing can overcome this by using a stochastic search process, but this does not mean that the technique is well suited for stochastic problems (Doole & Pannell, 2008). Stochastic dynamic programming is flexible enough to overcome all the restrictions listed above, but it has the major disadvantage of the 'curse of dimensionality' which places restrictions on the number of state variables that can be considered (Shoemaker, 1981; Bellman, 1957). A more detailed discussion of these issues is provided in Chapter 6.

Earlier economic analyses of controlling rangeland woody weeds primarily relied on one method of weed control such as burning (Macleod & Ludwig, 1991; Stafford Smith & Foran, 1988; Ludwig, 1988). Odom *et al*. (2005; 2003) used DP to derive the optimal combinations of weed controls based on the state of a woody weed (scotch broom (*Cytisus scoparius*, L.)) in eucalypt woodlands in New

South Wales. This was one of the first true IWM strategies for controlling woody weeds. Chalak-Haghighi *et al.*, (2008) used a similar framework to derive an IWM policy for Californian thistle (*Cirsium arvense*).

2.6 ECOLOGICAL ANALYSES OF WOODY-WEED CONTROL

To derive the optimal management strategy, it is imperative to have a clear understanding of the basic population dynamics of the target weed and the benefits from alternative control strategies (Hao *et al.*, 2005; Grice, 1998; Scanlan *et al.*, 1991). One of the greatest criticisms of bioeconomic modelling is the lack of biological realism (Bulte & van Kooten, 1999). Deacon *et al*., (1998, p.391) state that "[e]conomists have largely stuck with simple paradigms that most biologists regard as useful pedagogical metaphors, but of little practical value"[4]. Ecological models and experiments have been used to learn about weed population dynamics and the effects of weed control (Stokes *et al.*, 2006; Williams *et al.*, 2006; Russell & Fowler, 2004; Brault & Caswell, 1993). This information can be included in bioeconomics to provide more solid scientific analysis than relying on the commonly used simplified growth models.

2.6.1 Evaluating the state of woody weeds

An important difference between optimising crop-weed and woody-weed management strategies is in evaluating the state of the infestation. For annual crop weeds it is justified to base the state of the infestation on the number of seeds or plants per unit area, with most weed cohorts in a similar life-cycle stage (Jones *et al.*, 2006). However, the size of woody plants affects the efficacy of different control methods. For example, burning has a greater effect on younger plants than on adult plants.

[4] Bulte & van Kooten (1999) have highlighted that economists are not alone in underutilising the knowledge from other disciplines, as many bioeconomic models developed by biologist/ecologists are based on over simplified economic analysis.

Therefore, for woody-weed infestations it is important to divide the population into different size classes, to get a truly optimal management decision. More recent woody-weed optimisation models have sought to capture the state of the infestation in various ways. Aldrich *et al*. (2005) based the state of the infestation on the age of the weed stand (assuming that the stand was even-aged pre and post control). However, individuals within the stand will be in different life stages. Odom *et al*. (2005; 2003) was among the first economists to derive the optimal management decisions for woody-weed management by dividing the infestation into the number of seeds and standing plants.

2.6.2 Ecological analysis of woody weed populations

A long-term study (15 years) has been undertaken for scotch broom, an Australian woody weed, and its incursions into eucalypt woodlands in New South Wales. This research has provided vital information on the weed's population dynamics and control efficacies (Paynter *et al.*, 2003; Downey & Brown, 2000). This information has been used in a number of ecological population models to investigate effective weed management strategies for this weed (Stokes *et al.*, 2006; Neubert & Parker, 2004) and has been used by Odom *et al*. (2005; 2003) to derive an optimal IWM policy.

Although many studies have investigated weed control efficacies, long-term studies of woody weeds within rangeland systems are rare due to financial and time restraints (Grice, 2000; Downey & Brown, 2000; Grice *et al.*, 1994). Ecological modelling techniques can provide information on current and future weed populations, including densities, rates of spread, and the potential locations for different plants (see Begon *et al*. (2006)). This information can be coupled with ecological principles to identify weaknesses in species' life cycles (Buhle *et al.*, 2005; Caswell, 2001; Parker, 2000; Brault & Caswell, 1993). Additionally, population models can be used to simulate the effectiveness of different control decisions.

Various ecological models have been used to investigate the growth and temporal control of chinee apple (Grice & Maywald, 2003), prickly acacia (Mooy *et al.*, 1992), and parkinsonia (Raghu *et al.*, 2006). It needs to be noted that none of these models has accounted for the long-term effect of density-

dependence, so their use for management decisions is more relevant for the shorter term. Woody weed must be managed over a longer time horizon (20–100 years) (Liedloff & Cook, 2007; Dyer & Smith, 2003). Long-term empirical studies have not been conducted on chinee apple, prickly acacia or parkinsonia, although remote sensing was used to monitor prickly acacia at the landscape scale over a fifteen-year period (Lawes and Wallace 2008).

Whilst long-term studies are desirable, they are expensive and there is a long time lag in getting the required information. In the absence of long-term studies it is possible to use models as surrogates (Hester & Cacho, 2003). Simulation models can estimate growth rates and the population dynamics over time, based on known ecological information and expert opinion (Hester & Cacho, 2003; Doyle, 1991). This approach has three main benefits: (1) it can provide an acceptable approximation for management polices (while the long-term study is undertaken), (2) it helps to identify the parameters that need to be derived from the long-term study, and (3) it highlights, from a management stance, the parameters to which the system is most sensitive. Some parameters may be biologically important but irrelevant in influencing weed management decisions.

2.6.3 Key ecological drivers of woody weeds

Soil types, available nutrients, light, ground formations, and average rainfall all affect the establishment and growth of a woody-weed infestation (Higgins *et al.*, 2000). These landscape attributes tend to be stable over time and easy to measure. Other factors, however, such as density-dependency, wildfire impacts, and climate variations will change over time, affecting woody-weed population dynamics and weed management outcomes.

The following sections will discuss the effects of density-dependence on the population dynamics of woody-weed infestations. Traditionally, the density of a weed infestation is based on the number of plants per unit area (Gotelli, 2001). This measure can then be used for modelling the density-dependence of an infestation and the level of weed damage. As weed density increases, so does the level of competition due to density-dependence, resulting in increased mortality rates within the

infestation. Simultaneously, the level of damage will also increase. Even though the competition within the infestation and damage from the infestation increase with respect to weed density, density-dependence and weed damage are not necessarily linearly correlated. The following section will look at the theory of density-dependence with respect to the quantity and size of weeds within an infestation. The quantity and size of the weeds will also affect the level of weed damage. However, similar to traditional measures of density, the two are not necessarily linearly correlated, as they are entirely different processes and have different ecological effects.

2.6.3.1 Density dependence (self-thinning)

As populations grow, their density increases as does the competition for resources (Harper, 1977; Yoda *et al.*, 1963). The term 'density' is used to convey crowdedness or the competition for limited resources and associated mortalities. When a weed dies, there is decreased density; its resources become available to other individuals (Begon *et al.*, 2006). Weed density is often measured as the number of plants (or seeds) per unit area (Buckley *et al.*, 2005; Kriticos *et al.*, 2003a; Tiver *et al.*, 2001). However, the number of seeds or plants is an insufficient measurement for woody-weed management, where the size and number of the plants affect control costs, efficacies, damage to pasture production, and population dynamics. The crowdedness of a woody weed population is better represented by the 'woody stock'[5] than by the number of individuals (Fensham *et al.*, 2005). The measure of crowdedness also affects the measure of weed control success. Often, the measure of weed control efficacy is measured by the number of plants after a control event but the removal of large woody weeds will commonly result in a mass recruitment of seedlings in the cleared area. Cochard (1999) provides photographic evidence of this for parkinsonia. Consequently, the efficacy of control cannot be solely assessed on the number of individuals per unit area. Rather, it should be based on the state of crowdedness or wood stock.

[5] Wood stock refers to the above ground vegetative material of the population.

To illustrate, Figure 2-2 shows two populations of the same woody-weed species. The shaded areas represent tree canopies. The roots of woody weeds often extend beyond the drip-line of its canopy, resulting in competition for resources beyond their canopies, as indicated by the dotted circle (Kriticos *et al.*, 1999b; Scholes & Archer, 1997). Although Figure 2-2(a) has twice as many plants as Figure 2-2(b), the latter is obviously more crowded. In Figure 2-2(a) there is no competition for resources between plants, as their areas do not overlap and the mortality rate of seedlings is density-dependent. However, in Figure 2-2(b), the plants are competing for resources within similar areas, despite this plot having less plants. Moreover, the likelihood of seedlings establishing and surviving is far less in Figure 2-2(b), as rangeland woody-weed seedlings are shade intolerant (Higgins *et al.*, 2000). For example, there are two seedlings illustrated in the top section of Figure 2-2(b), one is within the drip line of the larger plant. It is unlikely it will have any effect on the larger plant but the larger plant will greatly reduce the seedling's probability of survival. On the other hand, the adjacent seedling (in the centre away from the larger plants) does not appear to be affected by the other plants. Unless the model is spatially explicit, only an average density-dependent mortality rate for these seedlings can be estimated. It may appear that density could be estimated by total crown cover (i.e. the use of aerial

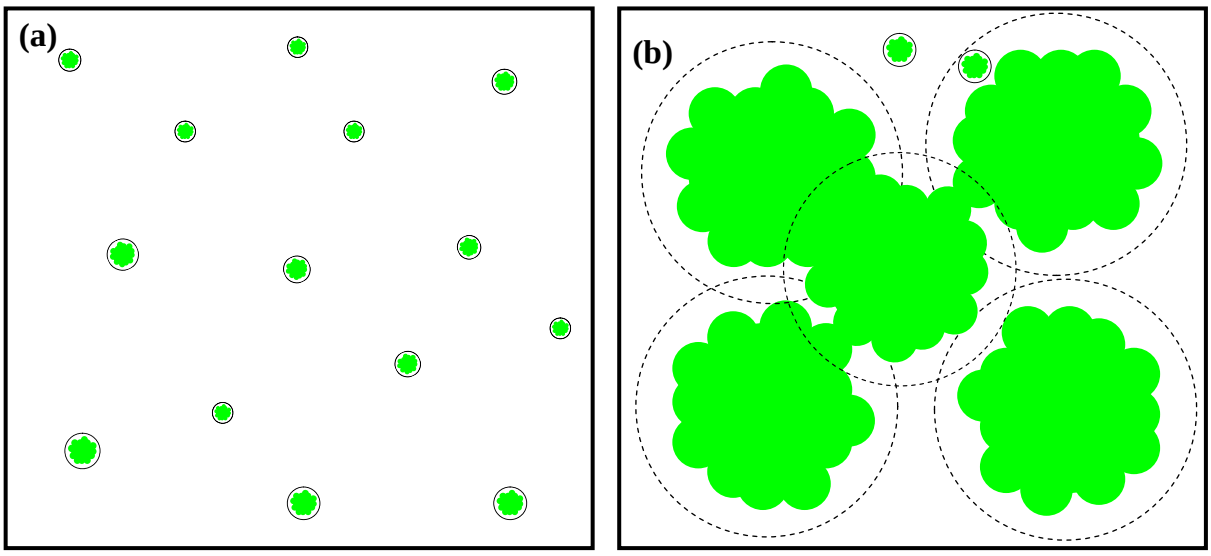


Figure 2-2: Illustrating the need to consider both the number and size of woody weeds per unit area, when evaluating population density, for; density-dependence, control evaluation, and production damage. Green areas represent tree canopies and the dotted lines are the area used by individual plants to attain nature resources, such as water, nutrients, and sunlight. Plot (a) has a younger woody weed population where n = 14, plot (b) is an older population with larger weedy weeds and n = 7.

photography) (Fensham *et al.*, 2005). This will not necessarily provide the information required to determine the effects of density-dependence on the different sized plants, unless the sizes of individual plants within the population structure are defined. Note that there is also less area for pasture production in Figure 2-2(b). This total damage could be estimated by total crown cover, despite there being plants of different sizes.

There have been empirical investigations into the effects of increasing plant sizes and the quantity of plants with respect to the effects of density-dependence on the population. To date, however, there is no unequivocal theory to explain the relationship between the number, size, distribution, and self-thinning (density-dependence) of plant populations (Kikuzawa, 1999). Yoda *et al*. (1963) derived the '–3/2 power law', where the number of plants decreases as biomass (mean weight) increases. Whilst there have been empirical studies that support this law (Westoby, 1984; White & Harper, 1970), there are others that have contravened it (Reynolds & Ford, 2005; Lonsdale, 1990; Weller, 1987; Ford, 1975). Parker (2000, p.738) states in her observations of *Cytisus scoparius* (scotch broom) that:

> *Of particular interest is the fact that total plant numbers, which is the traditional density measure in population biology, was entirely uncorrelated with all other measures of occupancy….it was biomass, and not simple numbers, that generated the demographic responses we expect from density dependence.*

Therefore biomass can reflect the population's size structure or intraspecies competition (Neubert & Parker, 2004; Parker, 2000). This indicates that competition between species (interspecies), and between plants of the same species (intraspecies) for woody vegetation should in fact be reserved for the degree of competition as a result of increased biomass or area occupied.

2.6.3.2 Modelling woody-weed competition

Both the total biomass and the number of plants per unit area not measures of competition, but measures of plant abundance per unit area (crowdedness). This can indicate the level of interspecies and intraspecies competition. Alternative measures of plant abundance per unit area (Pearcy *et al.*, 1989) include leaf area (Lotz *et al.*, 1996), area of breast height diameters (Liedloff & Cook, 2007), basal area (Liedloff & Cook, 2007; Campbell & Setter, 2002), dry weight measures (dry-biomass)

(Grice, 1996), and the area physically occupied within the landscape (crown cover) (Shugart & Smith, 1996). These measures can overcome the problem of using only the number of plants to represent crowdedness and capture the size of individual plants within the area. However, leaf area of woody weeds is a poor estimator of the impact on production (Lotz *et al.*, 1996). Additionally, measuring the leaf area of trees is laborious and beyond the scope of practical weed management options (Pearcy *et al.*, 1989). Breast height diameters and total basal areas are also of limited use to land managers in assessing when and how to control woody-weed infestations. The primary difficulty of this method is in the practicality of getting into infestation thickets and to the base of the plants to measure these diameters. Dried biomass also reflects the population size (wood stock), but is difficult to measure as the plant must be harvested, dried and weighed (Neubert & Parker, 2004; Parker, 2000). As woody weeds are sessile, competition for resources becomes a spatial process of capturing horizontal space within the rangeland landscape (Kriticos *et al.*, 2003a; Silvertown & Law, 1987; Smith & Goodman, 1986). Area is a limited resource and once fully occupied (captured), the woody-weed population will reach steady state. If area becomes available in the landscape (i.e. from weed control), the survival rate of woody-weed seedlings is positively influenced within this landscape gap (Jurena & Archer, 2003).

There are numerous woody-plant models, with many based on forestry. Porte and Bartelink (2002) separated them into two broad classifications: stand models and tree models. Stand models are primarily designed for timber projection and are less suited for species diversity. Tree models are more flexible and generally more appropriate in dealing with interspecies competition but rely heavily on descriptive relationships. Both stand and tree models include distance-dependent and distance-independent models. Distance-dependent is where the location of individual plants is known; distance-independent is where they are not known. Modelling for woody-weed management needs to be distance- independent, as it is impractical to measure the location of all the weeds in a field in order to manage them. Tree models include ‘patch models’, which generally examine vegetative dynamics at a spatial scale, with respect to area occupied by a small number of individual plants (Porte & Bartelink, 2002; Shugart & Smith, 1996). Patch models have been successfully used for woody weeds in the Australian rangelands, using process-based population dynamic models (Kriticos *et al.*, 2003a; Kriticos

et al., 2003b; Kriticos *et al.*, 1999c). However, model error increases as the number of weed cohorts increases. This is particularly problematic, as using different weed controls over time will increase the number of cohorts within an infestation, as not all individuals will change their state after control. An important subgroup of patch models is called “gap models”.

Gap models are common within forestry literature and have the capacity to capture intraspecies and interspecies competition. The concept behind gap models is based on “treefall gaps”, which occur in the forest canopies. The term “gap theory” refers to the processes involved in the recovery of the forest canopy (Moravie *et al.*, 1997). The dynamics of this recovery will depend on the growth rates of the different species of plants within the gap and the attributes of neighbouring plants, including their size (height and diameter) and horizontal position within the area (Shugart & Smith, 1996; Menaut *et al.*, 1990). Moravie *et al*. (1997) proposed that the dynamic processes of forest recovery can be primarily based on circular structures within landscapes, as crown diameters determines the level of energy received by plants through photosynthetic activity, as well as for other resources such as water and nutrients. The main drivers within the model where based on the size (diameter) and distance between neighbouring plants. Gap models have evolved into non-forestry research, including rangelands, where the location of each individual within the landscape is known (Menaut *et al.*, 1990). While gap and patch models will provide all the necessary information required for the management of woody weeds, they are computationally slow to run and data hungry for ecological information (Porte & Bartelink, 2002; Shugart & Smith, 1996; Menaut *et al.*, 1990). Moreover, the gap model developed for rangelands by Menaut *et al*. (1990) required the location of all individuals within the infestation, and even with this information the treatment of competition was found to be a shortcoming of the model. The population dynamics of woody weeds with the Australian rangelands is poorly understood (Fensham *et al.*, 2005; Shugart & Smith, 1996). Distance-independent gap models have been used to evaluate competition within forests (Porte & Bartelink, 2002). However, due to the lack of ecological information required about woody weeds, at this stage it is unlikely that gap or other patch models can be used in this research. Gap theory provides a concept of plant competition that can be applied to any population model that can provide both the quantity and size of individual plants within an area,

including multi species infestations. As Gap theory can be based on the competition for horizontal space, it can be assumed that, as the quantity of plants and/or the size of plants in an area increases, the average distance between plants will decrease and the level of completion will increase. An important distinction must be made about the term “gap”. It is not in reference to the space between the tree canopies (i.e. 1 - cover) but rather the gap in the landscape. For example, in Figure 2-2 it is the area outside the dotted line. In this way an area may be fully occupied, yet there is still space between the canopies which is commonly observed in rangeland systems. Conveniently, there is a correlation between the crown diameter and tree height (Grice *et al.*, 2002; 1996). Consequently, landholders can base their control decisions on the population’s plant height classes, which can be observed from outside the infestation. The following section will look at various other population models, including those listed by Porte and Bartelink (2002) as stand models.

2.6.3.3 Population modelling

The model required to derive optimal IWM policies of woody weeds must have the capacity to capture the population structure, density-dependence, efficacies of different control options, weed damage, and episodic events over time. Whilst it is possible to achieve many of these requirements with different equations, these models are usually reserved for problems with simpler population structures (Gotelli, 2001). Alternatively, mechanistic and process-based models require many variables (most of which are unknown). While cellular automata models can provide spatial information about probable spread rates and regions, the cells can only accommodate rudimentary ecological models before the processing capacity is exhausted. They are discrete models, with a grid of cells. Each cell is in one of a finite number of states. The population changes according to some fixed mathematical rule that determines the new state of each cell in terms of the current state of the cell and the states of neighbouring cells. In each time-step all the cells within the grid are updated simultaneously. Spatially explicit models are also difficult to parameterise and many of the required parameters are unknown. Markovian models, such as matrix models, depend on information about the present population state and assume future dynamics are independent of past events. It is also assumed that the transition probabilities between

states are constant over time (Caswell, 2001; Scanlan, 1994), but these restrictions can be relaxed. Firstly, whilst Markov chain models assume future population dynamics are based on the infestation's current state, it is understood that the population state retains a 'memory' of past events within it (Winkler, 2006; Grimm & Wissel, 2004). For example, if all the adult plants were removed in recent years this event would be captured within the current state of the infestation. Current management decisions therefore need not explicitly consider past control events. Secondly, matrix models can be dynamic with the probabilities within the matrix changing to reflect the effects of crowdedness and other factors such as climate (Winkler, 2006). One of the primary advantages of stage projection matrix models is the conceptual understanding they provide of population dynamics with relatively few parameters (Winkler, 2006; Raghu *et al.*, 2006; Caswell, 2001).

2.6.3.4 Matrix model structures

Land managers are often faced with weed management decisions without the knowledge about the history of the infestation; a Markovian matrix model satisfies this situation. Stage projection matrices are used to quantify the transition of a population through different life-cycle stages, as life stages are more reliable predictors of the dynamics of an individual than age (Caswell, 2001; Silvertown *et al.*, 1993). The life cycle stages of weed populations can be described by age using a Leslie matrix, or by stage classes using a Lefkovitch matrix (Lefkovitch, 1965; Leslie, 1948; Leslie, 1945). Although less common, dual classifications of age and stage can be undertaken with a Goodman matrix (Goodman, 1969). Matrix models have been successfully used to model the population dynamics of long-lived plants for both weed management and commercial forestry[6] (Paynter *et al.*, 2003; Parker, 2000; Lin *et*

[6] Like forestry, woody-weed management seeks to optimise its decisions over the long-term, based on the growth of long-lived plants. The primary difference between the two management problems is that: (1) in forestry the plants in question and their growth are seen as 'goods', but within rangeland systems they are seen as 'bads', (2) population structures in forestry are organised into stands (often of even-aged cohorts) whilst woody weeds are randomly distributed (and uneven-aged structures), and (3) at

al., 1996; Buongiorno & Michie, 1980). Examples of stage-projection matrix models for long-lived plants include *Pinus nigra* in New Zealand grasslands (Buckley *et al.*, 2005), *Eremophila maitlandii* and *Eremophila forrestii* (with episodic recruitments and deaths) in southern Australian rangelands (Watson *et al.*, 1997a), and *Cyperus rotundus* (tuber) population dynamics with 10 age classes (Neeser *et al.*, 1998).

Age matrices require one row and column for each year a plant can exist. This can result in very large matrices for long-lived plants and the transition between ages is often unknown and difficult to parameterise. For species that can live for several decades, it is more convenient to use life stage rather than age as the state. Although stage projection matrix models may appear less refined, the age-age projection matrix offers little insight into the population's habits (Caswell, 2001). Moreover, in the field ascertaining the life stages of woody weeds is easier and more reliable than estimating age by sight (Parker, 2000). Additionally, plants with identical ages may exist in different life stages, depending on their environmental conditions (Parker, 2000). Some woody weeds require a number of years to mature; therefore the juvenile elements of the stage should be based on age to retard the simulated plant from becoming an adult too early. A Goodman matrix (hereafter referred to simply as a stage projection matrix or SPM) is used to separate the juveniles by age and adults by size classes. This results in smaller matrices and population vectors than Leslie matrices, resulting in fewer calculations.

2.6.3.5 Episodic wildfires and rainfall

Fire and rainfall occur episodically within the Australian rangelands and have varying effects on the population dynamics of woody weeds (Fensham *et al.*, 2005; Higgins *et al.*, 2000). Fire can occur randomly from natural or unnatural events (Radford *et al.*, 2001b; Noble *et al.*, 1997; Noble, 1986; Harrington *et al.*, 1984a). Fire generally has little effect on established woody weeds and its impact is

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the end of the investment period all forestry timber is removed, whilst woody weeds may be left and ignored. Both systems undergo periodic maintenance which requires both labour and capital inputs.

soon lost, as most seedlings normally perish during the dry season (Vitelli & Pitt, 2006; Grice, 2002; Radford *et al.*, 2001b; Lonsdale & Miller, 1993).

Ecological research on the population dynamics of woody plants within the Australian rangelands has highlighted the importance of episodic recruitment as a result of climatic conditions (Radford *et al.*, 2001b; Higgins *et al.*, 2000; Kriticos *et al.*, 1999a; Brown & Carter, 1998; Watson *et al.*, 1997a). Years with exceptionally high rainfall can increase seedling germination and survival rates, changing population structures for decades. This is because, once established, woody weeds are long-lived (Kriticos *et al.*, 1999a; Brown & Carter, 1998; Carter & Cowan, 1993; Harrington *et al.*, 1984b). Land managers must consider the temporal effects of episodic woody-weed recruitment when determining the appropriate management strategy. To date, the stochastic population processes of woody-weed recruitment have largely been ignored by economists. Deterministic optimisation models of woody-weed management include those of Aldrich *et al*. (2005) for western juniper (*Juniperus occidentalis* Hook), Odom *et al*. (2005; 2003) for scotch broom (*Cytisus scoparius*, L.), and Chalak-Haghighi *et al*., (2008) for Californian thistle (*Cirsium arvense*)[7]. Chalak-Haghighi *et al*., (2008) suggested that further development in optimising perennial weed management should include stochastic processes. The analysis of Eiswerth & van Kooten (2007) is one of the few studies that derived stochastic optimisation management rules for a rangeland weed (yellow star thistle in California grasslands). This is discussed in more detail in Chapter 6.

2.6.3.6 Weed managers' requirements

One of primary problems with capturing the state of a complex system is the ability to actually measure the physical state of the system. As Cousens (1987) rightly highlighted, quoting an exact threshold value to a farmer is misleading as it implies a precision which is non-existent. An infested

[7] Although Californian thistle is not a woody weed, the analysis used for this weed showed many of the traits required for solving optimal management decisions for woody weeds.

area can change markedly from one season to the next and return to the original condition, without the total number and structure of infestation actually changing. Therefore, even if it is possible to model at a finer scale the actual application of the states may be very difficult and even impractical to observe out in the field. Additionally, the control decision may not even change with a finer scale of states. In a similar way, as the crowdedness of the infestation has been measured as the quantity and size of the trees ha^{-1}, this can be used to obtain a reasonable assessment of the population state and the appropriate management strategy.

2.6.3.7 Multispecies models – structure

Multispecies weed infestations are common within the Australian rangelands and to date, no study has looked at managing these complex systems (Lawes *et al.*, 2006; Grice & Maywald, 2003; Grice, 2000; Pannell, 1988b). In order to gain an understanding of controlling multispecies infestations, the dynamics of such infestations must also be understood. These infestations are particularly difficult to predict, especially when shocks such as weed management are introduced (Pannell, 1988b). "However, these more complex situations are the rule rather than the exception, and they cannot simply be ignored" (Pannell, 1988b, p.256). A weed manager needs to appreciate the succession process of the different weeds within multispecies infestations; otherwise the result will be suboptimal. Additionally, incorrect management could exacerbate the impact of some weeds (Scott, 2000). Multispecies weed infestations therefore also need to be investigated, with respect to control decisions. Some ecologists believe that matrix models are unsuited for multispecies infestations, as they are simple demographic models that do not support the required competition function (Winkler, 2006). However, matrix models are often used in commercial forestry to simulate the temporal dynamics of long lived multispecies populations. For example, Lin *et al*. (1996) developed a multispecies matrix models by conjoining individual species matrices for management decisions in commercial multispecies forest. This is discussed further in Chapter 4.

2.7 AIMS AND SUMMARY

Deriving sound economic woody-weed management policies requires an understating of key ecological processes, the efficacies and costs of different controls methods, and the potential benefits (Teague *et al.*, 2008; Hao *et al.*, 2005; Odom *et al.*, 2003; Grice, 1998; Scanlan *et al.*, 1991). This research seeks to establish an ecological-economic framework to derive optimal IWM polices for woody weeds in northern Australian rangelands by combining ecological and economic principles. These policies must be relevant to individual farmers at a field scale. There is a harmonisation between stage projection matrix (SPM) models and dynamic programming models as they both require the state of the infestation to have Markovian independence. Additionally, both models can capture stochastic processes, density-dependence and control efficacies (Caswell, 2001; Nemhauser, 1966). Their underlying principles are thus well suited for the optimisation of woody weed control. The modelled population dynamics will be density-dependent based on Gap theory. This theory considers both the number and size of individual plants within an area, to estimate plant competition. Each weed will be placed in different size classes. The level of competition of each individual (space occupied or contribution to crowdedness) is based on its size class. Furthermore, the stochastic recruitment of seedlings will also be included. This framework can be used to either maximise grazing profits or to establish the least costly method of reducing the weed infestation (when weed control is non-profitable).

This chapter addresses Aim 1 by presenting a review that defines woody weeds within the Australian rangelands, the methods for their control, and management strategies. The strengths and weaknesses of economic and ecological paradigms are discussed to establish the requirements for an economic framework to optimise woody-weed management. This review establishes the requirements of the woody-weed population models in Chapter 3, the multispecies woody-weed population models in Chapter 4, and the episodic recruitment events within the northern Australian rangelands in Chapter 5. This review also establishes a precursor to the optimisation framework developed in Chapter 6 and its application in Chapter 7.

3 MODELLING SINGLE SPECIES WOODY-WEED INFESTATIONS

This chapter seeks to establish a density-dependent, woody-weed population model. The model is then used within an ecological framework to investigate management of various rangeland woody-weeds (including those yet to be encountered). Chinee apple, prickly acacia, and parkinsonia have been chosen as case studies. All three weeds are monoecious perennial trees or shrubs. This selection was based on the available data and their tendencies to coexist (Lawes *et al.*, 2006).

The density-dependent stage projection matrix (SPM) will be used to estimate the temporal effects on woody weed populations from both natural and management events. Various measures of plant abundance were investigated for density dependence. Gap theory is used for density dependency as it is driven by both the number and size of individual plants within an infested area. It was selected for this study, as it can simulate the intraspecies and interspecies competition within woody weed infestations with respect to the quantity and size of woody plants. A woody-weed damage function is also developed within this chapter. This will assist in determining the temporal physical benefits of woody-weed control. These ecological analyses are not discounted for the time value of benefits, as this is undertaken in Chapters 6 and 7 in the economic optimisation of woody-weed management.

An eigenvalue analysis is also used to estimate population elasticities of new and steady-state populations to identify weaknesses in the weeds' population dynamics. The model is also used to simulate the efficacy of different weed management strategies (Neeser *et al.*, 1998) and the possible use of biological control (biocontrol) on woody weeds. The SPM models were implemented in MATLAB 7.0® (2007) using annual time intervals for a fixed area (one hectare).

3.1 METHODS

SPM models were constructed to simulate the population dynamics of three different woody-weed species in both riparian and upland zones. The models are based on a single hectare in the landscape with only one of the species present. Although the simulation is non-spatial, the single hectare model can be thought of as an average hectare in a zone over time. These models assist in estimating the different species' population structures and growth rates over time (Caswell, 2001). Eigenvalue analysis can be used to rank the different life cycle stages for management (Buhle *et al.*, 2005; Caswell, 2001).

Although there are significant ecological differences between chinee apple, prickly acacia and parkinsonia, their life cycles are similar. The problem can be simplified by assuming four main life cycle stages, new seed, seedbank, juvenile (a non-seed producing plant) and adult. The latter two are subdivided into different size (height) classes depending on weed characteristics. As there is a correlation between the crown diameter and tree height (Grice *et al.*, 2002; 1996), either can be used. Height is used for practical reasons as they can be observed on the ground from outside the infestation. Species vary in the number of juvenile or adult classes. The general model is first described and the derivation of parameters for each weed species is then explained. Variables and parameters are listed in Table 3-1.

The plant population at time *t* (years) is represented by a column vector, x_t, containing *n* elements, each element represents the number of individuals per hectare for the corresponding life stage. As this vector represents both the number and size of individual in an area, it also represents the population density. Uniform dispersal of seeds and plants over the area is assumed (Jones *et al.*, 2000). The population state transition is given by:

$$\mathbf{x}_{t+1} = \mathbf{H}\mathbf{x}_t \tag{3.1}$$

Table 3-1: Definition of model variables and parameters

Variables:	***Unit***	***Description***
$\mathbf{X}_t$	Weeds ha^{-1}	Weed population (size) at time *t*
NS	Seeds ha^{-1}	New weed seeds
SB	Seeds ha^{-1}	Seedbank
$J_{1...m}$	Plants ha^{-1}	Juveniles at stages 1 to *m*
$A_{1...q}$	Plants ha^{-1}	Adults at stages 1 to *q*
F	Seeds ha^{-1}	Fecundity
M		Probability of weed moving to next stage
R		Probability of weed remaining in its current stage
W	m^2	Area occupied by weeds
D	Percentage	Damage to pasture production at time *t*
π	\$ ha^{-1}	Grazing gross margins (cattle)
λ		Population growth rate
Parameters:	***Unit***	
H		Stage projection matrix (SPM)
$u_{1...w}$		Types of available weed control 1 to *w*
Φ	% of plants	Efficacy matrix for all weed controls
$\boldsymbol{\varphi}_{u_{k,1...n}}$		Efficacy of each weed control on each life stage
F		Average fecundity of an adult
γ	m^2	Maximum area available for occupation
$\boldsymbol{\omega}_l$	m^2	Area required by a weed of a given life stage to exist
$\boldsymbol{\psi}_l$	Proportion	Damage to pasture production per weed of given life stage
τ	Proportion	Maximum weed damage to pasture production
c_l		Elasticity for life stage.
$e_{i,j}$		Elasticity for elements of **H**
v_i	Weeds ha^{-1}	Maximum number of weeds in life stage (*i*) ha^{-1}
Subscripts:		
t		Time (years)
n		Number of life cycle stages for a weed species
m		Number of juvenile stages
q		Number of adult stages
w		Total number of different available type of weed controls
i,j		Denotes the elements in a matrix or vector
l		Life cycle stage of weed
wf		Weed free
Superscripts:		
T		Transpose matrix/vector

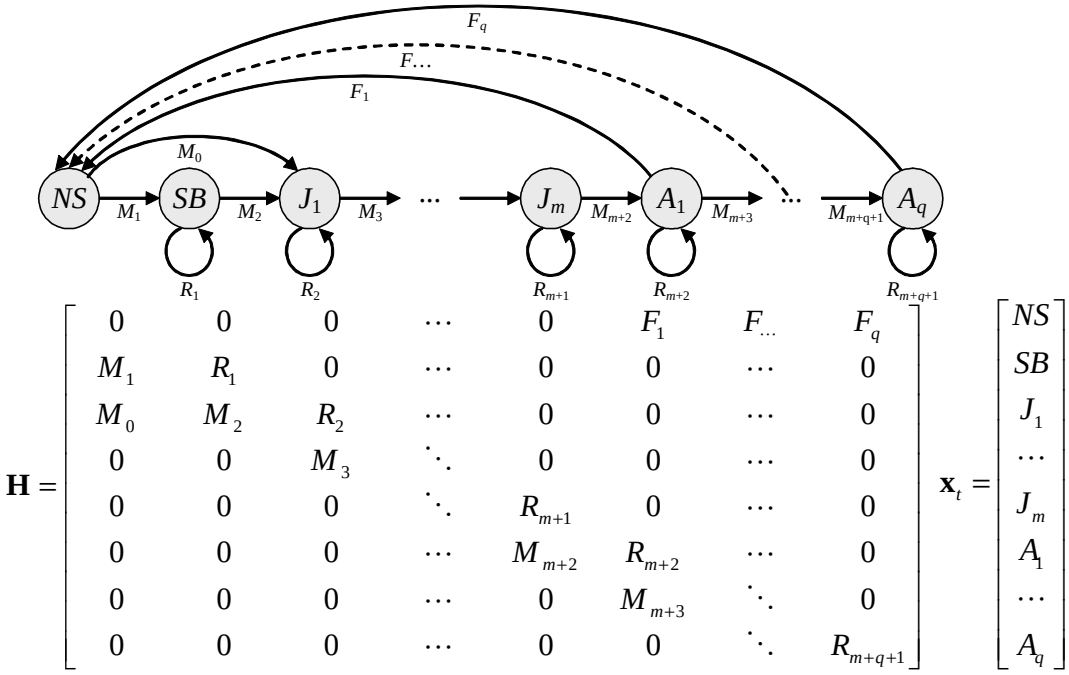


Figure 3-1: Life cycle diagram with the corresponding stage-classified population projection matrix (SPM) and population vector (adapted from Brault & Caswell, 1993).

where **H** is the annual SPM (Caswell, 2001; Brault & Caswell, 1993), of dimensions *n* x *n*; where *n* represents the number of life-cycle stages. The **H** matrix and corresponding column vector x_t are illustrated in Figure 3-1. The following steps are followed to derive *n*. First, the weed population is divided into four main stages, new seeds (*NS*), seedbank (*SB*), juveniles (*J*), and adults (*A*). The juvenile and adult stages are then broken down into sub stages (J_1,…J_m) and (A_1,…A_q), based on the time required to reach maturity and the plant's longevity, physical size, and fecundity. If it takes this plant a minimum of five years to reach maturity and start producing fruit, then $m = 4$. The value of *q* is derived as a compromise between model size and the need to account for differences in physical size and fecundity as a plant ages, i.e., if the life expectancy of a plant is 50 years and *m* is 4, ideally *q* would have a value of 46 (50–4). However, it would be difficult to obtain parameter values for each age group and it is impractical to follow each age group through time for the purpose of weed control. At the other extreme, *q* could be given a value of one, the fecundity set at the average for all mature age groups, and the survival rate set at a value that would result in the average plant longevity of 50 years. However, differences in plant size and fecundity between younger and older adults result in substantial differences in their impacts on pasture production. Therefore it is appropriate to divide the

adult stages into subgroups. If we let q=4, then n=10 for this weed (with 1 stage representing new seeds + 1 stage representing the seedbank + 4 juvenile stages + 4 adult stages). The fecundity for adult stages is represented by F_i, $1 \le i \le q$. The new seed stage (*NS*) provides flexibility in modelling the proportions of seeds from fecundity that will go into the seedbank (*SB*), germinate (J_1), or die. The probability of an individual remaining in its stage or moving on to the next stage is represented by R_i and M_i respectively. The mortality for any stage is 1– (R_i + M_i). The main assumptions of stage projection matrices are that all individuals in a given age group are identical and the probability of moving to the next stage is constant, regardless of the time already spent in a stage (Caswell, 2001; Brault & Caswell, 1993). This results in a deterministic model where the probability of an individual moving through its life-stages is predetermined. This assumption is valid for established plants, where population density has little impact on their survival due to the 'hierarchy of exploitation' of resources (Silvertown, 1982, p.125). However, this will not be the case for younger plants which are significantly affected by density-dependence. Therefore, their survival probability will be dependent on population density.

3.1.1 Introducing density dependence

The SPM for a new infestation is represented by H_0 (where density-dependence has no effect). If Eq. 3.1 is applied repeatedly to H_0, the population will grow exponentially. Exponential growth assumes all resources are limitless, and population density does not affect population growth (Lefkovitch, 1965), an unrealistic assumption for most ecological systems.

A density-dependent feedback mechanism is employed to simulate the impact of finite resources on population growth. The measure of density (crowdedness) used in this study is indicative of both the quantity and size of individuals per unit area, and is represented by a column vector x_t, capturing both the population's size and structure per unit area. A three step procedure was used to simulate this process. Firstly, the SPM for a new infestation H_0 was defined. Secondly, a steady-state infestation SPM, $\mathbf{H}_\infty$ is established based on the carrying capacity of an average hectare in the environment.

Thirdly, the transition between $\mathbf{H}_0$ and $\mathbf{H}_\infty$ is calculated. Cacho and Spring (2004) offer a plausible transition between these two matrices with a rectangular hyperbola based on total area occupied, resulting in a sigmoid growth curve (the method of determining this area is defined Section 3.1.2). The SPM at time *t* is given by:

$$\mathbf{H}_t = \frac{\gamma \mathbf{H}_0}{\gamma + \left(\frac{\mathbf{H}_0}{\mathbf{H}_\infty} - 1 \right) W_t} \tag{3.2}$$

Here the matrix division is an element by element operation and does not require matrix inversion. The maximum area available for occupation is represented by γ, in this case one hectare. The total area occupied by the population (*W*) at *t* is calculated by $W_t = \boldsymbol{\omega}^T \mathbf{x}_t$, where $\boldsymbol{\omega}$ is a column vector representing the average area required by individuals in each stage. The superscript *T* indicates this vector is transposed. When $W_t = W_{t+1}$, the population is at a steady-state. This means individuals within the population continue to go through the life cycles, while the total area of the population remains constant. This implies the impact on graziers becomes constant as the invasion matures.

3.1.2 Using Gap theory to estimate plant area

Gap theory was applied to estimate the elements of $\boldsymbol{\omega}$ (Shugart & Smith, 1996). This theory assumes that a unit area of land can only support a certain number of individuals based on their physical size and that competition exists for exploiting openings in the canopy. Biologically, this will be governed by competition between species (interspecies) and between adults and juveniles of the same species (intraspecies) for limiting resources such as light, water, and nutrients (Kikuzawa, 1999). In this modelling framework competition is primarily driven by plant crown diameters. The crown is a major source of energy through photosynthetic activity, which in turn drives plant growth (Moravie *et al.*, 1997); it is also correlated to the area occupied by the root systems to exploit water and nutrients (Kriticos *et al.*, 1999b; Scholes & Archer, 1997).

Gap theory provides a simple, yet plausible, mechanism of weighting the impact of individual plants on population structures that are quantity and size dependent. It is assumed that a plant requires a minimum amount of area in order to exist in the landscape. The area available for occupation (γ in Eq. 3.2) is set at 10,000m^2. The elements of $\boldsymbol{\omega}$ represent the average area (m^2) individuals require at various life stages. For a life stage i, $\boldsymbol{\omega}_i$ is derived by dividing the total available area (γ) by the maximum number of individuals in that life stage that can physically exist in that area (v_i):

$$\boldsymbol{\omega}_i = \frac{\gamma}{v_i} \tag{3.3}$$

For example, if a hectare can only accommodate 2,000 young adults (assuming there are no other weeds present in any other life stages) then, on average, a young adult requires 5m^2 to exist. This is also the amount of area each young adult will contribute to the population's total area infested. The value of $\boldsymbol{\omega}$ will change between species as they require different amounts of land for each life stage. The carrying capacity of upland and riparian zones also differ due to available resources for plant life. The maximum number of individuals that can physically occupy a hectare are calculated based on past studies, observations, or expert opinions.

The area required to support a plant does not, however, represent the area lost from production. A young adult plant, for example, that requires 5m^2 of the landscape to survive will not cause all this area to be lost from production. This is due to other vegetation, such as pasture, coexisting with the weed (this will be discussed in Section 3.1.4).

3.1.3 Population elasticities

The growth rate of a population (λ) is given by the dominant eigenvalue of matrix $\mathbf{H}$, and the continuous-time growth rate (r) can be derived from $r = \ln \lambda$ (Brault & Caswell, 1993). Although the calculus behind eigenvalues can be challenging, an eigenvalue itself can simply be thought of as the value of the transition between two vectors (Caswell, 2001). For example, if the value of a vector

doubles after being multiplied by a matrix, then the dominant eigenvalue of the matrix is two. If the value of the vector remains unchanged, then the eigenvalue is one. Conversely, if the dominant eigenvalue λ = 1.0, then *r* = 0, implying $\mathbf{x}_{t+1} = \mathbf{H}_{\infty}\mathbf{x}_t = \mathbf{x}_t$ and the population has reached steady-state. Therefore, by definition the dominant eigenvalue of the SPM at steady-state $(\mathbf{H}_{\infty})$ equals 1.0, and this premise was used to derive the demographic parameters of this matrix.

Perturbation analysis assists in determining which life cycle stage has the greatest effect on population growth rate. Biologists have used perturbation analyses to evaluate the sensitivity of population growth rates to changes in different life stages. Such analyses identify the stages and processes that are most critical for the survival of endangered species (Brault & Caswell, 1993) and these can be used to identify an invasive species' "Achilles heel" (Parker, 2000). The two common perturbation analyses are sensitivity (Caswell, 2007; Buckley *et al.*, 2003; Neeser *et al.*, 1998) and elasticity analysis (Stokes *et al.*, 2006; Parker, 2000; Silvertown *et al.*, 1993). Sensitivity analysis will often generate non-zero values for zero-valued elements of the SPM. This is possible as the analysis asks what would happen to the population growth rate if parameter h_{ij} is changed, irrespective of whether it is physically impossible (Caswell, 2001). Elasticity analysis only provides values for non-zero elements as it looks at a proportional change of h_{ij}, and a proportional change of a zero element is zero. Because elasticities are a proportional change they allow for an unbiased comparison of parameters which may have different orders of magnitudes or unit measures (Davis, 2006).

The elasticities of λ with respect to h_{ij} are often interpreted as the "proportional contribution" of each h_{ij} to λ, as the elasticities sum to one (Caswell, 2001; Brault & Caswell, 1993). For example, if the elasticity of a particular h_{ij} element is 0.15, then this element contributes 15 % to λ. However, caution is required when interpreting this analysis. The targeting of this stage with weed management will not result in a 15 % reduction in growth rate, for two reasons. Firstly, if a life cycle stage is set to zero and every individual is required to go through this stage, then the population will become extinct rather than decrease by just 15 % (Caswell, 2001). Secondly, the analysis assumes *ceteris-paribus* (holding every other element in the matrix is constant). In reality, there are interactions between different life

stages. Nevertheless, this analysis can assist in ranking the stages for management, based on their impact on the population's growth rate. The formula to estimate the elasticities (e_{ij}) of elements is defined by Caswell (2001, p.226) as:

$$e_{ij} = \frac{h_{ij}}{\lambda} \frac{\partial \lambda}{\partial h_{ij}} \tag{3.4}$$

To determine the elasticity of a life stage requires the summation of all the elasticities of the elements in that life stage. This type of analysis has also been employed for weed control analysis by Davies (2006), Parker (2000), and Silvertown *et al.* (1993). The elasticity of each life cycle stage c_j is a column addition of e_{ij}, calculated as:

$$c_j = \sum_{i=1}^{i=n} e_{ij} \tag{3.5}$$

This represents the change in the value of λ caused by a proportional marginal change in the individuals in life stage *i*.

The fecundity elasticity has been included in the adult stages, as management of these stages will decrease the population's size as well as its fecundity. This follows the suggestion that the population dynamics of a weed infestation can be greatly modified by removing large plants which produce the most seed (Cacho & Spring, 2004). Parker (2000) employed a similar technique, but the elasticities of fecundity were summed separately to those of other adult parameters. This was done to investigate the effect of biocontrol, by calculating how reduction of seed production may affect λ.

To verify the elasticity-analysis ranking of the stages, the model was run and all the individuals of the different stages were removed, as a once-off management strategy. Although it is unlikely that this can be practically achieved, it does provide an insight into which life cycles stages should be targeted for

the greatest effect. Initially, it was assumed that management was 100 % effective and all targeted individuals were removed.

3.1.4 Measuring damage from woody weeds

It has been proposed that the relationship between weed density and damage can be explained with linear (Jones *et al.*, 2000; Jones & Vere, 1998; Denne, 1988), sigmoidal (Zimdahl, 1980), and rectangular hyperbola functions (Jones, 2003; Cousens, 1985a). Jones *et al*. (2000) assumed a linear weed density-damage relationship for grazing systems but they did not discredit the possibility that it may be a rectangular hyperbola relationship. Research has since shown that these relationships are neither linear nor sigmoidal (Zimdahl, 2004). Historically, rectangular hyperbola damage function models have been based on cropping systems rather than grazing systems (see (O'Donovan *et al.*, 2007; Holst *et al.*, 2007; Fernandez-Quintanilla *et al.*, 2006; Jones, 2003; Cousens, 1985a)). However, Carter *et al*. (1991, p.1) estimated that '*…once a 10% canopy cover of woody species is reached, grass production is severely depressed ….*' They found that a 15–20 % coverage results in a 50 % reduction in herbage production, and a 50 % coverage results in a 100 % loss of herbage production, suggesting a rectangular hyperbola damage function. Based on Cousens (1985a):

$$D_t = \frac{\boldsymbol{\psi}\mathbf{x}_t}{1+\frac{\boldsymbol{\psi}\mathbf{x}_t}{\tau}} \tag{3.6}$$

where D is the damage (proportional loss) to pasture production at time t due to weed damage, $\mathbf{x}_t$ represents both the population and density of weeds ha^{-1}, $\boldsymbol{\psi}$ is a damage index vector for the average proportion of pasture production lost per weed in each life stage (as weed density approaches zero), and τ is the maximum proportion of damage as weed density approaches infinity (Figure 3-2). The implications of this model are (Jones, 2003; Cousens, 1985a):

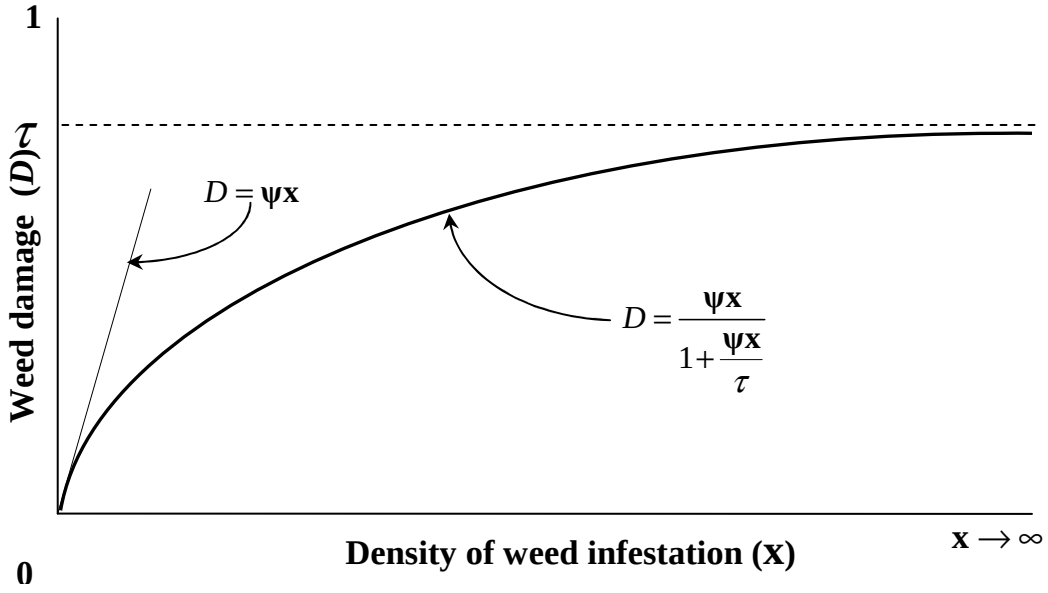


Figure 3-2: The rectangular hyperbola model for the relationship between weed density-damage (after Cousens, 1985a).

When no weeds are present they can have no effect on pasture production, therefore $\mathbf{x}_t = 0 \Leftrightarrow D_t = 0$, and the curve must pass through the origin. For low weed densities, the effect of each additional weed on pasture production is additive because there is little intraspecific competition between the weeds. Therefore, at low weed density the relationship is almost linear, $D_t = \psi \mathbf{x}_t$. As the weed population increases so too does the intraspecific competition between weeds. Each additional weed will increase the level of damage, but at a diminishing rate. Weed damage can never exceed one. As weed density increases it is predicted that the expected damage will tend to an upper asymptotic bound, represented as τ, somewhere between zero and one.

As the population matures, its density will tend to increase due to increased quantity and size of individuals existing in the area. Therefore the average distance between the plants decreases and interference competition between individuals increases. This results in a decreased competitive effect of each plant on pasture production. This effect results from the plasticity of plant form, where an organism changes its form in response to ecological and environmental conditions. Hence, the plant size decreases as crowding increases. This can also be interpreted as the marginal diminishing return from weed damage on pasture production; every new weed will have an impact, but to a lesser degree.

Grazing gross margins of grazing per unit area (π_t), without weed management costs, for a given weed density can be derived from Eq. 3.6 as (Cousens, 1985b):

$$\pi_t = \pi_{wf}\left[1-\frac{\boldsymbol{\psi}\mathbf{x}_t}{\left(1+\frac{\boldsymbol{\psi}\mathbf{x}_t}{\tau}\right)}\right] \tag{3.7}$$

where π_{wf} represents the weed-free gross margins from grazing per hectare. This model implies that when no weeds are present $\mathbf{x}_t = 0 \Leftrightarrow D_t = 0$ resulting in $\pi_t = \pi_{wf}$.

A number of studies have found that weed growth rates in cereals can be suppressed by increasing crop density (Eslami *et al.*, 2006; Lemerle *et al.*, 2004; Walker *et al.*, 2002). Rangeland grazing is largely based on the utilisation of natural resources (Bortolussi *et al.*, 2005a), therefore it was assumed that the density of pasture production is constant and, so too is the impact of the pasture plants on weeds.

3.1.5 Setting model parameters for case study weeds

The population dynamics of weeds are different for riparian and upland zones, as riparian zones contain higher levels of nutrients, moisture, and biodiversity (Lawes *et al.*, 2006; Grice *et al.*, 2000). Therefore, parameters were estimated for both zones. Riparian zones are assumed to be the area within 50m of a river or creek (Lawes *et al.*, 2003).

Most woody-weed studies in the Australian rangelands have been conducted on chinee apple (Grice, 2002), prickly acacia (Radford *et al.*, 2002), parkinsonia (Grice *et al.*, 2004a; Lawes *et al.*, 2003), bellyache bush (Bebawi & Campbell, 2002), mesquite (van Klinken & Burwell, 2005), and rubbervine (Grice, 1997). Surveys on plant height and canopy diameter have been used as surrogates for plant age, to estimate the population structure of an infestation (Lawes & Grice, 2007; Grice *et al.*, 2004a; Downey & Brown, 2000; Watson *et al.*, 1997a). However, there is little information on adult mortality

rates (Stokes *et al.*, 2006). Where no published data existed, expert opinion was used to set parameters and functional relationships.

The process of using expert opinions involved obtaining all the available information from the literature to parameterise the SPM. The parameters that were not available were estimated so that the model outputs were similar those found in the literature (when available). Some of this information would be in grey literature from government departments such as DPI weed guideline booklets. Then results were generated for population structures and size over time, as well as the estimated mean time and variance of time an individual could spend in each life stage (Caswell, 2001). These results were presented to rangeland weed ecologists at CSIRO in Townsville and DPI in Charters Towers. Based on their feedback the models were reparameterised and the outputs made comparable to those in the field and literature. The effects of weed management were then investigated to see how the model responded and to again see if the results were similar to those achieved in experiments, as not all experimental results are published. This was repeated until it was felt by all parties that the model produced realistic outputs. This could take up to ten iterations, to achieve satisfactory results.

The following considerations are used to parameterise stage projection matrices:

- The zones that a species occupies.
- Size and age of a plant at maturity (defined as the point at which it starts to set seed).
- Maximum expected age, size, and fecundity.
- Fecundity of different sized adult plants.
- Germination rates for new seeds and the seedbank.
- Probability of surviving over time.
- Quantity and size (density) of plants in population.

Model results can then be compared to field observations or presented to experts for verification.

Although the model is based on annual increments, the census of fecundity, seedbank and individuals was based on different times of the year. The census for fecundity was done at the end of the fruiting

season while the censuses of the seedbank and individuals in the population were done at the end of the dry season (Parker, 2000).

3.1.5.1 Demographics of chinee apple

Chinee apple can occupy riparian and upland zones (Lawes *et al.*, 2006; Grice *et al.*, 2004a). It generally takes five years to reach maturity and has usually grown to a height of two metres in this time (Grice, 2002; Grice, 1996). There are four juvenile stages in the model (m = 4), with heights for the various stages at <0.2, 0.2–0.4, 0.4–0.7, and 0.7–1.0 m. It can survive for several decades and can grow to 8 m tall (Grice, 2002). The number of mature stages was set at q = 4, with heights of 1–2, 2–3, 3–5, >5 metres. Therefore, the number of life cycle stages was set at n = 10 (1+1+4+4). Once they reach maturity, the plants will produce approximately five seeds per season; this will increase to over 4,000 seeds when the plant is fully matured (Grice, 1998; Grice, 1996). Fecundity was set between 9–4,000 seeds tree^{-1} for different size adults in riparian zones. In glass house experiments, only five per cent of fresh seeds will germinate (Grice, 2002; Grice, 1996). This proportion will increase if seeds are passed through the gut of an animal but will decrease if left under the canopy of the parent tree. In the early stages of the life cycle, chinee apple plants have low survival rates. However, by the time they reach maturity, they are well established and their survival rate increases significantly (Grice, 2002).

Based on information in the literature as well as expert opinion, the demographic parameters and population dynamics of chinee apple infestations were estimated and are presented in Table 3-2 and Table 3-3. Table 3-2 provides the different life cycle stages, height classes, area occupied, and the damage to production, used in Eqs. 3.3 and 3.6. The non-zero elements of the $\mathbf{H}$ matrix in Figure 3-1 are presented in Table 3-3, for riparian and upland zones. The new and steady-state infestation parameters are listed in the $\mathbf{H}_0$ and $\mathbf{H}_\infty$ columns, respectively. With the changes in the probability of survival in riparian and upland zones (Table 3-3), it has been assumed that the demography and damage indices per individual plant do not change between riparian and upland zones (Table 3-2).

Table 3-2: Chinee apple demographic parameters

	Height (m)	Quantity (ha^{-1}) $\mathbf{z}_i$	Area occupied (m^2) $\boldsymbol{\omega}_i$	Proportion of production loss per plant $\boldsymbol{\psi}_i$
New Seeds (*NS*)	0	–	0	0
Seed Bank (*SB*)	0	–	0	0
Juvenile small (J_1)	< 0.2	2 674 311	0.004	0.000 001
Juvenile Medium (J_2)	0.2 – 0.4	136 871	0.073	0.000 027
Juvenile Large (J_3)	0.4 – 0.7	35 368	0.283	0.000 106
Juvenile Largest (J_4)	0.7 -1.0	13 990	0.715	0.000 268
Adults Small (A_1)	1 – 2	4 299	2.326	0.000 873
Adults Medium (A_2)	2 – 3	1 513	6.610	0.002 480
Adults Large (A_3)	3 – 5	584	17.136	0.006 429
Adults Largest (A_4)	> 5	189	52.952	0.019 868

Table 3-3: Chinee apple stage projection elements for riparian and upland zones of new (H_0) and steady-state infestations (H_∞)

		Riparian		Upland	
	Stage Transition	New Infestation $\mathbf{H}_0$	Steady state $\mathbf{H}_\infty$	New Infestation $\mathbf{H}_0$	Steady state $\mathbf{H}_\infty$
New Seeds (*NS*)	M_0	0.004	0.002	0.003	0.002
	M_1	0.600	0.417	0.600	0.484
Seed Bank (*SB*)	R_1	0.600	0.417	0.600	0.484
	M_2	0.004	0.002	0.003	0.002
Juvenile Small (*J*1)	R_2	0.100	0.060	0.100	0.074
	M_3	0.400	0.241	0.200	0.147
Juvenile Medium (J_2)	R_3	0.100	0.070	0.100	0.081
	M_4	0.650	0.452	0.500	0.403
Juvenile Large (*J*3)	R_4	0.100	0.077	0.100	0.086
	M_5	0.800	0.619	0.800	0.690
Juvenile Largest (J_4)	R_5	0.100	0.087	0.100	0.093
	M_6	0.850	0.741	0.850	0.787
Adults Small (A_1)	R_6	0.720	0.649	0.720	0.679
	M_7	0.100	0.090	0.100	0.094
	F_1	9	9	5	5
Adults Medium (A_2)	R_7	0.770	0.715	0.770	0.739
	M_8	0.100	0.093	0.100	0.096
	F_2	880	880	750	750
Adults Large (A_3)	R_8	0.900	0.858	0.900	0.889
	M_9	0.035	0.033	0.035	0.035
	F_3	3 000	3 000	2 000	2 000
Adults Largest (A_4)	R_9	0.960	0.956	0.960	0.958
	F_4	4 000	4 000	3 000	3 000

3.1.5.2 Demographics of prickly acacia

Prickly acacia infests upland and riparian zones (Radford *et al.*, 2001c). These plants can grow higher, denser and produce more seeds in riparian zones than in uplands (Radford *et al.*, 2001c; Carter, 1994). Prickly acacia reaches maturity two to three years after germination (Parsons & Cuthbertson, 2001; Radford *et al.*, 2001c; Mackey, 1998) and reaches a height of approximately two metres in that time (Parsons & Cuthbertson, 2001; Mackey, 1998). Therefore, $m = 2$, with height classes <0.5, and 0.5–2.0 m. Prickly acacia can live up to 60 years (Kriticos *et al.*, 1999a; Mackey, 1998; Carter, 1994), and grow to over 10 m tall (Parsons & Cuthbertson, 2001; Mackey, 1998).

The number of mature stages was set to $q = 4$, with height classes of 2–3, 3–4, 4–5, and >5 m. Therefore the number of life stages is $n = 8$ (1+1+2+4). Reported seed production ranges from 50 to over 200,000 seeds tree^{-1} (Radford *et al.*, 2001a; Radford *et al.*, 2001c; Mackey, 1998; Carter, 1994; Carter *et al.*, 1991). Radford *et al.* (2001b) observed population densities of 103 plants ha^{-1} in riparian zones and predicted 200,000 seeds tree^{-1}. However, they did not mention the size of the trees. Fecundity in riparian zones was set between 150–35,000 seeds tree^{-1} for different adult sizes in the model. Scarification by age, fire, or animal digestion increases germination (Radford *et al.*, 2001a; Mackey, 1998; Bolton *et al.*, 1987). Seed may remain viable for several years (Bolton *et al.*, 1987). However, Carter *et al.* (1991) reported only 20 % of seeds remained viable after 18 months in the field. Of the seeds that germinated, Radford *et al.* (2002) observed 91–100 % of the seedlings died within 12 months. They also found no statistical difference between seedling morality rates in upland and riparian zones. Nevertheless, drought conditions can cause severe reductions in prickly acacia populations (Carter & Cowan, 1993) and it is possible that the mortality rates for upland seedlings is higher. Mortality tends to decrease as prickly acacia increases in height (Radford *et al.*, 2002). Radford *et al.* (2001c) reported average tree densities between 207–2,665 ha^{-1} for riparian zones. They did not indicate whether the populations were at steady-state.

Based on information in the literature, as well as expert opinion, the demographic parameters and population dynamics are estimated for prickly acacia infestations. These are presented in Table 3-4 and

Table 3-5. Table 3-4 provides the different life cycle stages, height classes, area occupied, and the damage to production, used in Eqs. 3.3 and 3.6. The non-zero elements of the **H** matrix in Figure 3-1 are presented in Table 3-5, for riparian and upland zones. The new and steady-state infestation parameters are listed in the $\mathbf{H}_0$ and $\mathbf{H}_\infty$ columns, respectively.

Table 3-4: Prickly acacia demographic parameters

	Height (m)	Quantity (ha^{-1}) $\mathbf{z}_i$	Area occupied (m^2) $\boldsymbol{\omega}_i$	Proportion of production loss per plant $\boldsymbol{\psi}_i$
New Seeds (*NS*)	0	–	0	0
Seed Bank (*SB*)	0	–	0	0
Juvenile Small (J_1)	< 0.5	318 310	0.031	0.000 013
Juvenile Medium (J_2)	0.5 – 2.0	8 842	1.131	0.000 477
Adults Small (A_1)	2 – 3	2 121	4.714	0.001 990
Adults Medium (A_2)	3 – 4	1 070	9.348	0.003 946
Adults Large (A_3)	4 – 5	643	15.553	0.006 565
Adults Largest (A_4)	> 5	264	37.937	0.016 013

Table 3-5: Prickly acacia stage projection elements for riparian and upland zones of new (H_0) and steady-state infestations (H_∞)

		Riparian		Upland	
	Stage Transition	New Infestation $\mathbf{H}_0$	Steady state $\mathbf{H}_\infty$	New Infestation $\mathbf{H}_0$	Steady state $\mathbf{H}_\infty$
New Seeds (*NS*)	M_0	0.100	0.013	0.020	0.005
	M_1	0.370	0.340	0.370	0.357
Seed Bank (*SB*)	R_1	0.370	0.340	0.370	0.357
	M_2	0.100	0.013	0.020	0.005
Juvenile Small (J_1)	R_2	0.060	0.015	0.050	0.022
	M_3	0.150	0.036	0.080	0.035
Juvenile Largest (J_2)	R_3	0.150	0.051	0.100	0.055
	M_4	0.500	0.168	0.500	0.220
Adults Small (A_1)	R_4	0.720	0.505	0.720	0.612
	M_5	0.150	0.073	0.150	0.105
	F_1	150	150	75	75
Adults Medium (A_2)	R_5	0.760	0.662	0.760	0.716
	M_6	0.150	0.063	0.150	0.095
	F_2	7 000	7 000	3 500	3 500
Adults Large (A_3)	R_6	0.835	0.727	0.835	0.787
	M_7	0.100	0.042	0.100	0.064
	F_3	27 000	27 000	13 500	13 500
Adults Largest (A_4)	R_7	0.910	0.907	0.910	0.909
	F_3	35 000	35 000	17 500	17 500

3.1.5.3 Demographics of parkinsonia

Parkinsonia is found in upland and riparian zones. However, Lawes *et al.* (2007; 2003) estimated 80 % of the population is close (< 1 km) to permanent water. Parkinsonia can produce seeds 2–3 years after germination (Anon, 2000), when they are around 1.9 metres tall (Anderson *et al*. in press), therefore m = 2, with height classes of <0.5, and 0.5–2.0 m tall. Parkinsonia can grow to 8 m tall (Vitelli & Madigan, 2004; Cochard, 1999) and adults are grouped into three stages, q = 3, with heights of 2–3, 3–5, >5 metres. Therefore the number of life cycle stages is n = 7 (1+1+2+3). Seed production is about 10 seeds for a small adult and rapidly increases to over 13,000 for the largest adults (Anderson *et al*. in press). Fecundity was set at 1,000–10,000 seeds tree^{-1} in riparian zones. Seeds can remain dormant for up to 50 years (Anon, 2000). Seedlings have high mortality rates in the rangeland system (Lawes, 2004). Parkinsonia is semi-drought tolerant, surviving long periods of very low moisture in the soil (Lawes, Abbott, and Whiteman in press). Anderson *et al.* reported areas with an average of 9,000 plants ha^{-1}, in northern Queensland. It was expected that mass recruitment of seedlings would occur after clearing dense infestations, due to decreased competition and large seedbanks (Cochard & Jackes, 2005). Grice *et al.* (2002; 1996) estimated the relationship between the plant's canopy diameter and its height as canopy diameter (cm) = 1.18 height (cm) – 4.9.

Based on information in the literature as well as expert opinion, the demographic parameters and population dynamics are estimated for prickly acacia infestations were estimated. These are presented in Table 3-6 and Table 3-7. Table 3-6 provides the different life cycle stages, height classes, area occupied, and the damage to production, used in Eqs. 3.3 and 3.6. The non-zero elements of the **H** matrix in Figure 3-1 are presented in Table 3-7, for riparian and upland zones. The new and steady-state infestation parameters are listed in the $\mathbf{H}_0$ and $\mathbf{H}_\infty$ columns respectively.

Table 3-6: Parkinsonia demographic parameters

	Height (m)	Quantity (ha^{-1}) $\mathbf{z}_i$	Area occupied (m^2) $\boldsymbol{\omega}_i$	Proportion of production loss per plant $\boldsymbol{\psi}_i$
New Seeds (*NS*)	0	–	0	0
Seed Bank (*SB*)	0	–	0	0
Juvenile Small (J_1)	< 0.5	203 718	0.049	0.000 023
Juvenile Medium (J_2)	0.5 – 2.0	25 800	0.388	0.000 182
Adults Small (A_1)	2 – 3	2 986	3.349	0.001 571
Adults Medium (A_2)	3 – 5	930	10.752	0.005 043
Adults Large (A_3)	> 5	368	27.155	0.012 736

Table 3-7: Parkinsonia stage projection elements for riparian and upland zones of new (H_0) and steady-state infestations (H_∞)

		Riparian		Upland	
	Stage Transition	New Infestation $\mathbf{H}_0$	Steady state $\mathbf{H}_\infty$	New Infestation $\mathbf{H}_0$	Steady state $\mathbf{H}_\infty$
New Seeds (*NS*)	M_0	0.030	0.005	0.010	0.003
	M_1	0.600	0.531	0.600	0.566
Seed Bank (*SB*)	R_1	0.600	0.531	0.600	0.566
	M_2	0.030	0.005	0.100	0.003
Juvenile Small (J_1)	R_2	0.350	0.094	0.200	0.089
	M_3	0.350	0.094	0.200	0.089
Juvenile Largest (J_2)	R_3	0.350	0.129	0.300	0.168
	M_4	0.450	0.166	0.350	0.196
Adults Small (A_1)	R_4	0.600	0.438	0.600	0.513
	M_5	0.200	0.105	0.200	0.141
	F_1	1 000	1 000	750	750
Adults Medium (A_2)	R_5	0.720	0.638	0.720	0.680
	M_6	0.200	0.091	0.200	0.129
	F_2	5 000	5 000	4 000	4 000
Adults Largest (A_3)	R_6	0.500	0.499	0.500	0.499
	F_3	10 000	10 000	7 500	7 500

3.1.6 Modelling woody-weed control

The population model is expanded to include weed control decisions:

$$\mathbf{x}_{t+1} = (\mathbf{H}_t \mathbf{x}_t) - (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_t}) \tag{3.8}$$

where u_t is the control decision at time *t*, represented by an element from vector **U,** of possible management actions (which is assumed to be either no-control, burning, chemical, or mechanical removal of plants). The mortality rate for each control decisions $(\boldsymbol{\varphi}_u)$ on each life-cycle stage is represented in the efficacy matrix $\boldsymbol{\Phi}$, being *n* x *w* in dimension, where *w* is the number of available control methods, in this case four control options are used. The values of the elements within $\boldsymbol{\varphi}$ are between zero and one. They represent the mortality rate, above natural mortality rates, with respect to the effect on each respective life-cycle stage. i.e. if herbicide resulted in 90 % of largest adults dying, then the last element in $\boldsymbol{\varphi}$ has value of 0.9. As herbicide has no effect on seeds, the first two elements of $\boldsymbol{\varphi}$ will have a value of zero. Management of the infestation occurs at the end of the period but prior to the following season (*t*+1). If no weed control is undertaken, then $\boldsymbol{\varphi}$ equates to a vector of zeros and the model is reduced to $\mathbf{x}_{t+1} = (\mathbf{H}_t \mathbf{x}_t)$. It was assumed in the following simulation that weed management is able to remove 100 % of the targeted individuals.

3.1.7 Modelling biocontrol of woody weeds

Biocontrol must be treated separately from other control methods, as its effect often remains for the duration of the infestation and its efficacy does not diminish over time. Once the decision to use biocontrol has been made, it cannot be reversed. It can be used in combination with other control methods such as burning, chemical, or mechanical removal of plants as part of an integrated weed management (IWM) strategy. The efficacy of biocontrol such as seed-eaters can be included within the **H** transition matrix, i.e. if 20 % of all new seeds perish due to the presence of the biocontrol agent. The

element for new-seeds (*F*) within **H** will therefore be reduced by 20 %. The biocontrol population model is:

$$\mathbf{x}_{t+1} = (\mathbf{H}_t^{bio}\mathbf{x}_t) - (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_t}) \tag{3.9}$$

In this way, while both u_t and biocontrol can be used as part of an IWM strategy, their effects and decision of biocontrol and u_t remain separate.

3.2 RESULTS

Using the parameters of the three plant species in riparian and upland zones (Table 3-2 to Table 3-7); predictions were generated on a per hectare basis in the northern rangeland landscape. All the simulations were started with a population of 2000 seeds at t = 0. Results from these simulations assisted in determining which management strategies are most likely to be effective for the different case-study weeds within riparian and upland zones.

3.2.1 Population simulations

The times required by the different species in the different zones to reach steady-state in the absence of management, as well as population densities, and weed damages are presented in Figure 3-3. Chinee apple takes approximately150 years in upland zones to reach steady-state. Parkinsonia is the fastest species to occupy the available area, requiring only 25 years in riparian zones. The “area occupied” lines in Figure 3-3 represents weed densities, which are functions of the quantity and size of individuals in the populations (W_t in Eq. 3.2). The impact of weeds on beef production are represented by the “Damage” lines in Figure 3-3 and they are defined by Eq. 3.6. Both area occupied and damage exhibited sigmoidal population growth functions. All three species resulted in approximately 70–83 % weed damage in riparian zones and 30–45 % loss in upland zones, when at steady-state. The main difference among species is the timing of the impact on graziers, with prickly acacia and parkinsonia

both having an impact four times sooner than chinee apple in riparian and upland zones (Figure 3-3). However, chinee apple has the greatest impact on pasture production, with decreases of 83 and 45 % in riparian and upland zones, respectively.

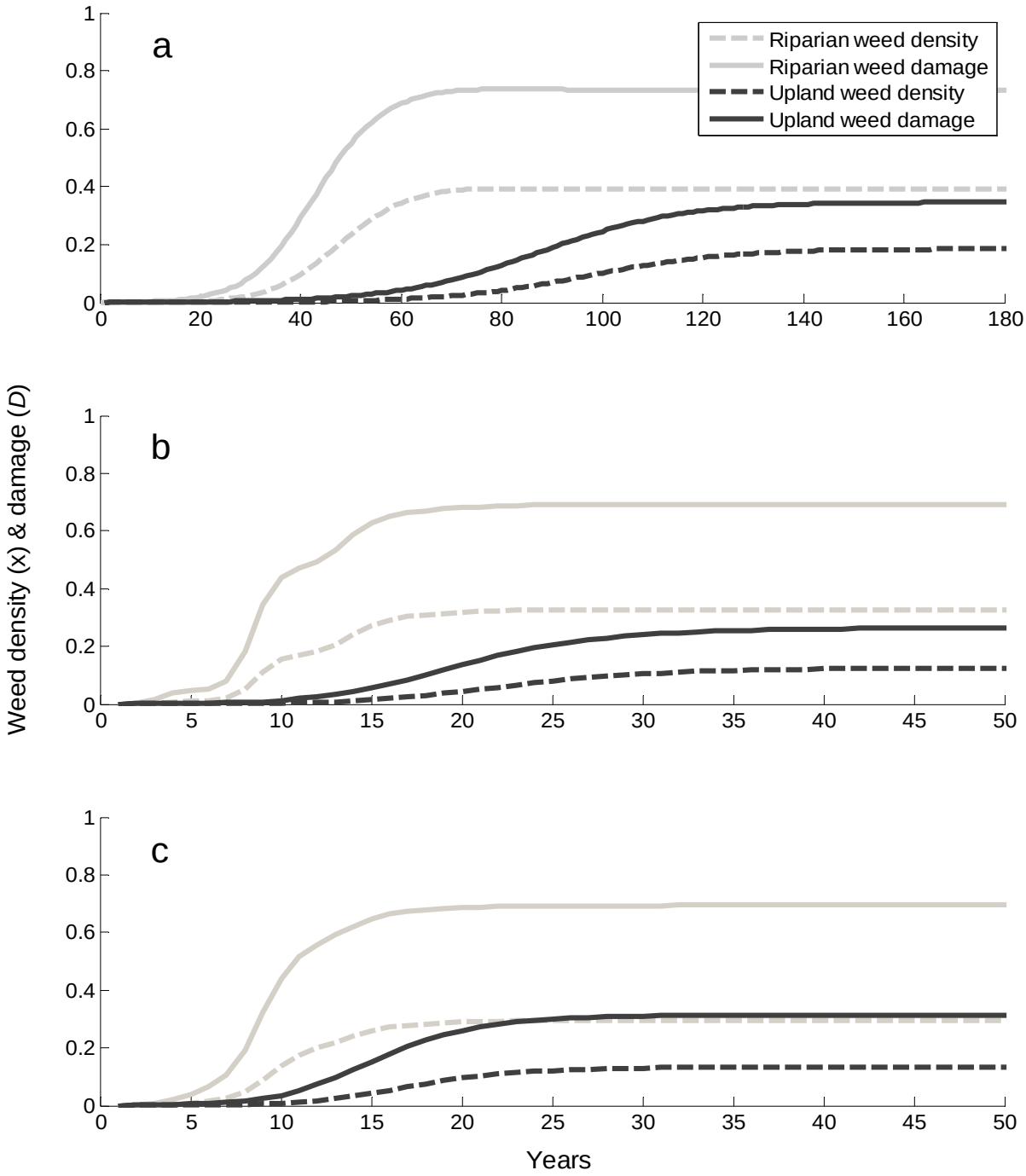


Figure 3-3: Temporal simulations of weed density (X) and damage (D) without any control for (a) chinee apple, (b) prickly acacia, and (c) parkinsonia infestations, in both riparian and upland zones. Note different scales in horizontal axes for chinee apple to enhance readability.

3.2.2 Relationship between weed damage and density

Weed damage as a function of weed densities (Eq. 3.6) (being the level of crowdedness) are presented in Figure 3-4 for the three species. As density increases, weed damage increases at a decreasing rate towards an asymptote. The maximum densities and damage shown in Figure 3-4 are determined by the population asymptotes in Figure 3-3.

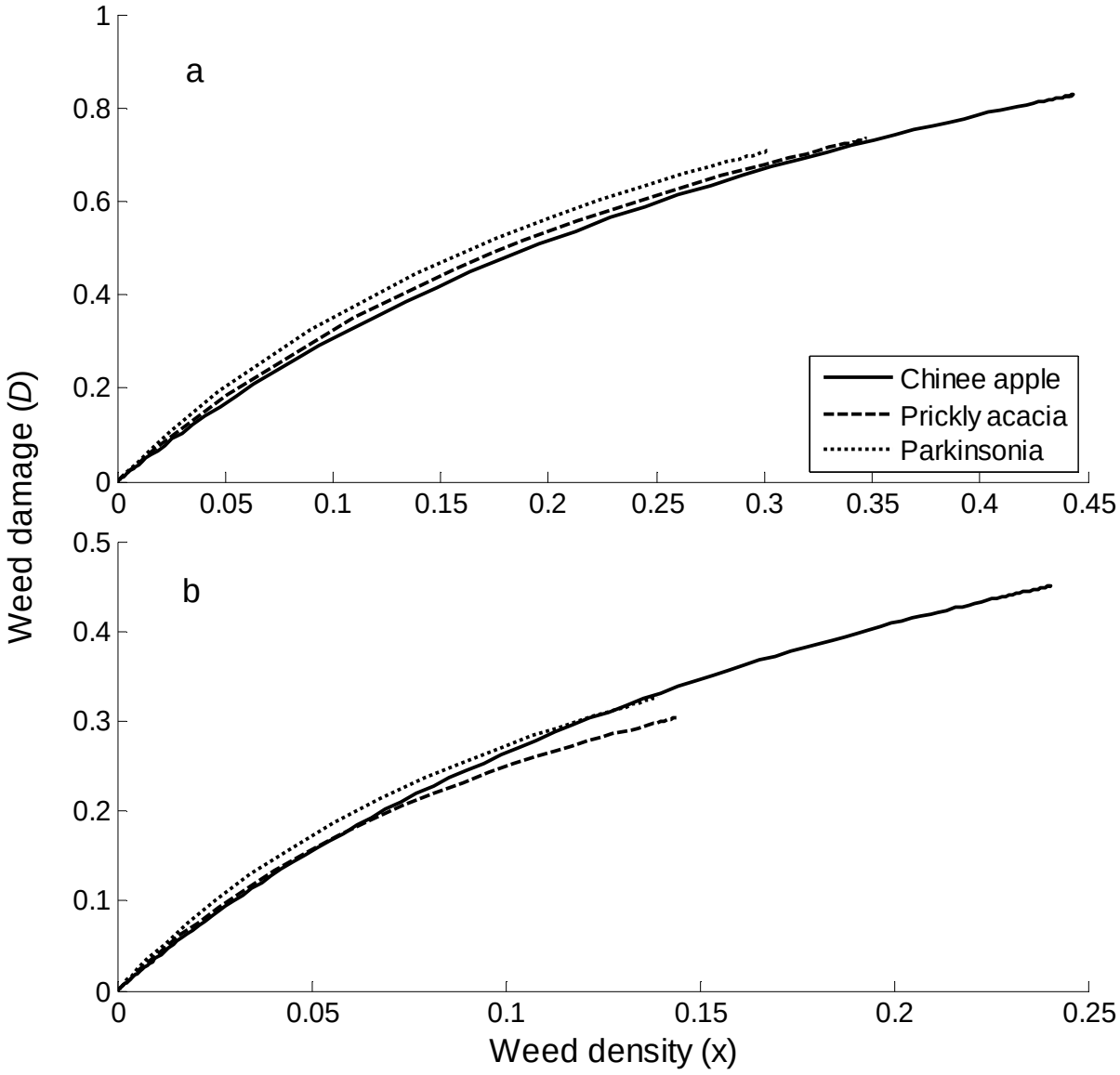


Figure 3-4: The relationship between weed density (x) and weed damage (D) for chinee apple, prickly acacia, and parkinsonia infestations, in (a) riparian and (b) upland zones.

3.2.3 Time and variance of life stages

Caswell (2001) illustrates a method for evaluating stage projection matrices to predict the mean and variance of the time individuals will spend in each stage of the life cycle. This method was used to test whether the model predictions for the three species conform to expected results based on demography of the plants in riparian and upland zones (Figure 3-5). Tests were conducted for new and steady-state

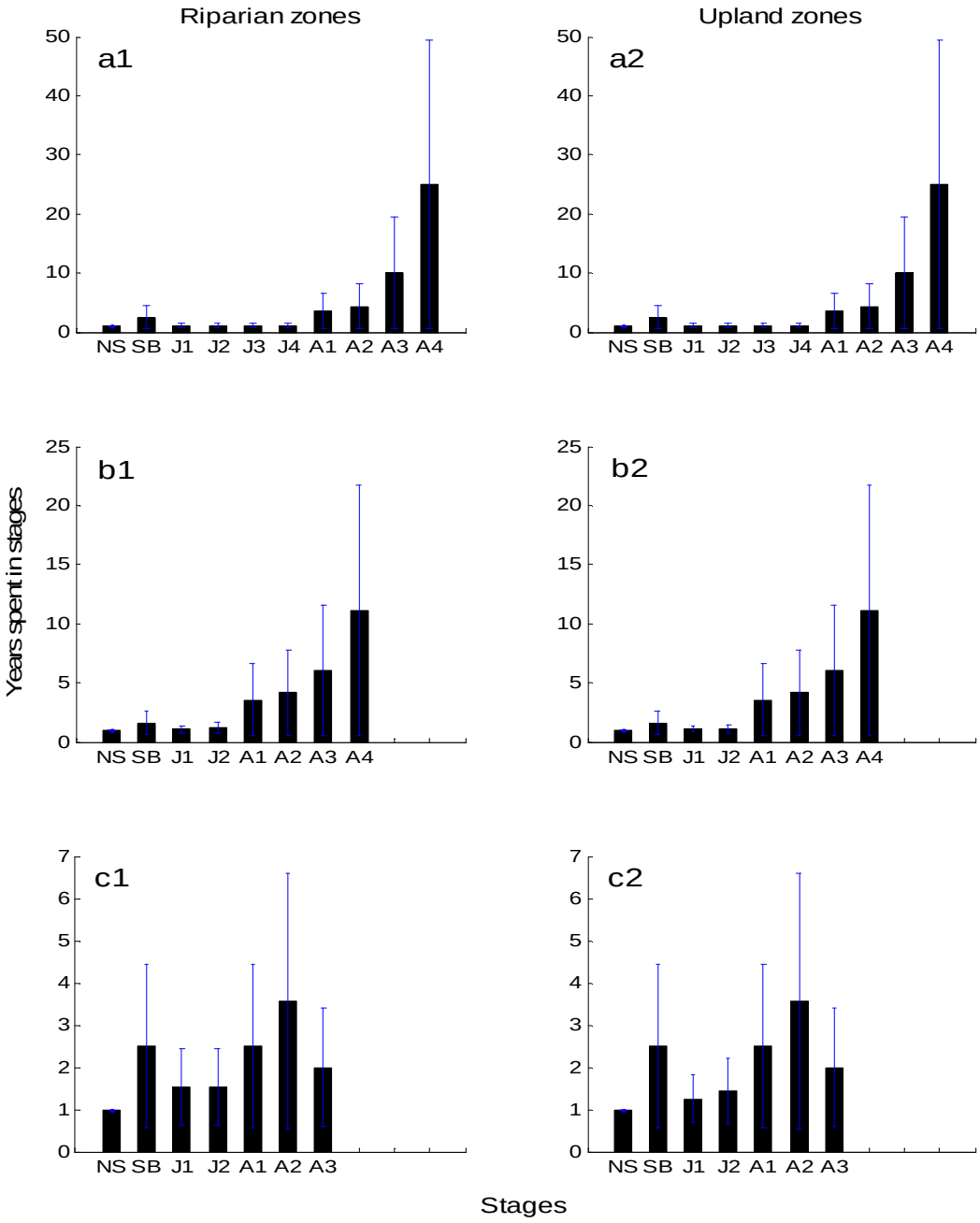


Figure 3-5: Estimated mean (bars) and standard deviation (lines) of the time spent in each life stage for (a) chinee apple, (b) prickly acacia, and (c) parkinsonia infestations, in (1) riparian and (2) upland zones

infestations and no observable difference between them could be detected. The analyses presented in Figure 3-5 were based on new infestations, as this represents the life expectancy of an individual plant in the landscape before density-dependence sets in. The analysis indicates that chinee apple is the longest lived species, living to approximately 47 years, assuming it reaches the final life stage. Life expectancy for prickly acacia and parkinsonia plants is approximately 26 and 11 years, respectively. There are little differences between the life expectancy of individuals living in riparian and upland zones (Figure 3-5). This is because the analysis represents only the average time spent in each life stage, not the probability of progressing through the stages and reaching maturity.

3.2.4 Population structure

The estimated steady-state population structures vary for the three different species (Figure 3-6). Prickly acacia and parkinsonia have fewer stages in their life cycle than chinee apple, as indicated by the missing columns in the graphs. Prickly acacia has the greatest number of juveniles (19,624 ha^{-1} and 3,826 ha^{-1} for riparian and upland zones, respectively), chinee apple has the least (2,117 ha^{-1} and 968 ha^{-1} for riparian and upland zones, respectively), and parkinsonia is in-between (9,720 ha^{-1} in the riparian zones and 2,711 ha^{-1} in upland zones). Chinee apple has more adults (738 and 191 for riparian and upland zones, respectively), than parkinsonia (377 in the riparian zones and 163 upland zones) and lastly prickly acacia (318 and 128 in the riparian and upland zones, respectively).

The number of smallest chinee apple adults (A1) is greater than the number of largest juveniles (J4), (Figure 3-6 (a)). This is possible because it only takes one year for an individual to progress through any juvenile stage, yet it may remain for a number of years in any adult stage, as reflected in Figure 3-5. Additionally, chinee apple plants will spend most of their lives in the largest adults stage (A4), if they survive to that point. These results indicate that the parameters used in the stage matrices are plausible, as they reflect the expected time length of each life stage for the three species based on literature data and expert opinions.

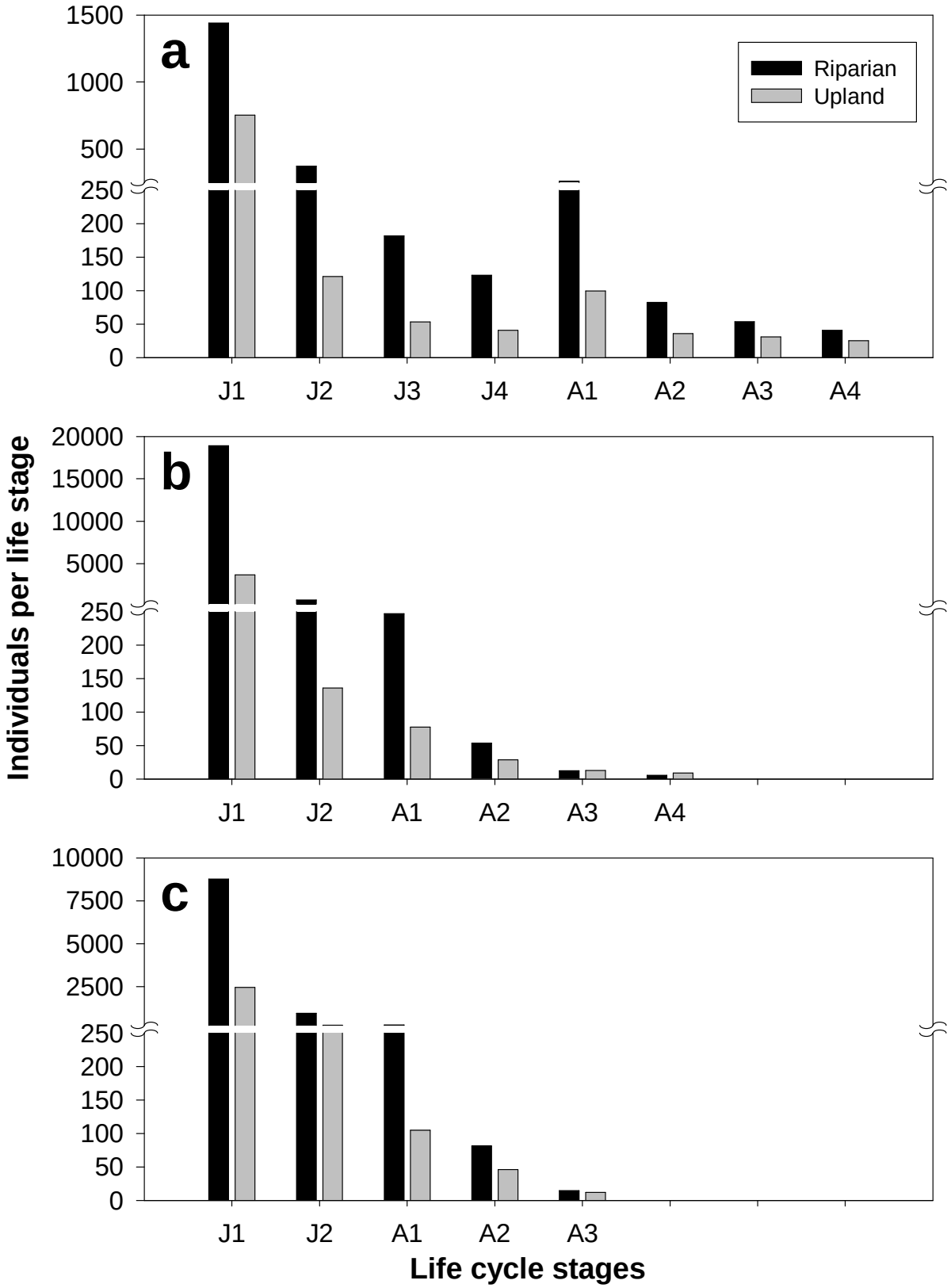


Figure 3-6: Estimated number of individual weeds ha^{-1} in juvenile and adult stages of steady-state infestations of (a) chinee apple, (b) prickly acacia, and (c) parkinsonia, in riparian and upland zones.

3.2.5 Elasticities

Population growth rates and corresponding elasticities for the three species are presented in Figure 3-7 for new and steady-state infestations in both riparian and upland zones. The growth rates of steady-state infestations are by definition equal to zero $(\ln\lambda = r = 0)$, with corresponding λ = 1.0. Parkinsonia has the highest annual growth rate for new infestations in both riparian and upland zones, λ = 2.196 and λ = 1.538. New infestations of prickly acacia have a similar annual growth rates λ = 2.116 and λ = 1.403 for riparian and upland zones. Chinee apple has the slowest annual growth rates λ = 1.184 and λ = 1.086 for riparian and upland zones.

The elasticities of the different life stages reflect their contribution to the population growth rate (Figure 3-7) and therefore they can be used to rank the stages for management purposes. For example, medium sized adults (A2) should be targeted first for new infestations of chinee apple in riparian zones, and large adults (A3) in upland zones. Once the population matures to a steady-state, then the largest adults (A4) should be targeted for both zones. Similar trends exist for the other species (Figure 3-7).

The elasticities for fecundity are included in Figure 3-7 to investigate the potential effect of a biocontrol agent. Controlling new seed (fecundity) has a less affect on the longer lived plant (chinee apple). However, caution is required in ranking this management option in combination with other management strategies. The targeting of fecundity via biocontrol agents and removal of adult stages would result in double counting of the effects from reduced fecundity on population growth, if both the elasticities of A_i and F (Figure 3-7) were used.

The following sections are based on a steady-state chinee apple infestation in a riparian zone as a case study for evaluating management strategies of targeting different life stages. According to the results in Figure 3-7 (a1), the ranking of stages for the management of a steady-state population is A4, A3, A2, A1 then any of the juvenile stages (as they all have the same weighting) and lastly seeds.

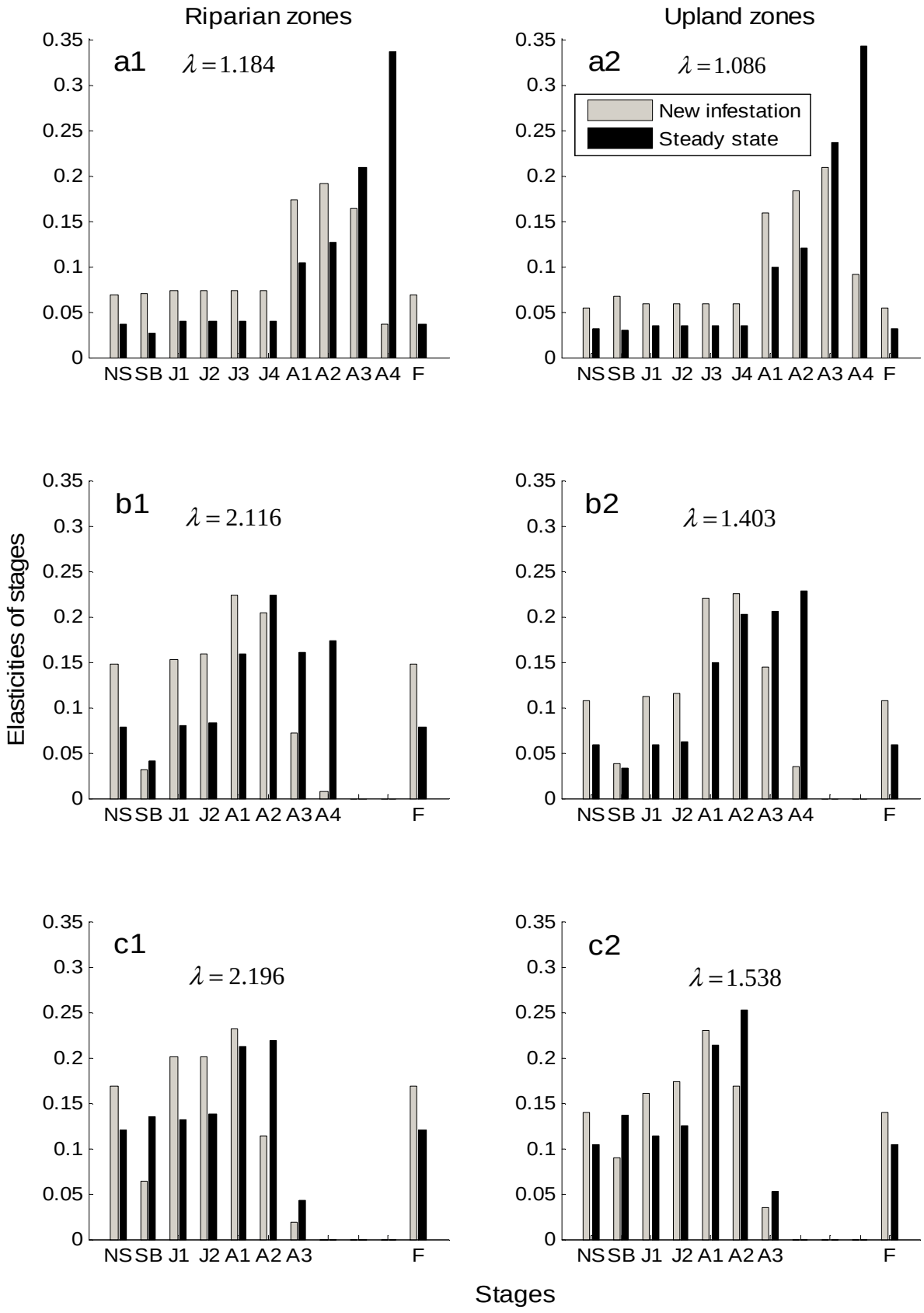


Figure 3-7: Estimated elasticities of the different life stages in new and steady-state infestations of (a) chinee apple, (b) prickly acacia, and (c) parkinsonia, in (1) riparian and (2) upland zones. Growth rate (λ) values are only given for new infestation, as steady-state infestations by definition have λ values of 1.0.

3.2.6 Management of life stages

To determine the effectiveness of targeting different life stages, a population simulation is run and a single kill of all the plants in a given life stage is undertaken. Each simulation starts from a steady-state population of chinee apple in a riparian zone.

In Figure 3-8 the area between the “no management” line and the various “weed damage” lines is the benefit gained from targeting the various life cycle-stages. The benefits of managing a weed infestation are given not only by the magnitude in the change of weed damage but also the duration of the reduction. The greater the difference between these two lines, the greater the benefit of management. These ecological analyses are not discounted for the time value of benefits, as this is undertaken in Chapters 6 and 7 in the economic optimisation of woody-weed management. The graph has been split into sections (a) and (b) to aid in comparing the different life stages. Targeting the largest adults (A4) will result in weed damage decreasing from 82 % to 56 % in the following year (Figure 3-8 (a)), whereas targeting largest juveniles (J4) will require two years to reach the maximum benefit, while weed damage is only reduced to 80 %. Figure 3-8 indicates that the targeting of adults will result in greater and faster impact on damages than targeting any other stage. The new seeds (NS), seedbank (*SB*), and fecundity (*F*) stages have the lowest elasticities and are therefore ranked the lowest for weed management. Note that the new seeds (*NS*) stage cannot be seen in Figure 3-8 (a) as it has exactly the same result as targeting fecundity (*F*), due to new seeds being a function of fecundity. The rankings of life stages using this analysis (Figure 3-8) corresponds exactly to the elasticity analysis (Figure 3-7), A4, A3, A2, A1, juveniles and lastly seeds, so these results provide another form of model validation.

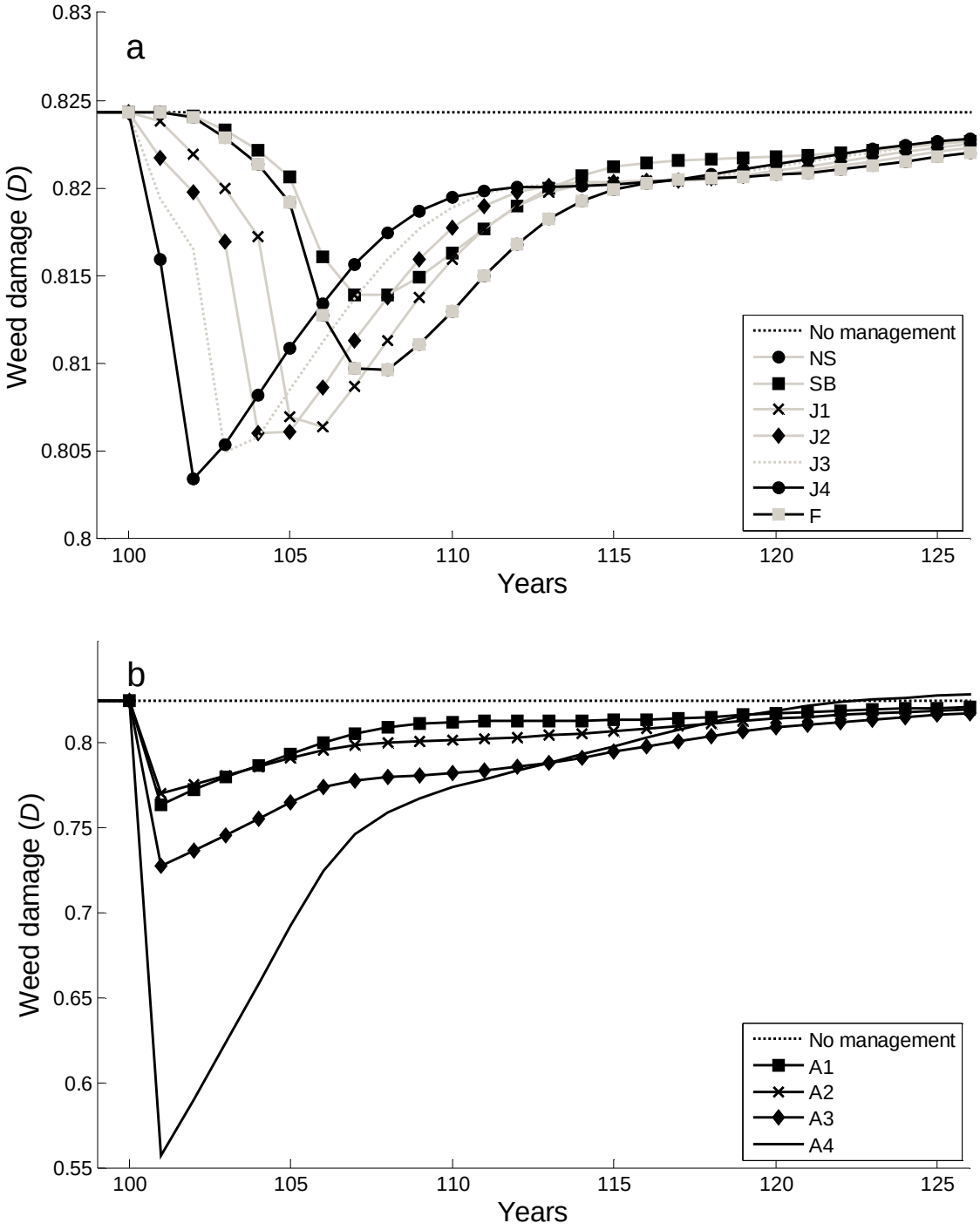


Figure 3-8: Estimated changes to the proportion of grazing loss for a once-off management of different life stages. Simulations start from a steady-state (starting at 100 years) infestation of chinee apple in a riparian zone. (a) represents the removal of non-adult stages, (b) represents the removal of the adult stages; note scale difference in graphs.

3.2.7 Periodic management of woody weeds

If all the adults stages are removed from a steady-state population of chinee apple in riparian zones, grazing pasture production will increase from 19 (1–0.81) % to 93 (1–0.07) % in the year following

management (Figure 3-9)[8]. As before, the total benefit of the single management is the area between the "no management" and the "once only" lines. Benefits of this single management will continue for up to 25 years although they will diminish as the plant repopulates the area. An alternative management strategy is to periodically manage the infestation. For this example (Figure 3-9), if the infestation were managed every 10 years it would result in a continued increase in pasture production compared with non-management but will not result in eradication. Alternatively, eradication could be attempted but this will involve a more frequent management strategy. Eradication is possible if all the adults are removed every 3 years for approximately 25 years (Figure 3-9), assuming there is no immigration of seeds into the area.

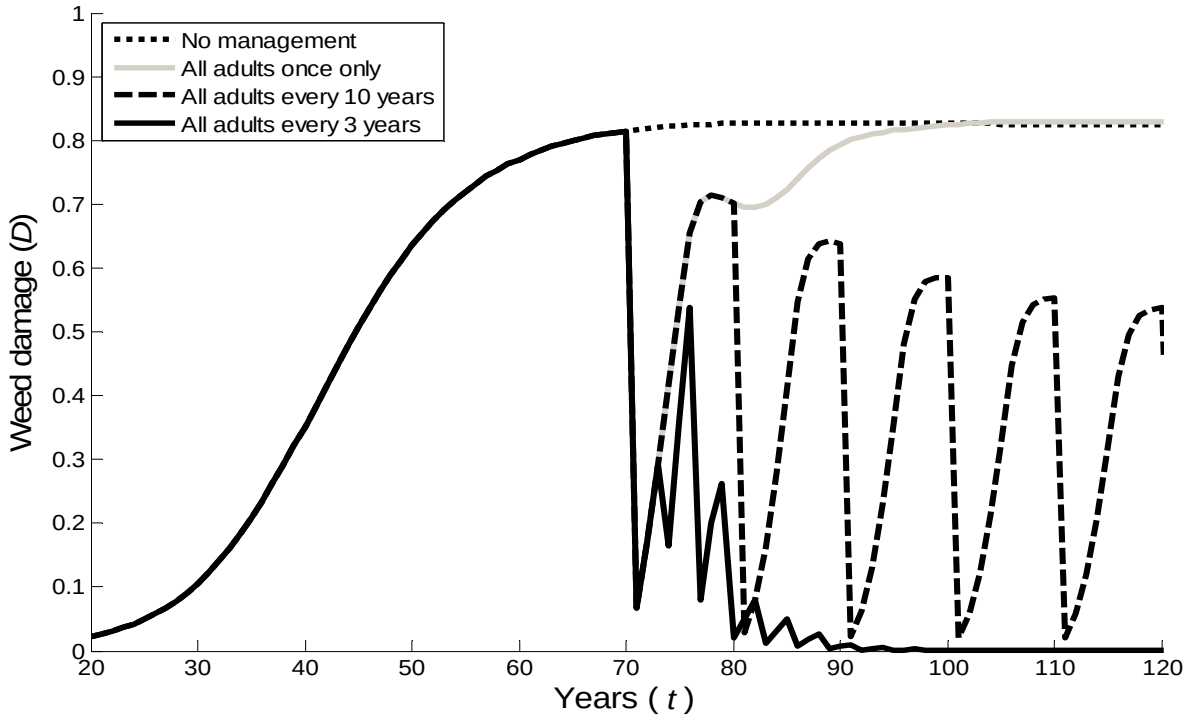


Figure 3-9: Temporal simulations of changes in pasture production with the removal of all chinee apple adults (riparian zone), once only, every 10 years, and every 3 years.

[8] This assumes that the time lags for pasture species to re-occupy the controlled area are shorter than a year, so they are captured within the annual time step. Part of weed damage is due to areas being locked up due to weed thickets. By removing the weeds, these areas will be instantly available to cattle, while other areas will take longer to recover. At this stage it is unknown what time lag, if any, is needed in the model.

3.2.8 Determining the potential of biocontrol

Biological agents such as seed-eating insects will not remove 100 % of the seeds. Similar to Parker (2000), the fecundity values were manipulated to establish hypothetical λ values to evaluate the potential of biocontrol agents. An alternative analysis is also presented where changes in weed damage are estimated for the manipulated fecundity values. These two analyses of measuring the potential benefit of biocontrol will be contrasted.

It was assumed that all infestations were at steady-state, i.e. chinee apple (t = 150), prickly acacia (t = 50), and parkinsonia (t = 50) in both riparian and upland zones. Fecundity was reduced by a given per cent and the model was run for 100 years. The λ values and weed damage for the final year of the simulation were recorded. This process was repeated with reductions in fecundity being between zero and 100 %, at intervals of one per cent. The results from these simulations are presented in Figure 3-10.

If growth rates do not change (i.e. λ remains at 1.0), it can be concluded that the level of biocontrol tested is ineffective. In all species in both riparian and upland zones, seed-eating biocontrol agents must remove over 95 % of the seeds produced to have an effect on population growth rates (lambda), with the exception of chinee apple in upland zones which requires 85 % reduction in seed production to affect population growth (Figure 3-10). Once lambda is < 1.0, the population will tend towards extinction.

An alternative approach is to investigate changes in weed damage with respect to a reduction in fecundity (Figure 3-10). In all the simulations, the decline in weed damage with respect to reduced fecundity is slow and relatively linear until fecundity is reduced by 80 %. After this point there is a sharp decline in weed damage with respect to fecundity reduction. In all instances a 100 % reduction in fecundity results in zero weed damage, implying no weeds are present in the area at the end of the simulation.

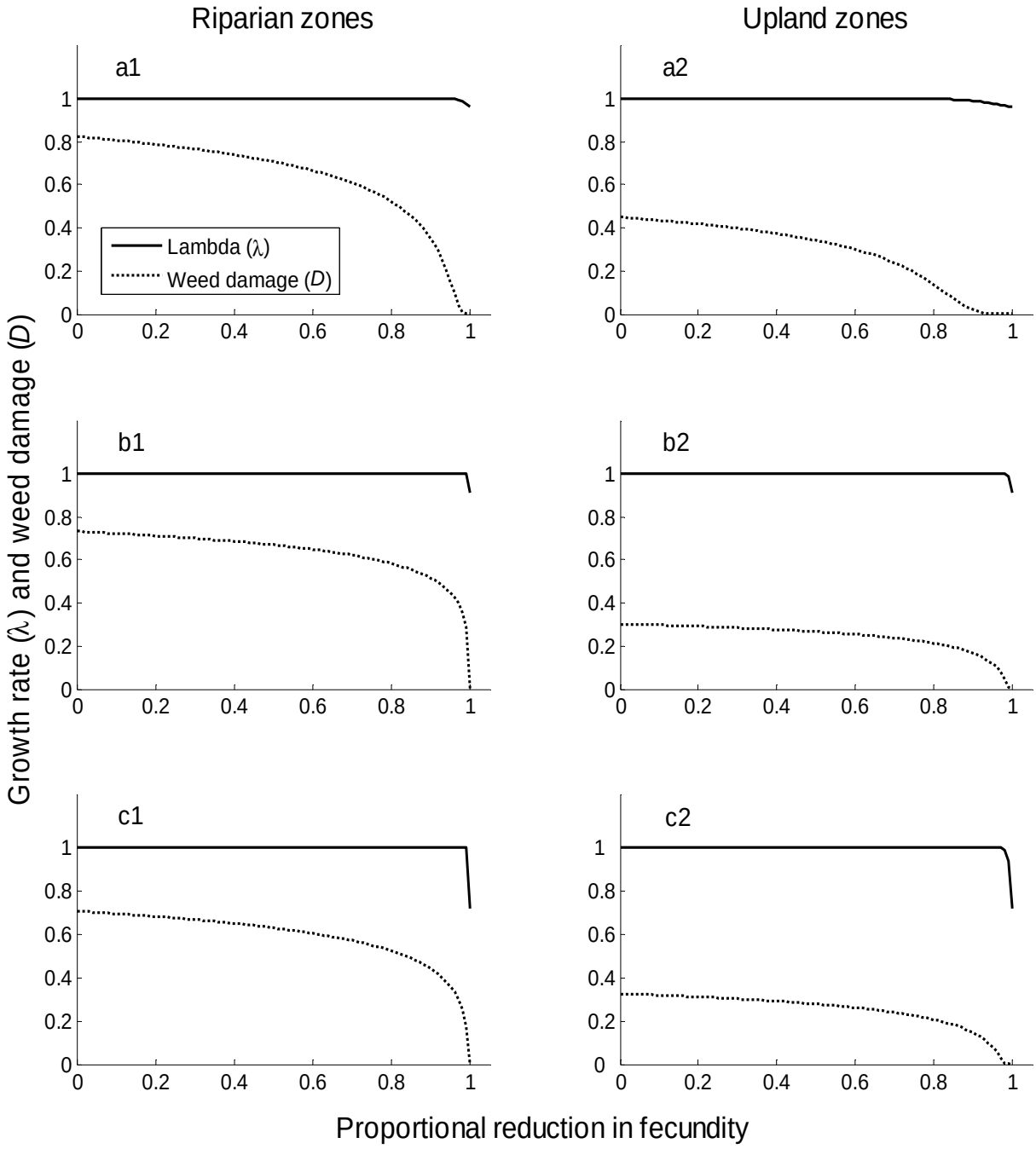


Figure 3-10: Changes in growth rage (λ) and weed damage (D) in the final year of the simulation of managing steady-state populations with respect to proportional reduction of fecundity over a 100 year period, (a) chinee apple, (b) prickly acacia, and (c) parkinsonia, in (1) riparian and (2) upland zones.

3.3 DISCUSSION

3.3.1 Temporal benefits of woody-weed control

Simulation results highlight the temporal benefits of weed management. This is important because the frequency of management is partially dependent on the grazier's goal. For example, the grazier may see the management as a once-off investment that can have benefits for up to 25 years. Alternatively, the grazier may seek to remove plants as an ongoing management strategy to mitigate the weed's cost. Alternatively, the grazier may be seeking to eradicate the infestation in the most effective manner, which may not be possible in a single year due to the survival of the seedbank and undetectable seedlings. A necessary condition for eradication is not to allow the weeds to reach maturity and produce seeds, while the seedbank is depleted over time due to mortality of seeds and germination.

Woody weeds often live for decades and have low recruitment rates (Radford *et al.*, 2002; Grice, 1998), resulting in some infestations requiring in excess of a century to reach a steady-state. This can hinder short-term experimental approaches that attempt to capture population dynamics because long-term studies are required. These long-term experimental approaches can be costly. Their interpretation is also complicated by climate, edaphic and biotic factors which hinder the differentiation between population dynamics (including density-dependence processes) and environmental 'noise' (Cousens, 1995). Although modelling can assist in determining the future behaviour of weed infestations, their impact on pasture production and the efficacy of different management strategies, it is only as good as the input data used. The use of modelling and experimental methods should not be thought of as mutually exclusive. Rather, both tools should be used in conjunction to answer ecological problems. The model developed in this chapter offers insights into which population dynamic parameters are most sensitive to change, contingent on the given state of the population at any time.

3.3.2 Utilising this modelling framework

Modelling approaches undertaken in this study can be used as a framework for predicting the density and structure of other woody weed species, aiding in determining the efficacy of various weed-management practices. In this study, the population dynamics of three woody weeds: chinee apple, prickly acacia and parkinsonia, in both riparian and upland zones, were investigated using stage-projection matrices. These were explicitly constructed to investigate the impacts of weed dynamics on individual graziers. Grice and Maywald (2003) also constructed a population model for chinee apple bit it did not explicitly account for density-dependence, resulting in exponential population growth. They stated that it was a "…*very simple deterministic model* …. [to] *facilitate the exploration of ideas about efficient ways to deal with long-lived invasive rangeland shrubs*." (Grice & Maywald, 2003, p.352). Their modelling approach did not determine the size or density of individual infestations or the associated economic costs imposed on individual graziers. Another significant difference between their model and the model presented here is the spatial scale used. Their model investigates what might happen on a regional scale, whereas the model presented here is based on a per hectare basis. Although woody weeds can infest wide regions, their management is often undertaken at a smaller scale, such as at the field level. The average per-hectare measurement here is more in line with the information required by graziers to make their weed-management decisions.

In order to determine when and to what degree a weed population will affect grazing pasture production, knowledge is required of the weed's steady-state occupancy, colonisation rates and its density-damage ratio. According to Carter *et al*. (1991) grass production can be significantly suppressed by woody weeds. They reported a 20 % canopy cover of an area will result in a 50 % reduction in pasture production, and a 50 % canopy cover results in a 100 % loss of pasture production. Figure 3-4 adequately captures this relationship, with a 20 % canopy cover of chinee apple, prickly acacia, and parkinsonia, resulting in 51, 53, and 56 % weed damage respectively.

All three species have similar life stages. However, their population dynamics result in different colonisation rates, life expectancy and population structures. Parkinsonia is predicted to be the faster coloniser of an area, closely followed by prickly acacia (Figure 3-3). Both of these species are predicted to occupy the area approximately four times faster, resulting in production weed damage sooner than chinee apple. These results are in line with the eigenvalues (λ) given in Figure 3-7, where parkinsonia has the highest and chinee apple has the lowest values. Once the population reached steady-state however, chinee apple infestations resulted in higher damage than the other species. Life expectancy for chinee apple (47 years) is much longer than for both prickly acacia (26 years) and parkinsonia (11 years); consequently chinee apple populations also have a higher proportion of individuals in the adult stages (16–17%) than prickly acacia (1.6–3.3%) and parkinsonia (3.5–5.7%). Overall, parkinsonia is the fastest invader of an area, is the shortest lived, and has a higher proportion of juveniles in its population; chinee apple is the opposite in all three areas.

3.3.3 Managing woody-weed infestations

The simulations presented here assumed that 100 % of the targeted population is removed. In reality this is rarely the case. Efficacies of different management strategies can be incorporated into this modelling framework by changing the proportion of individuals removed from the population for the given weed management practice. The elasticity analysis gives insight into which stages should be targeted, coupled with the efficacies of weed management. This aids in deciding on the most effective management control method. However, an optimal control program requires explicit considerations of the management costs (Buhle *et al.*, 2005) and this is discussed later.

Larger adult life stages of steady-state infestations were ranked as the main targets for management, therefore the removal of all adults can result in higher pasture production gains. There are a few reasons why a grazier may find it easier and more efficient to target all adult trees rather than selecting only the larger adults. Firstly, the larger plants may be in the middle of the infestation and therefore

inaccessible. Secondly, it may be easier to use mechanical removal of all plants, by such means as a plough-blade or double chain pulling.

3.3.4 Effectiveness of biocontrol on woody weeds

Biological agents have been used to target seed populations (Julien *et al.*, 2007). These agents are generally used for more than one season and are not capable of removing all seeds, unlike in the above simulation. Biocontrol agents such as seed-eating insects were investigated for prickly acacia and parkinsonia to help control weed infestations (van Klinken, 2006; Cochard & Jackes, 2005; Briano *et al.*, 2002; Radford *et al.*, 2001a). To date, seed biocontrol agents have not been extensively investigated for chinee apple. They are often overlooked due to the perception that any successful weed management outcomes will be clearly noticeable (Reid, 2006). Changes to recruitment, by reducing seeds, will have little effect on a long-lived weed in the short term (Cousens *et al.*, 1986). This is evident in the chinee apple simulation (Figure 3-8), where a 100 % reduction in seed for a year required eight years to have its maximum effect. It is possible that the effect will be unobservable as the decrease in the proportional loss was only 0.015 in that year. A correlation between longevity of the species and the effect of targeting seeds can also be observed in Figure 3-7. Elasticities indicated that the effect of targeting of seeds (*NS*, *SB*, and *F*) is lowest for the longest-lived weed, chinee apple, and highest for the shortest-lived weed, parkinsonia. This is in line with past studies that have shown seed biocontrol agents have less impact on longer-lived weed species. Therefore, the efficacy of biocontrol must be higher for longer-lived species than their shorter-lived counter parts to be effective (Hulme, 2006). One of the main benefits of biocontrol however, is its long-term and persistent impact on weed populations.

The effects of changes in fecundity on growth rate (λ) were investigated in this study to evaluate the potential for biocontrol. Similar to Parker (2000), Figure 3-10 shows that fecundity needs to be reduced by 95 % in order to reduce the population growth rate $(\lambda < 1.0)$, with the exception of chinee apple in upland zones, which only required a 85 % reduction in seed production to affect population growth. If

λ is < 1.0 the population tends towards extinction. What is not indicated by the elasticity analysis is the possibility for the population to remain at steady-state but occupy less area. In fact, this is the premise of biocontrol, where the aim is to create an ecological balance between the weed and its natural enemy to reduce the population to a new "general equilibrium position" (steady-state) (Norris *et al.*, 2003; Sindel, 2000a). Biocontrol is not intended to eradicate a targeted weed but rather reduce its effect (Hulme, 2006; Norris *et al.*, 2003; Sindel, 2000a). In fact, eradication of a weed with a biological agent may be undesirable as it can give rise to the opportunity of reinfestation of the weed without its mortal enemy, as the biocontrol would have died out with the original infestation (Sindel, 2000a).

An alternative analysis was taken to Parker (2000), to determine benefits from biocontrol agents with respect to the decreased size of the steady-state density and consequent reduction in damage to pasture production. The model indicated that weed damage can be reduced with the introduction of a seed-predator. These simulations were run over a 100-year timeline to allow the biological agent and the weeds to find their new steady-states. In all the simulations, the decline in weed damage with respect to reduced fecundity is relatively linear until fecundity is reduced by 80 %. Beyond this point, there is a sharp decline in damage as fecundity decreases. In all instances a 100 % reduction in fecundity resulted in zero damage, implying that no weeds were present and therefore extinct in the area.

For two reasons, it is often thought that targeting seeds where an abundant seedbank is present will rarely result in significant impacts on the population (Maron & Gardner, 2000). Firstly, safe-site theory predicts that when seedbank densities are large, changes to seed input will have little impact on the weed populations as the number of propagules will exceed available suitable micro-sites. Secondly, a decrease in the seedbank will result in less germinants, therefore decreasing the effect of density dependence between germinants. This results in a higher proportion of germinants surviving. Thus the total number of germinants remains constant, despite the decreased seedbank. However, Maron and Gardner (2000, p.267) argue that 'in addition to acting as a buffer, the seedbank acts as a memory that "stores" the cumulative effects of herbivory through time.' Additionally, seeds are the only vectors

through which a new infestation can occur in other areas (Grice, 1998). Therefore, management of the seedbank can have positive results, although it may take time.

3.3.5 Density dependence of woody weeds

It is not unusual for a plant population to have a sigmoidal growth function in terms of biomass or area occupied, where the population initially expands exponentially and then tends to asymptote due to density dependence (Gotelli, 2001). In contrast, the number of woody plants will conform to a quadratic function with respect to time. Initially the number of plants increases, with the population having many smaller plants. As the plants mature and increase in size, the area can no longer sustain so many larger plants. Due to self-thinning, the number of plants will decrease. This is the consequence of natural succession and the plasticity of existing plants capturing area as it becomes available, restricting the recruitment of seedlings. Note that even when the population reaches a steady-state level of damage, the population structure will continue to change for a number of years. Figure 3-3 shows that a chinee apple population in a riparian zone reaches a steady-state (zero growth) in approximately 80 years. However, the population structure within this infestation continues to stabilise for another 40 years. During this latter period, the number of seedlings continues to decrease and proportional damage caused by adults increases. Throughout this chapter, the time it takes the population to reach steady-state is taken to be when the level of damage stabilises, as this is the effect experienced by graziers.

A number of different plant abundance measures where investigated in the preliminary modelling of density dependence for woody-weed populations. The results were not presented, as they had resulted in unrealistic simulations and presenting them would detract from the Aims outlined in Chapter 1. The following will give a brief overview of the findings.

Grice (1996) established it was possible to estimate both biomass (W) and canopy diameter (D) of chinee apple as functions of plant height (H):

$$\log_{10} W = 0.0082H - 1.79 \tag{3.10}$$

$$D = 1.18H - 4.9 \quad (3.11)$$

Dry biomass was initially investigated as a driver but resulted in unrealistic population structures, with too many young juveniles and old adults and too few medium-sized plants not conforming to population structures observed out in the field. This was due to a non-linear exponential relationship between dry weight and canopy area. While canopy diameter with respect to height is a linear functional form (Eq. 3.11), area occupied is exponential due to the functional form (πr^2) and biomass is also exponential to the power of ten (Grice, 1996). This results in both canopy areas (area occupied) and biomass having exponential growth functions of height but in an unparallel relationship. Density dependence was based on area occupied, taken from Gap theory, rather than on dry biomass, as it results in more conventional population structures. Another problem with using biomass is that it is possible for a plant to grow in height and stem diameter whilst having decreasing biomass. Such examples were given by Parker (2000) where a plant loses a limb, or seasonal fluctuations result in changes in plant herbage due to environmental conditions. Crown area (cover) was also investigated; however, assuming that a steady-state is only reached when the whole area is covered, the model predicts too many plants per unit area, as rangeland woody-weed infestations are patchy, with large spaces between crowns. One option was to derive a function based on the crown diameter of each plant but this would result in an estimate based on an estimate. The alternative was to ask a theoretical question of how many cohorts of only one life stage could occupy a hectare. This area per plant was assumed to be the minimum area (including area beyond the canopy) a plant required to survive in each life stage. If this is repeated for all life stages, then it is possible to estimate the occupancy of an area based on the number and size of plants. If a plant dies then there is a gap within the landscape for other plants to occupy, either by plasticity or recruitment.

One problem often encountered with modelling plant population density dependence is overcompensating for shocks causing the model to exhibit population oscillations (Buckley *et al.*, 2001). Most woody plant populations do not exhibit oscillating population dynamics as a result of a shock, due to their slow recovery rates compared to faster growing weed species. The matrix model

presented here shows it has the ability to accommodate shocks to the system without overcompensating. In both Figure 3-8 and Figure 3-9, the populations returned smoothly to a steady-state without 'population oscillation'. In both these figures, the recovering infestation resulted in marginally higher levels of damage than those of steady state, when the infestation was significantly impacted by control. This did not result in population oscillation because a rectangular hyperbola was used to model density dependence. Provided the simulation is given a sufficient timeline, the population will tend back to a steady-state. Additionally, this modelling framework can simulate population dynamics when started at any population structure. This also satisfies the Markov chain theorem, where future states are only dependent on the current state and are independent of past events or states.

An effective weed management strategy needs to predict the current and future impacts of weeds on production systems and resulting profits to determine what type and frequency of weed control should be employed, if any at all (Vitelli, 2000). Weed management can be thought of as an investment with initial setup costs and with benefits spread over time, due to the time-value of money. The sooner the returns, the greater the benefit. This is further investigated in Chapters 6 and 7.

By determining the weed's population dynamics, it is possible to predict the varying impacts weeds may have on production systems, based their on population structures. Additionally, the management of different age/size individuals will result in different efficacies of weed management (Cacho & Spring, 2004). The analysis showed that the targeting of adults stages results in the greatest increase in pasture production. This happens sooner than when targeting the juvenile or seed stages. It was assumed that pasture grasses had instantly occupied the controlled areas and in reality there will be a time lag in recovery. However, there will be a time lag for all life stages, so this analysis can still be used in assessing lifestages for control. This type of analysis can aid in the ranking of life cycle stages from an economic standpoint.

3.4 SUMMARY

In this chapter, a density-dependent stage projection matrix (SPM) model is developed for woody-weed populations. Due to the low natural resources within the Australian rangelands, even fully infested woody-weed populations do not have full cover. Gap theory assumes that a minimum landscape area is required for a plant to survive, based on its own size and the quantity, size and distance of neighbouring plants. The premise of Gap theory is that a gap must occur in the landscape in order for new recruitment or the growth of existing vegetation. A method is developed for estimating the minimum area for each species and plant size. The SPM model is used to explore the population dynamics and management of chinee apple, prickly acacia, and parkinsonia. This selection of weeds was based on the available data. The SPM was validated by reviewing the number of plants in each life stage, when at steady state, and by estimating the mean and variance of time spent by individuals in each life stage (Caswell, 2001). An eigenvalue elasticity analysis was successfully used for both new and steady-stage infestations. The analysis provided the growth rates of new infestations and determined that targeting adult plants, rather than seeds or juveniles, had a greater impact on woody populations.

A damage function is also constructed for woody weeds, where damages are based on species, life stages and quantity of plants. Initially the damage increased almost linearly and then tended to an upper asymptotic bound, due to the diminishing impact of additional weeds. Both the area occupied and damage exhibits sigmoidal population growth functions. At steady state, all three species resulted in approximately 70–83 % weed damage in riparian zones and 30–45 % damage in upland zones. Both prickly acacia and parkinsonia reaches maximum damage around three times faster than chinee apple. Different frequencies of control were also investigated, along with assessing the control benefits of targeting different life stages. Both elasticity analysis and model simulations were used to assess seed biocontrol for woody weeds. The elasticity analysis indicates no change for low seed-biocontrol efficacies (similar to other studies); however, benefits can exist for even low biocontrol efficacy rates. This is due to biocontrol resulting in new lower steady states for woody weeds.

4 MODELLING MULTISPECIES WOODY-WEED INFESTATIONS

Multispecies weed infestations are common in the Australian rangelands. To date, no study has looked at managing these complex systems (Grice & Maywald, 2003; Grice, 2000; Pannell, 1988b; Harper, 1977). If coexisting weed species within an infestation are ignored, weed management may produce unanticipated outcomes, resulting in an inefficient allocation of resources. Some of these coexisting weeds include *Ziziphus mauritiana* (chinee apple), *Acacia nilotica* (prickly acacia), and *Parkinsonia aculeata* (parkinsonia), which can colonise riparian and upland zones (Lawes *et al.*, 2006). In this chapter a multispecies model is developed, using these three species to determine whether the extra complexity is required for weed management decisions.

There are three common weed management decisions for multispecies infestations: (1) whether management of an infestation is warranted from an economic perspective; (2) if managed, which species should be targeted; and (3) what life cycle stage is most vulnerable for control. Ecological theory can provide vital information about complex multispecies systems and can determine effective control strategies for decisions 2 and 3, which assists in decision 1.

In Chapter 3, density-dependent stage projection matrix (SPM) models were used to simulate single-species infestations. The objective was to identify the life cycle stage(s) to be targeted for the most effective management strategy from an ecological perspective. In this chapter a similar approach is used for multispecies infestations (Figure 4-1). A model is required to simulate the dynamics of multispecies weed infestations, while keeping the species and their life cycles identifiable from one another.

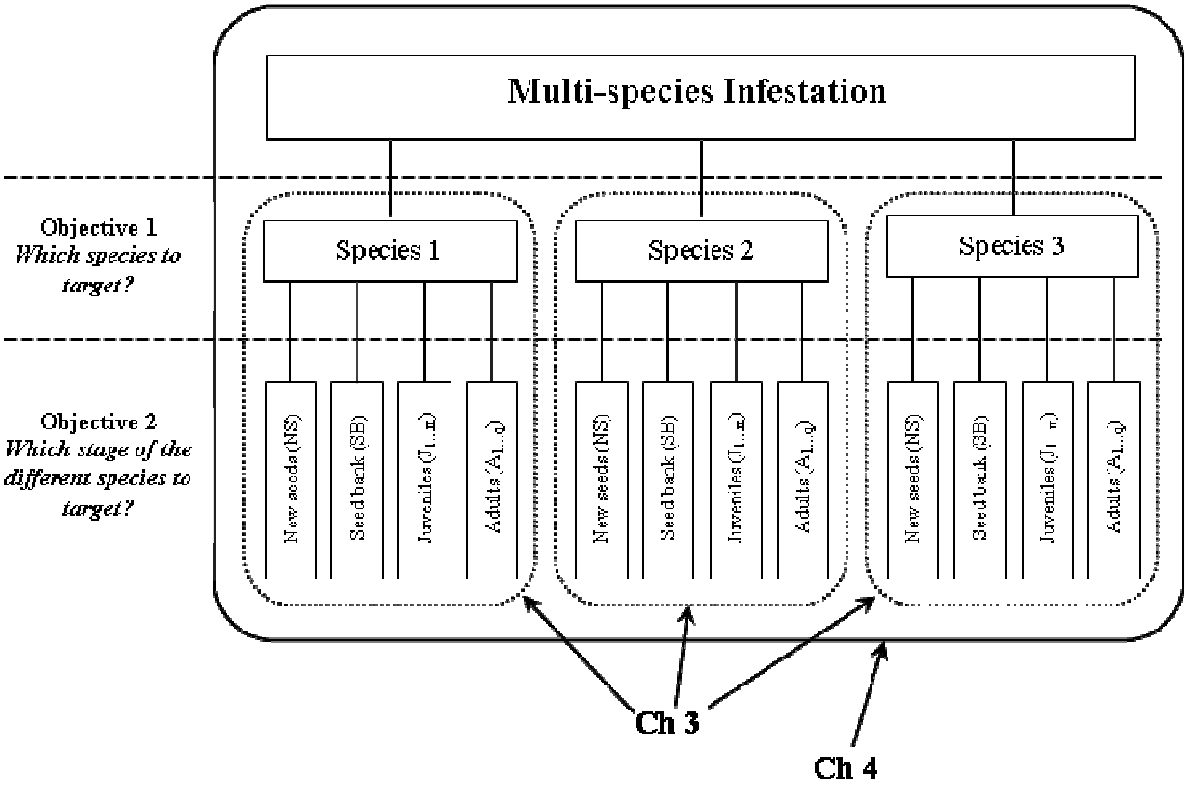


Figure 4-1: The objectives of managing multispecies infestations in rangeland systems.

Noble and Slatyer (1980) outlined important succession processes within plant communities:

- The state of the infestation directly after a disturbance (control) is based on plants and propagules (being seeds in this case study) that have survived the disturbance.
- After the disturbance, there are recruitment and regrowth pulses due to decreased competition for space and resources.
- Following the initial growth pulse, recruitment and regrowth slows as competition increases and established plants are difficult to displace.
- Without further disturbances, long lived species will eventually dominate plant communities.
- These succession processes are important for weed management decisions and will be further investigated within the modelling framework developed in this chapter.

Multispecies stage-projection models have been developed by forestry researchers to aid in management decisions. These models are a class of multi-state models that take advantage of the matrix formulation to structure the demography of the species and the interactions between them, allowing prediction of ecological outcomes for economic management decisions (Caswell, 2001; Lu & Buongiorno, 1993). There are similarities between forestry and woody-weed management, as both are based on multiple species of long-lived perennial plants requiring long-term forecasting, operating in heterogeneous terrain and climatic conditions. They also offer relatively low production yields per area. The primary difference lies in that economic returns of foresters increase with the success of the perennial plants, whereas the economic returns of graziers decrease with the success of perennial weed species. Using these modelling principles, the single-species SPM model developed in Chapter 3 is expanded into a multispecies density-dependent SPM model for Australian rangeland systems. The demography of the case study weeds and their parameterisation remains as presented in Chapter 3. Due to slight changes in the in notations for these parameters, they are defined again in Table 4-1.

It is not the specific purpose of this study to determine an efficient management strategy for these particular species. Rather, it is to establish if the extra complexity of multispecies infestation significantly affects weed management decisions. The model was constructed in MATLAB 7.0® (2007). For weed management purposes the model must be able to simulate the impact of the whole infestation whilst keeping the species and their life cycle stages identifiable from one another. This can be achieved with a multispecies SPM model (Zhao *et al.*, 2005; Lin *et al.*, 1996; Lu & Buongiorno, 1993).

Table 4-1: Definition of model variables and parameters

Variables:	***Unit***	***Description***
$\mathbf{X}_t$	Weeds ha^{-1}	Weed population (size) at time *t*
NS_k	Seeds ha^{-1}	New weed seeds of species *k*
SB_k	Seeds ha^{-1}	Seedbank of species *k*
$J_{k,1\ldots m}$	Plants ha^{-1}	Juveniles at stages 1 to *m* of species *k*
$A_{k,1\ldots q}$	Plants ha^{-1}	Adults at stages 1 to *q* of species *k*
F_k	Seeds ha^{-1}	Fecundity of species *k*
M_k		Probability of weed moving to next stage of species *k*
R_k		Probability of weed remaining in its current stage of species *k*
D	% ha^{-1}	Damage to pasture production
W	m^2	Area occupied by weeds
λ		Infestation growth rate
β	% ha^{-1}	Benefit of weed control (reduction in damages)

Parameters:	***Unit***	
$\mathbf{H}$		Stage projection matrix (SPM)
$u_{1\ldots \mathrm{w}}$		Types of available weed control 1 to *w*
$\mathbf{\Phi}$	% of plants	Efficacy matrix for all weed controls
$\boldsymbol{\varphi}_{u_{k,1\ldots n}}$		Efficacy of each weed control on each life stage of species *k*
γ	m^2	Maximum area available for occupation
$\boldsymbol{\omega}_{k,l}$	m^2	Area required by a weed of a given life stage to exist, of species *k*
$\boldsymbol{\psi}_{k,l}$	Proportion	Damage to pasture production per weed of given life stage, of species *k*
τ	Proportion	Maximum weed damage to pasture production
$c_{k,l}$		Elasticity for life stage of species *k*.
$e_{i,j}$		Elasticity for elements of **H**
z		Maximum number of weed species
$v_{k,l}$	Weeds ha^{-1}	Maximum number of weeds in life stage (*l*) ha^{-1} for species *k*

Subscripts:	
t	Time (years)
k	Weed species
n	Number of life cycle stages for a weed species
s	Number of woody weed species
m	Number of juveniles stages
q	Number of adult stages
w	Total number of different available type of weed controls
i,j	Denotes the elements in a matrix or vector
l	Life cycle stage of weed
wf	Weed free

4.1 METHODS

The multispecies model developed here is based on the original forestry growth model by Buongiorno and Michie (1980) that was extended by Solomon *et al.* (1986) to distinguish tree species (Zhao *et al.*, 2005; Lin *et al.*, 1996; Lu & Buongiorno, 1993), and was presented as follows:

$$\mathbf{x}_{t+1} = \mathbf{H}(\mathbf{x}_t - \mathbf{s}_t) + \mathbf{c} \tag{4.1}$$

and

$$\mathbf{H} = \mathbf{A} + \mathbf{R} \tag{4.2}$$

where $\mathbf{x}_t$ is a population column vector that represents the individuals from all the species, $\mathbf{s}_t$ is a column vector representing the various trees harvested, $\mathbf{c}$ is a column vector representing a constant "ingrowth" (seedlings) into the smallest size class (Namaalwa *et al.*, 2005), **H** is a fixed annual multispecies SPM that is derived from two sub-matrices. Sub-matrix **A** is the transition probability matrix of individuals remaining or advancing to the next life cycles stage; and sub-matrix **R** represents the effects different tree species have on others species' ingrowth.

This model requires a few alterations to be suitable for a multispecies woody weed infestation: firstly, the harvesting of the trees $\mathbf{s}_t$ is replaced with weed management, $\mathbf{s}_t \Rightarrow (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_{kt}})$; as presented in Chapter 3, where $\boldsymbol{\varphi}_{u_{kt}}$ is the mortality rate for *k* species from using control decision *u* at time *t*. The elements of $\boldsymbol{\varphi}$ have values between zero (no effect) and one (total removal). Management of the infestation occurs at the end of the period, but prior to the following season (*t*+1).

In Chapter 3 density-dependence (mortality and plasticity in size) was based on the level of competition between plants for horizontal space (Gap theory) (Shugart & Smith, 1996). Another ecological theory of competition within a mixed species population is "alien thinning", being the mortality in one species due to stress from the density of a neighbouring species (Harper, 1977).

However, it is doubtful that species-to-species competition is significant (Kohyama, 1992; Hubbell & Foster, 1987; Hubbell & Foster, 1986). The size and distance of competitors is more likely to have an effect on individual plant establishment and development, than the species of neighbouring plants[9] (Kohyama, 1992; Silvertown & Law, 1987). For example, a species may dominate over another species by occupying an area faster than its counter part, or by capturing an area for a longer time period. As this model is density dependent, based on available area in the landscape, it will capture this effect. However, if there is interspecific competition based on some other biophysical attribute, such as one species poisoning another, this relationship is unknown and not modelled. Therefore the **R** sub-matrix, representing species-to-species competition, in Eq. 4.2 is set to zero and therefore **H** = **A**. Here **H** is replaced by the density-dependent transition matrix $\mathbf{H}_t$ developed in Chapter 3.

It is unlikely that there will be a constant number of individuals established every year from all the species modelled. The establishment of seedlings is dependent on the number of new seeds (*NS*), the size of the seedbank (*SB*), and the germination rate. The quantity of new seeds is a function of the number and size of adult plants. If there are no adults, there are no new seeds. Consequently, no new seedlings are produced when the seedbank is depleted, resulting in no immigration. Therefore, $\mathbf{c} = 0$ as fecundity and germination are integrated into $\mathbf{H}_t$. With these modifications, Eqs 4.1 and 4.2 are reduced to a similar form as the SPM in Chapter 3, but with the capacity for modelling multispecies infestations, as:

$$\mathbf{x}_{t+1} = (\mathbf{H}_t \mathbf{x}_t) - (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_t}) \tag{4.3}$$

[9] If there is interspecific competition between specific species, this can be added into this modelling framework as is given in Eq.4.2, by creating an interspecific matrix similar in size to $\mathbf{H}_t$ in **Figure 4-2**, where the diagonal elements will be zero, and the other elements are parameterised for species-to-species competition.

The population column vector $\mathbf{x}_t = [\mathbf{x}_{klt}]$ now represents the number of individuals of all the weed species (k) in each life cycle stage (l), at time t (Zhao *et al.*, 2005; Lu & Buongiorno, 1993). See Figure 4-2 for a graphical representation. Species are denoted by subscript k = 1, 2, 3, representing chinee apple, prickly acacia and parkinsonia. As in Chapter 3 the four main life cycle stages are new seeds (*NS*), seedbank (*SB*), juveniles (*J*), and adults (*A*). The juvenile and adult stages are then broken down into sub stages $(J_1, J_2, ..., J_{m_k})$ and $(A_1, A_2, ..., A_{q_k})$, based on the species' demography. The SPM $\mathbf{H}_t$ contains the projection matrices for all the woody weeds at time t (Zhao *et al.*, 2005; Lin *et al.*, 1996). $\mathbf{H}_t$ has dimensions 25 x 25 based on the sum of life stages of chinee apple, prickly acacia and parkinsonia (10 + 8 + 7). The probability of an individual remaining in its stage or moving on to the next stage is represented by R_{kl} and M_{kl}. The mortality rate for stage l of species k is 1– (R_{kl} + M_{kl}). The fecundities of adult stages are represented by F_{kl}. A key assumption that every individual can be placed in a life cycle stage is retained (Caswell, 2001; Brault & Caswell, 1993).

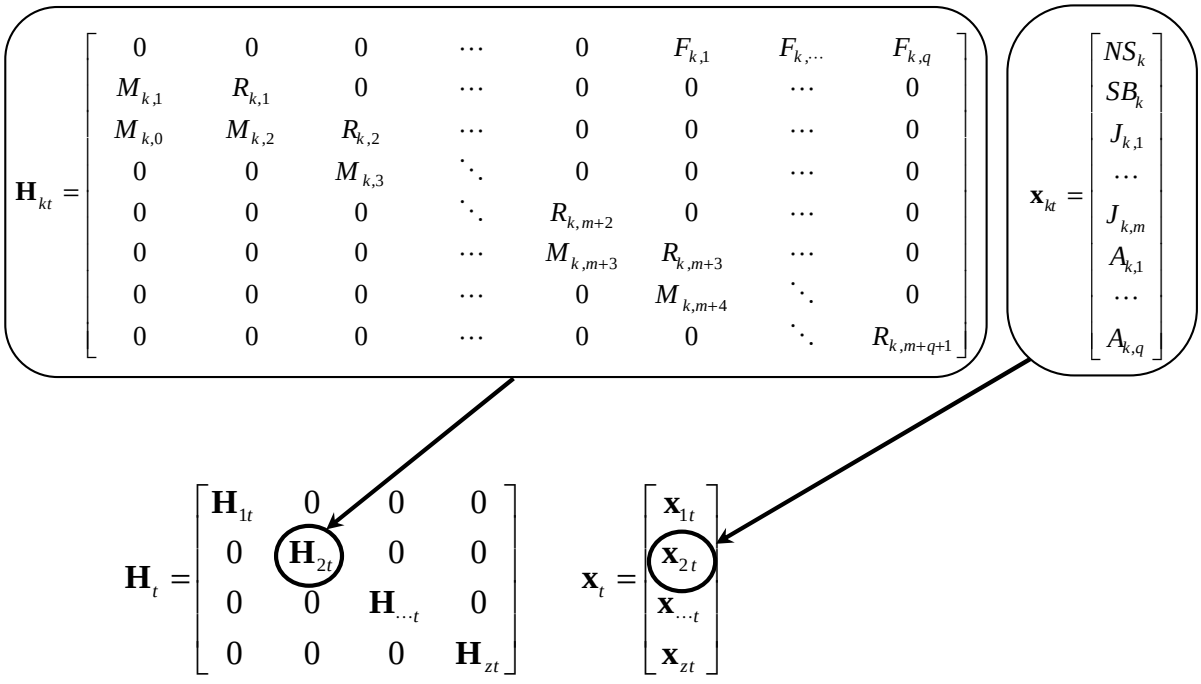


Figure 4-2: Multispecies stage-classified population projection matrix (SPM) ($\mathbf{H}_t$) and population column vector ($\mathbf{c}_t$). The sub-matrices and population vectors for each species are $\mathbf{H}_{kt}$ and $\mathbf{X}_{kt}$, respectively. $\mathbf{H}_t$ allows for competition between species but the zero elements between the sub-matrices prevent individuals becoming another species, which is clearly impossible.

4.1.1 Density dependence for multispecies infestations

Similar to the single-species population model presented in Chapter 3, there is a transition between the exponential-growth of new populations ($\mathbf{H}_0$) and asymptotic growth at steady-state ($\mathbf{H}_\infty$) due to density dependence. The elements of H_{k0} in Figure 4-2 are the same as those used in the single-species H_0 matrix in Chapter 3 for species *k*. Due to the 'hierarchy of exploitation' of resources, the establishment and survival of younger plants is significantly affected by population density (Silvertown, 1982, p.125). Individuals of the different species affect each other, through the density-dependent feedback mechanism. However, the matrix does not allow an individual from one species to enter another species' stages (see zero elements in H_t). The transition between $\mathbf{H}_0$ and $\mathbf{H}_\infty$ is modelled with a rectangular hyperbola (Cacho & Spring, 2004) resulting in a sigmoid population growth curve. The SPM at time *t* is given by:

$$\mathbf{H}_t = \frac{\gamma \mathbf{H}_0}{\gamma + \left(\frac{\mathbf{H}_0}{\mathbf{H}_\infty} - 1 \right) W_t} \tag{4.4}$$

Here the matrix division is an element by element operation and does not require matrix inversion. Gap theory is used as a surrogate for the measure of competition between neighbouring plants (Shugart & Smith, 1996). The maximum area available for occupation is represented by γ, in this case one hectare. The total area occupied by the infestation (*W*) at *t*, is calculated as $W_t = \boldsymbol{\omega}^T \mathbf{x}_t$, where $\boldsymbol{\omega}$ is a column vector representing the average area required by individuals in each life stage (*l*) for each species (*k*) being $\boldsymbol{\omega} = [\boldsymbol{\omega}_{kl}]$, the vectors inside $\boldsymbol{\omega}$ vectors are the same as those used in Chapter 3 for single-species infestations. It is possible to combine the $\boldsymbol{\omega}_k$ vectors, as they have a common unit of measure. Seeds are assumed not to require any area to exist.

4.1.2 Damage from multispecies infestations

The damage function for multispecies infestations has the same form as that used in Chapter 3:

$$D_t = \frac{\boldsymbol{\psi}\mathbf{x}_t}{1+\frac{\boldsymbol{\psi}\mathbf{x}_t}{\tau}} \tag{4.5}$$

where D_t is the damage (proportion lost) to pasture production at time *t* due to all the weeds in the infestation $\mathbf{x}_t = [\mathbf{x}_{klt}]$ ha^{-1}, $\boldsymbol{\psi} = \boldsymbol{\psi}_{kl}$ is a damage index vector for the average loss of pasture production per weed of each species in each life stage (as weed density approaches zero), and τ is the maximum proportion of damage as weed density approaches infinity. The three will result in different levels of production damage with respect to their density, because their parameters differ. Therefore D_t for multispecies infestation may vary as the species composition changes, even if the total population is relatively stable. This will therefore also affect the benefits derived from weed control.

4.1.3 Benefits of targeting different life stages

In Chapter 3, perturbation analyses is used to identify the life cycle stage that had the greatest effect on population growth rate, to identify weeds' 'Achilles heel' (Caswell, 2001; Parker, 2000). The formula to estimate the elasticities (e_{ij}) of elements is defined by Caswell (2001, p.226) as:

$$e_{ij} = \frac{h_{ij}}{\lambda}\frac{\partial\lambda}{\partial h_{ij}} \tag{4.6}$$

The elasticity of all the species and their life stages $\boldsymbol{c}_l$ is a column addition of e_{ij} calculated (Davis, 2006; Parker, 2000; Brown *et al.*, 1999) as:

$$c_l = \sum_{i=1}^{i=n} e_{ij} \tag{4.7}$$

4.2 RESULTS

The following simulations are for multispecies infestations of chinee apple, prickly acacia, and parkinsonia in both riparian and upland zones. The parameters used in this chapter remain unchanged from those in Chapter 3. The multispecies simulations start with 2,000 seeds for each species at $t = 0$ and are run until $T = 300$ (Figure 4-3). Both zones were estimated to reach steady-state in approximately 350 years (Figure 4-3), where all the damage is from chinee apple. The accuracy of

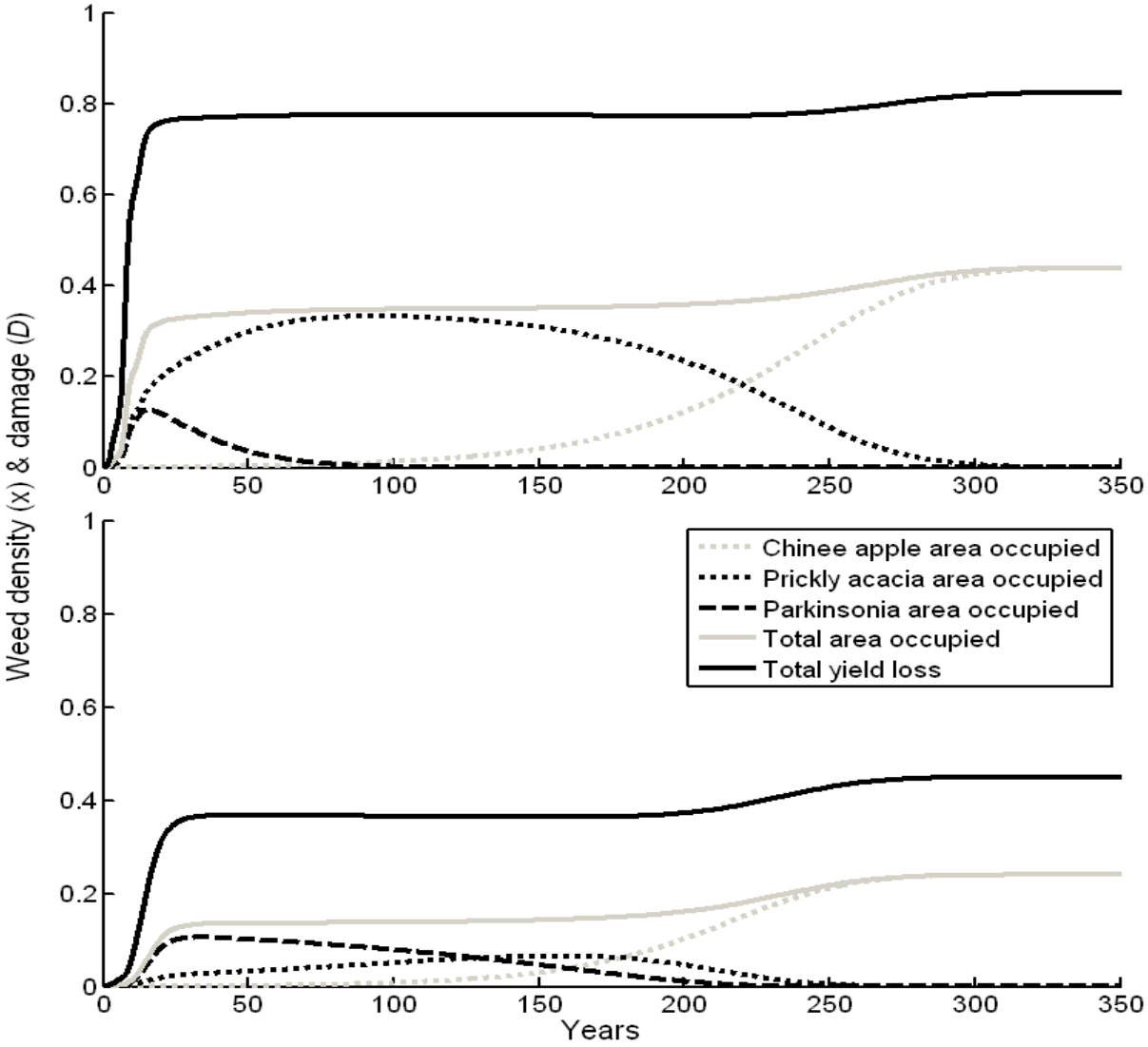


Figure 4-3: Temporal simulations of chinee apple, prickly acacia, and parkinsonia multispecies infestations in (a) riparian and (b) upland zones. Area occupied indicates the density crowdedness species contributes to the infestation over time. Total damage is the proportional reduction in pasture production due to the infestation over time.

these distant forecasts is unknown and arguably irrelevant, as graziers' weed management requirements lie within the foreseeable future. The model does, however, provide insight into the density-dependence processes operating between the species over time. The model can be started at any point in time with any combination of weed species.

4.2.1 Elasticities of species and life stages

From an ecological perspective, weed control should target the species and life stages that have the greatest effect on population growth (Caswell, 2001). Figure 4-3 indicates that different species will dominate multispecies infestations over time. Parkinsonia will initially dominate over prickly acacia and chinee apple. Then prickly acacia will dominate. In time chinee apple will become the dominant species and the multispecies infestation will essentially become a single-species infestation. Therefore management goals may change over time based on the state of the infestation.

For the weed management decisions, the state of multispecies weed infestations goes beyond purely density, as the cost and efficacy of control will change with the sizes and structure of the subpopulations from the different species. Therefore, elasticity analysis (Eqs. 4.6 and **3.5**) is used to investigate the contribution of the different species and their life stages. Figure 4-4 shows the elasticities for the three species. The elasticities generated for the multispecies infestation were exactly the same as those generated for the dominant species in the single-species infestations in Chapter 3. Elasticities show a similar pattern as density, with changes over time and changes in the dominant species.

The use of elasticities as a decision tool for multispecies infestation is problematic from a weed management perspective. Figure 4-4 indicates that weed control for this riparian multispecies infestation should focus on parkinsonia t_0 to t_{10}, prickly acacia t_{10} to t_{20} and chinee apple > t_{20}, with all other species outside these time periods to be ignored. From an integrated weed management (IWM) perspective, controlling more than one species and more than one life stage is a more realistic strategy,

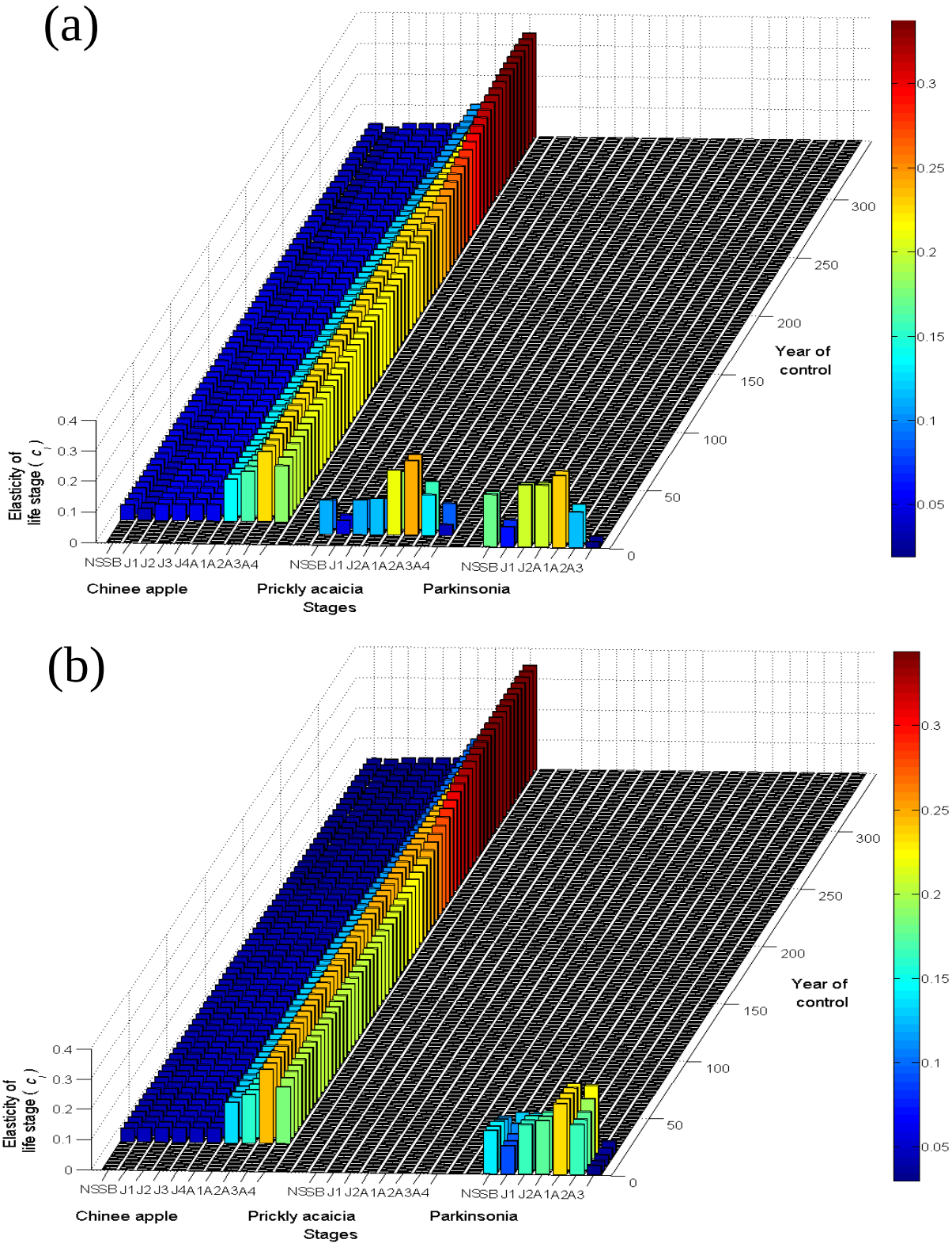


Figure 4-4: The column elasticities (c_i) of the life stages of chinee apple, prickly acacia, and parkinsonia in (a) riparian and (b) upland zones with five year time steps. Column elasticities represent the amount each life cycle stage from each species contributes to the multispecies infestation. The adult life stages include fecundity. Life stages include new-seed (*NS*), seedbank (*SB*), small juveniles (J_1), medium juveniles (J_2), large juveniles (J_3), largest juveniles (J_4), small adults (A_1), medium adults (A_2), large adults (A_3), and largest adults (A_4). The life stages with the largest elasticities should be targeted first for the greatest impact on the infestation.

especially when there is a number of near dominant species; i.e. the 100–year mark in Figure 4-3 (b). Although the different species may respond slightly differently to various control methods, all three species will be affected by the applied control. For example, if the infestation is bulldozed, then all the species within the infestation will be treated. Therefore, it must be determined if management decisions will significantly change with the added complexities of multispecies infestation dynamics.

4.2.2 Evaluating elasticities for multispecies infestations

The validity of using eigenvalue elasticity analysis for multispecies infestations is now investigated. As stipulated in Section 3.1.3, the elasticities of λ with respect to h_{ij} are often interpreted as the "proportional contribution" of each h_{ij} to λ, as the elasticities sum to one (Caswell, 2001; Brault & Caswell, 1993). Therefore, any life stages from any species not contributing to the infestation does not need to be targeted for weed control, and more control effort should be given to the life stages and species with the largest contributions (elasticities). In this modelling experiment, only the species and life stages identified in the elasticity analysis (Figure 4-4) will be expected to influence the level weed damage. Controls were assumed to occur in time periods t = 20, 100, and 250. In each control event there is a single removal of all the adults from (a) all species, (b) chinee apple, (c) prickly acacia, or (d) parkinsonia, for both riparian and upland zones. If the elasticity analysis is correct, then only dominant species in the year of control (Figure 4-4) will affect the infestation, as all other species and life stages do not contribute to the infestation as they have elasticity values of zero.

Figure 4-5 and Figure 4-7 show the effects of targeting different species within a multispecies infestation. The elasticity analysis estimates the contribution of different life stages to populations. However, the removal of life stages from various species with elasticity values of zero still had an impact on the infestation (Figure 4-5 and Figure 4-7), assuming the species was still present at the time of control. Therefore, these species and their life stages still contributed to the multispecies infestation, even though the elasticity analysis indicated that they were unimportant.

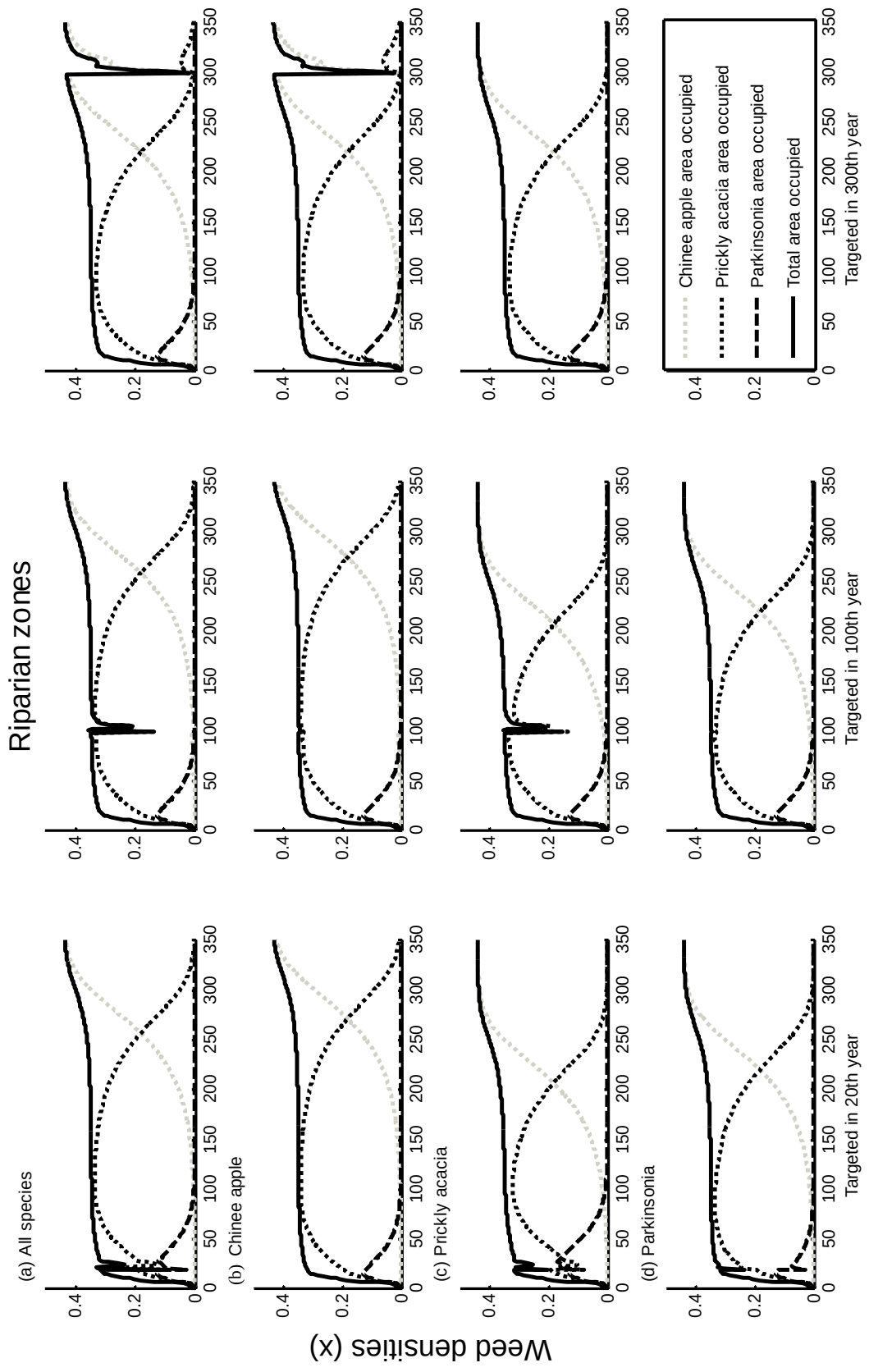


Figure 4-5: The effects of a single control event on population structure and area occupied of a multispecies infestation in riparian zones. Weed control is applied in years 20, 100, and 250, by removing all the adults from (a) all species, (b) chinee apple, (c) prickly acacia, and (d) parkinsonia in riparian zones.

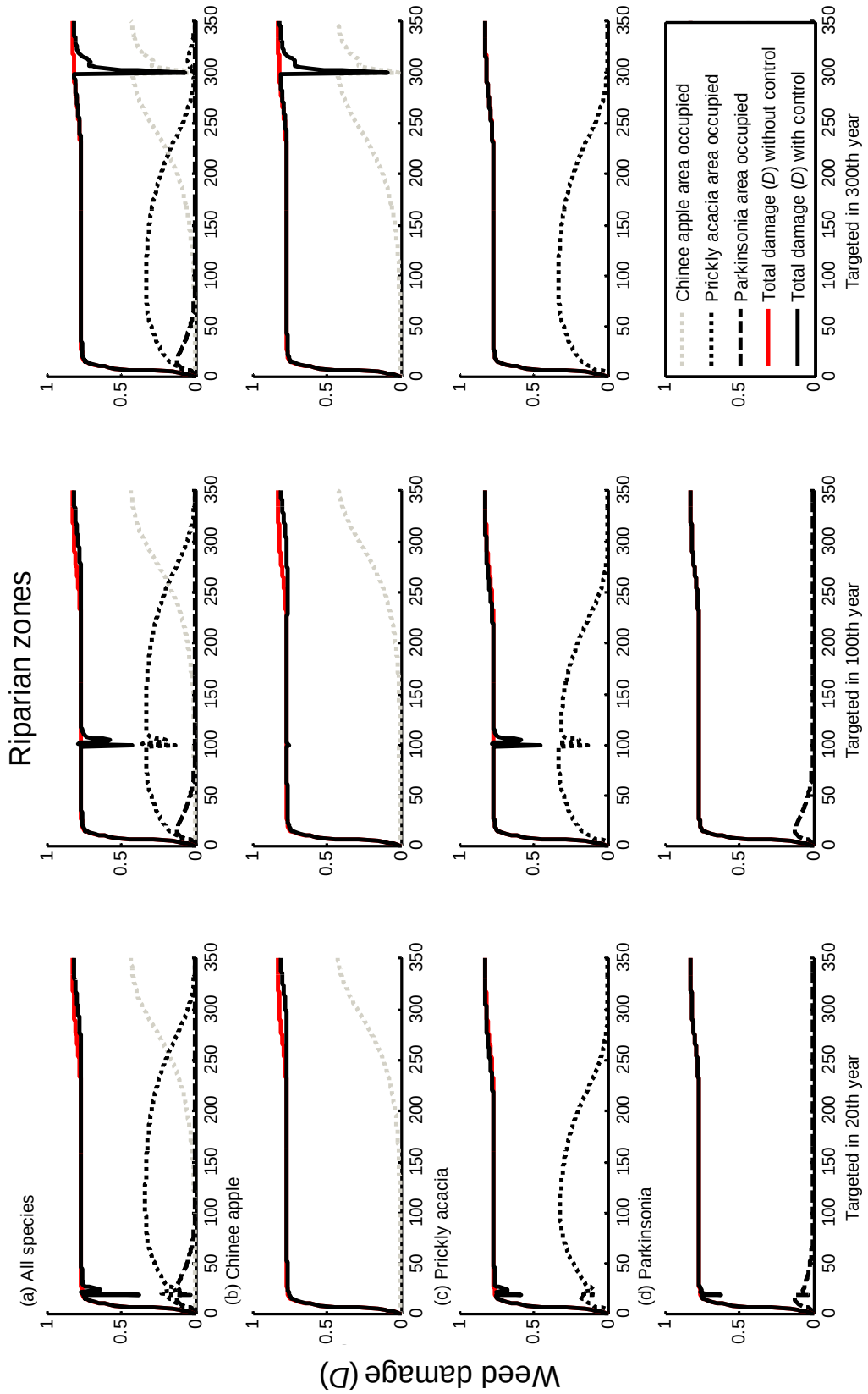


Figure 4-6: The effects of a single control event on weed damage (*D*) in riparian zones. Weed control is applied in years 20, 100, and 250, by removing all the adults from (a) all species, (b) chinee apple, (c) prickly acacia, and (d) parkinsonia in riparian zones. The density of the targeted weed is also presented.

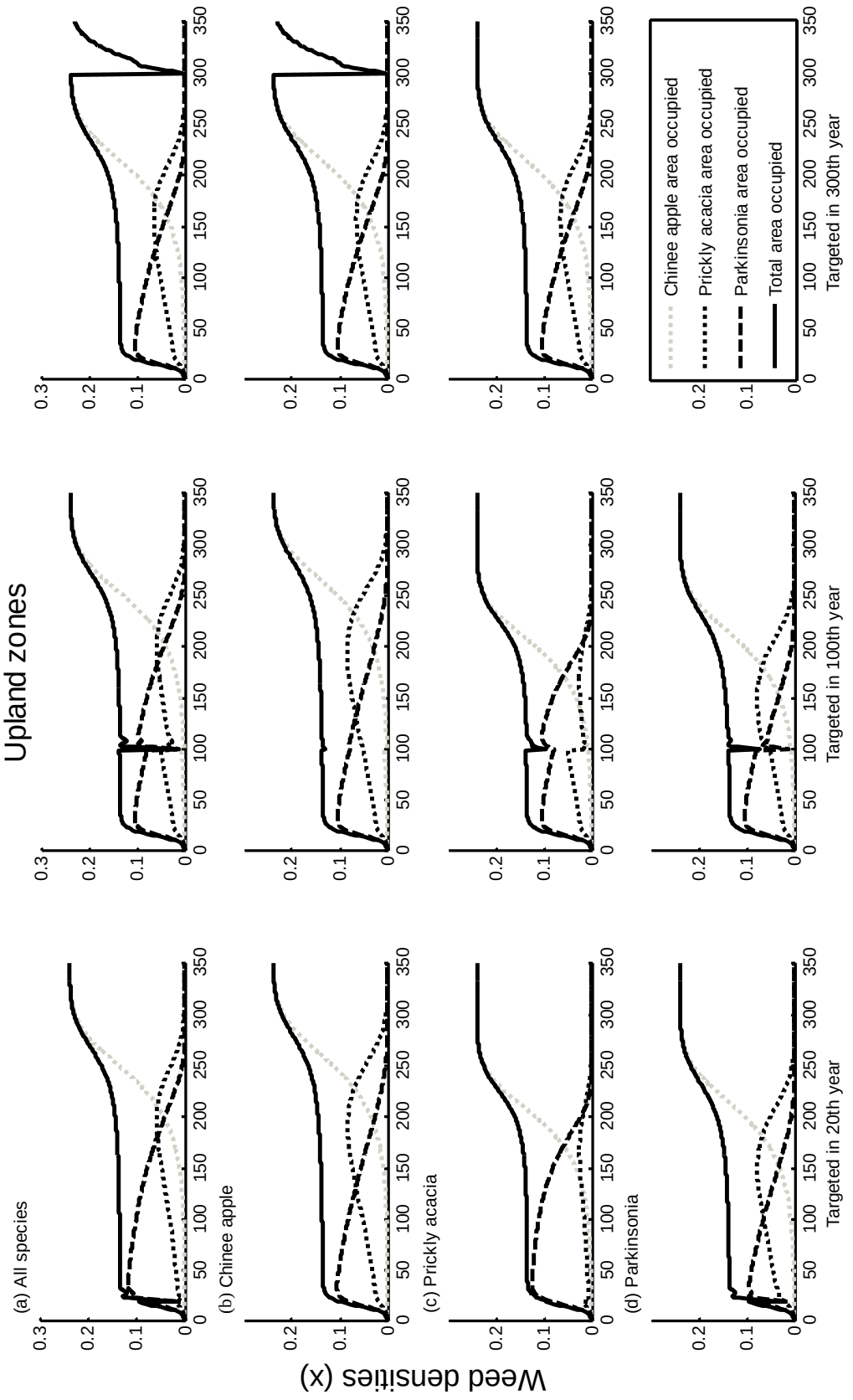


Figure 4-7: The effects of a single control event on population structure and area occupied of a multispecies infestation in upland zones. Weed control is applied in years 20, 100, and 250, by removing all the adults from (a) all species, (b) chinee apple, (c) prickly acacia, and (d) parkinsonia in riparian zones.

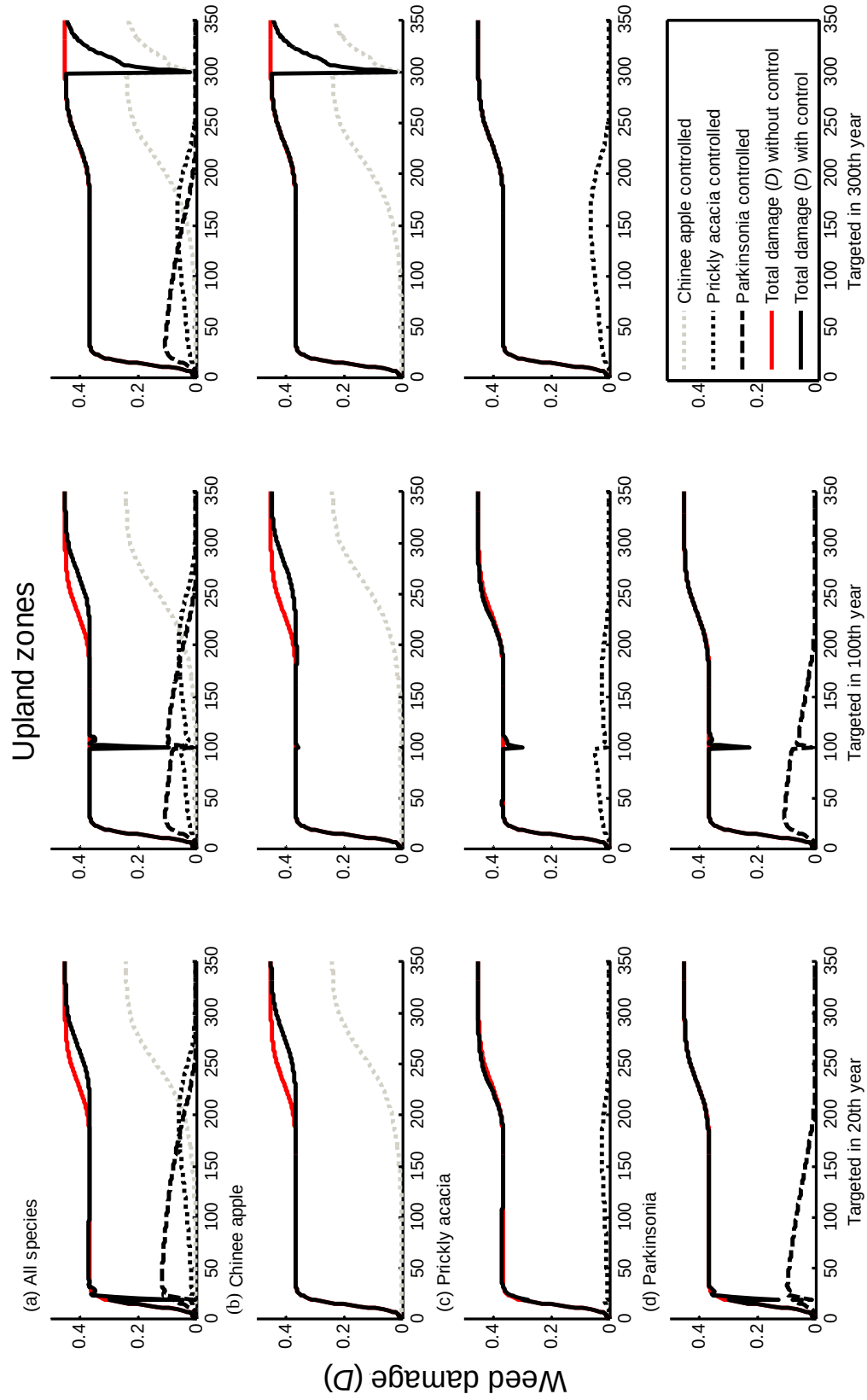


Figure 4-8: The effects of a single control event on weed damage (*D*) in upland zones. Weed control is applied in years 20, 100, and 250, by removing all the adults from (a) all species, (b) chinee apple, (c) prickly acacia, and (d) parkinsonia in riparian zones. The density of the targeted weed is also presented.

The results in Figure 4-6 to Figure 4-7 show the benefits of adopting IWM over single species control. Apart from steady-state infestations of chinee apple (that have essentially become monocultures) in upland zones, the control of all species will result in less weed damage than controlling only a single species. This includes species with elasticities of zero value, for example targeting any species in the upland infestation in year t = 100 (Figure 4-8), where only chinee apple had an elasticity value greater than zero. Moreover, some species can in fact benefit from the control event. Controlling all the species at t=100 in upland zones (Figure 4-7(a)), resulted in parkinsonia increasing its area occupied shortly after the control event, due to decreased competition. Clearly, elasticity analysis of multispecies infestations has little to offer in the management of woody-weed infestations. However, elasticity analysis of single-species infestations may provide useful information in understating the possible succession of species that will occur after a control event.

4.2.3 Evaluating elasticity analysis for single infestations

This section quantifies the benefits of elasticity analysis for single species infestations, with an aim to develop an alternative analysis for multispecies infestation with respect to Objectives 1 and 2. Following Chapter 3's investigations into the relationship between the elasticities and the impact of controlling life stages, this section quantifies the relationship. This analysis was not required in Chapter 3 as the elasticity analysis was sufficient for single-species infestation. Chapter 3 showed a strong relationship between the elasticity analysis' ranking of life stages and the impact of controlling them on the population, i.e. the life stages with the largest elasticities reduced the infestation's size and damage to pasture production by the greatest amount and for the longest time period (Figure 3-8). The correlation between elasticities and the cumulative effect on the infestation is further investigated through mathematical analysis, for both single and multispecies infestations.

The benefits of managing a weed infestation are given by the magnitudinal decrease in production damage and the duration of the reduction. Similar to Chapter 3, these ecological analyses are not discounted for the time value of benefits, as this is undertaken in Chapters 6 and 7 in the economic

optimisation of woody-weed management. It is also assumed that pasture grasses will immediately reoccupy controlled areas. Figure 4-9 illustrates the effects of a single removal of all individuals within each life stage (*l*) of a riparian chinee apple infestation, i.e. all the largest adults (A4) are removed and the population is allowed to return to steady-state. This simulation starts with a riparian steady-state chinee apple infestation. By targeting juveniles and seeds, the full impact of control is delayed for a number of years (Figure 4-9). In essence, the sum of the benefit of controlling each life stage is the area between the damage lines with and without control; calculable as:

$$B_l = \sum_{t=1}^{T} \left(D - D_{lt}^{*} \right) \tag{4.8}$$

where B_l are the benefits obtained for T=100 years, from controlling life stage l, D is damage without weed control, and D* is damage with weed control. These benefits (B_l) are given in Figure 4-10, based on the same population as in Figure 4-9. The largest adults (A4) will result in $B_{10} = 1.45$; i.e. the benefit over time from controlling this life stage equates to a gain of 1.45 hectares of weed-free grazing production.

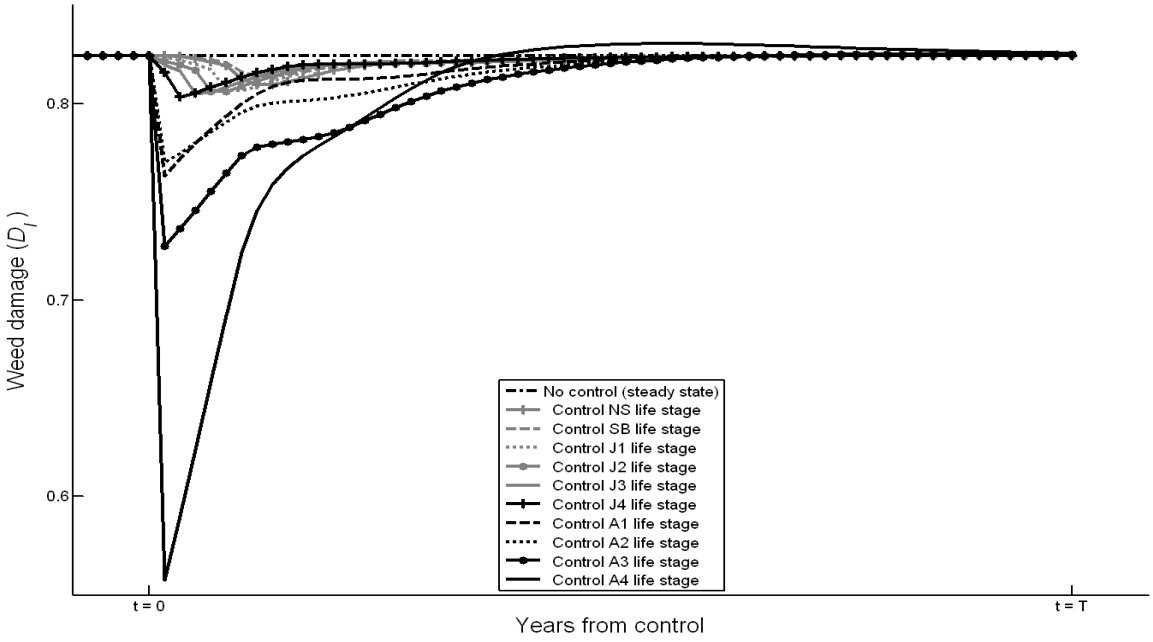


Figure 4-9: Illustrating the benefits (B_l) of a single control, removing all individuals within each life stage. Simulation starts from a riparian steady-state chinee apple infestation. The benefit of controlling each life stage includes the magnitudinal decrease in damage and the longevity of the effect.

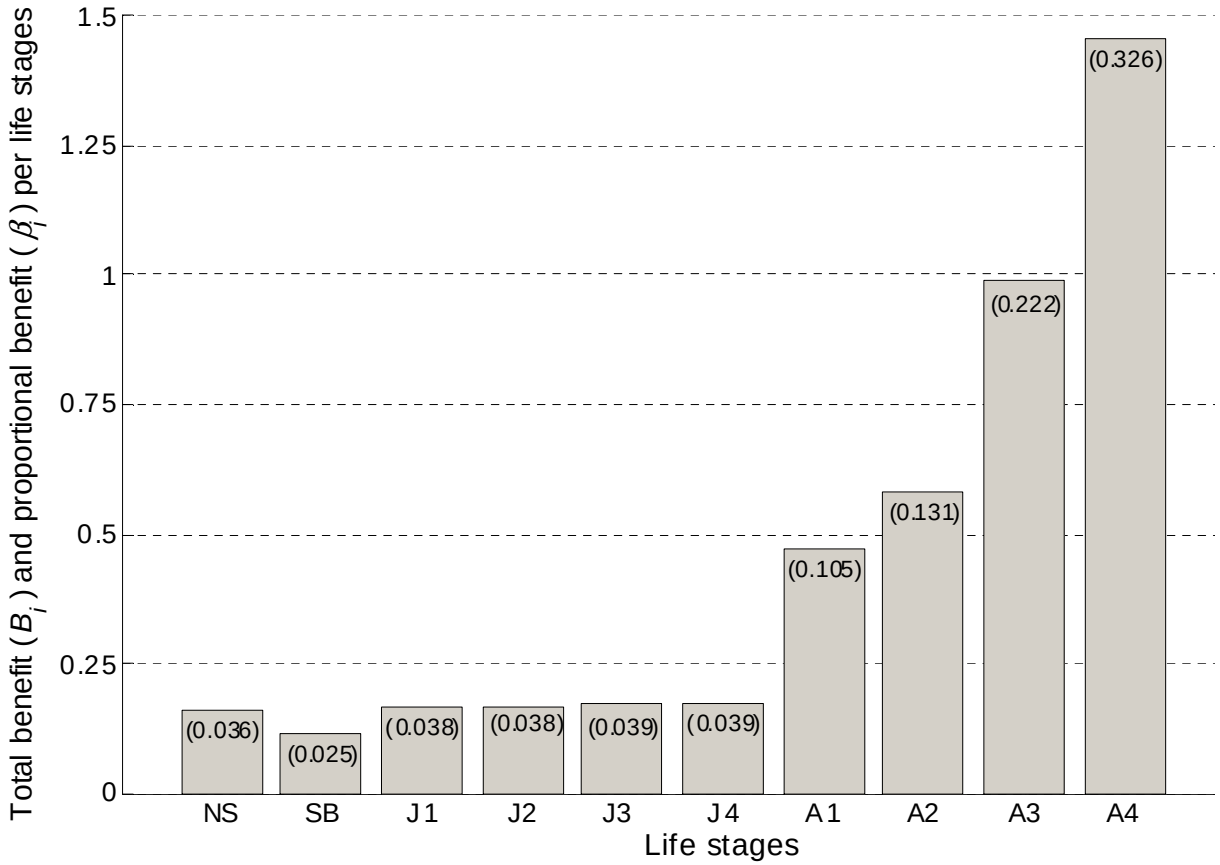


Figure 4-10: Total benefit (B_l) (reduced damages) over time from a single control event, removing all individuals within each life stage. The results are based on a steady-state chinee apple infestation in a riparian zone. The proportional benefit (β_l) of each life stage is given within the bars.

As with ecological elasticity analysis, the aim is to identify the life cycle stage that has the greatest impact on the population growth (Achilles heel).

Similar to the elasticity analysis, where each life stage represents a proportion of the growth rate and $\sum c_l = 1.0$; the proportional benefits of controlling individual life stages are calculated by normalising the benefits so they add up to 1.0, as follows:

$$\beta_l = \frac{\sum_{t=1}^{T}\left(D - D_{lt}^{*}\right)}{\sum_{t=1}^{T}\sum_{l=1}^{l=n}\left(D - D_{lt}^{*}\right)} \tag{4.9}$$

where β_l is the proportional benefit of controlling each life stage. The β_l results are reported within the bars in Figure 4-10. To compare the proportional benefits (β_l) with eigenvalue elasticities (c_l) of each life stage for all three monoculture weed species and zones (Figure 4-11). These analyses are based on singles species steady-state infestations of woody weeds. Both analyses result in a similar weighting for the different life stages. Whilst there are some slight differences in rankings, this only occurs when the life stages have similar values to each other. For example, the elasticity analysis ranks the life stages for prickly acacia in riparian zones as A4($C_8 = 0.175$)>A3($C_7 = 0.161$)>A1($C_5 = 0.159$), whereas proportional benefit analysis ranks A1($\beta_5 = 0.162$)>A4($\beta_8 = 0.160$)>A3($\beta_7 = 0.157$). One of the reasons for the difference is that the eigenvalue elasticity analysis is derived from a snap-shot of $\mathbf{H}_t$, whereas the proportional analysis is a cumulative measure that uses $\mathbf{H}_t$ over time. Elasticity analyses are often used on fixed transition matrices (Caswell, 2001). This study shows that elasticity analysis can also be used on density-dependent stage transition matrices, and the elasticity values will change with the changes of the elements within transition matrix. Each analysis can be thought of as a snap-shot of the matrix at that point in time. It does not consider how the matrix will change post-control, due to decreased crowding, which affects the recovery of the infestation. Since $\mathbf{H}_t$ changes over time due to density dependence, the proportional analysis is in fact more applicable for weed management decisions. From a weed management perspective there is little difference between these values, when combined with all other decisions and unknowns. Whether one life stage is slightly higher than another is irrelevant for weed management decisions. Either type of analysis can be used for single-species infestation management decisions, with similar likelihoods of weed management success, identifying similar life stages to control.

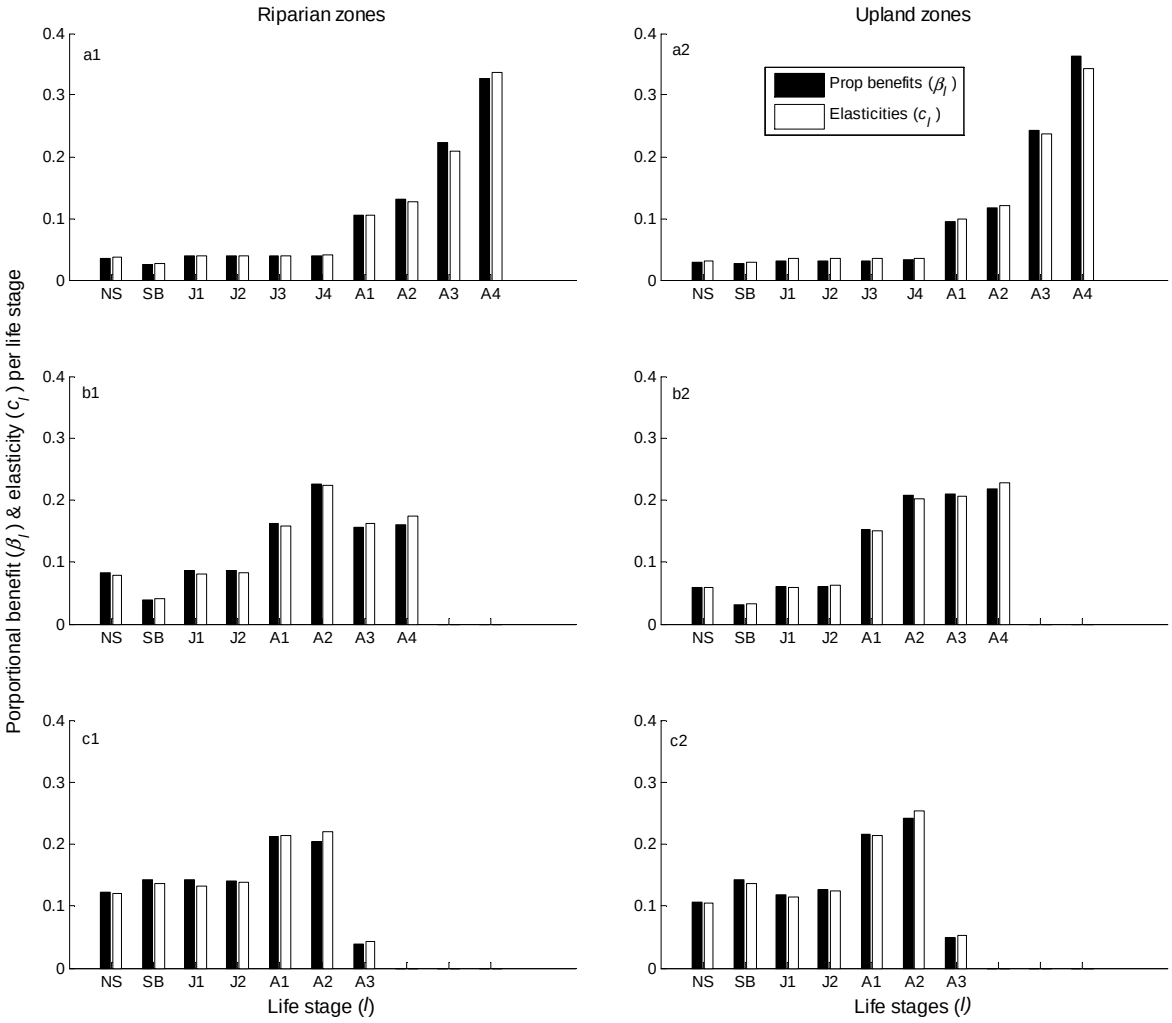


Figure 4-11: Comparing the proportional benefit (β_l) from managing the different life cycle stages to the elasticities (c_l) of the different life cycle stages of steady-state infestations of (a) chinee apple, (b) prickly acacia, and (c) parkinsonia, in (1) riparian and (2) upland zones.

4.2.4 Proportional benefit analysis

Eq. 4.9 is expanded for multispecies infestations, where the proportional benefits (β_{kl}) are calculated as:

$$\beta_{kl} = \frac{\sum_{t=1}^{T}\left(D - D^{*}_{klt}\right)}{\sum_{k=1}^{k=s}\sum_{l=1}^{l=n}\sum_{t=1}^{t=T}\left(D - D^{*}_{klt}\right)} \tag{4.10}$$

where k is the species targeted and l is the life stage. These β_{kl} results are reported in Figure 4-12 for (a) riparian and (b) upland zones. As before, the benefit of control is the difference in damage with and without control. Control consists of all the individual plants in each life stage and species being removed in turn, while leaving all the other life stages and species untouched, and the benefits are derived for $T = 100$. The initial state of the infestation prior to the control event (year of control) is the same as the infestations presented in Figure 4-3. For example, when $t = 0$ the infestation in Figure 4-3 is started with 2000 seeds of each species, and the total damage is zero. Figure 4-12 indicates that the greatest benefit from targeting seeds is in the very early years of the multispecies infestation. Once the infestation is established, say at $t > 15$, then weed control shifts from seeds to plants, see $t = 10$ in Figure 4-12.

Similar to previous analysis, the primary species for management shifts from parkinsonia to prickly acacia and eventually to chinee apple. Now the weighting of the dominant species has decreased and includes the benefit of controlling other species and their life stages. This is in line with IWM, as the goal of weed control moves from managing individual species to the infestation as a whole.

Once an infestation is established there are benefits from controlling any species or life stages. For riparian zones the greatest benefit is from controlling parkinsonia adults and later chinee apple adults (Figure 4-12a). For upland zones, first parkinsonia should be targeted (with five year sampling time steps) and then chinee apple adults (Figure 4-12b). In reality weed control would focus on the three main life stages of seed (S), juveniles (J), and adults (A). These have been aggregated to estimate the proportional benefits (β_L) of the main life stages (Figure 4-13). For example, these results indicate that for newer riparian infestations ($\sim t = 20$), the ranking of species is prickly acacia, chinee apple, and parkinsonia and the ranking of stages is adults (A), juveniles (J), and lastly seeds (S).

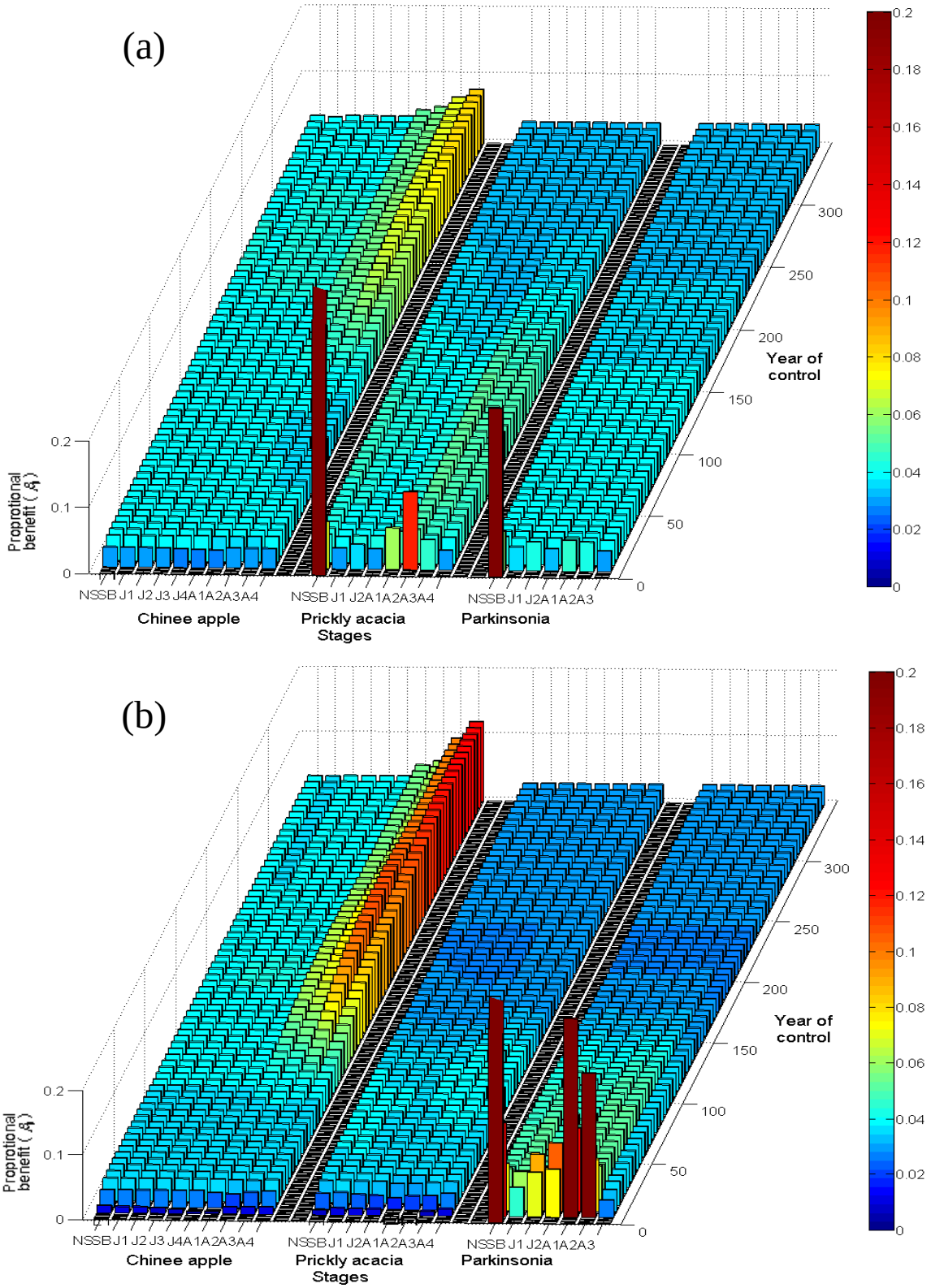


Figure 4-12: The proportional benefit (β_l) from targeting different life stages of chinee apple, prickly acacia, and parkinsonia in (a) riparian and (b) upland zones (with five-year sampling increments). Column values are the amount that each life cycle stage of each species contributes to the damage of multispecies infestations. The adult life stages include fecundity. Life stages include new-seed (*NS*), seedbank (*SB*), small juveniles (J_1), medium juveniles (J_2), large juveniles (J_3), largest juveniles (J_4), small adults (A_1), medium adults (A_2), large adults (A_3), and largest adults (A_4).

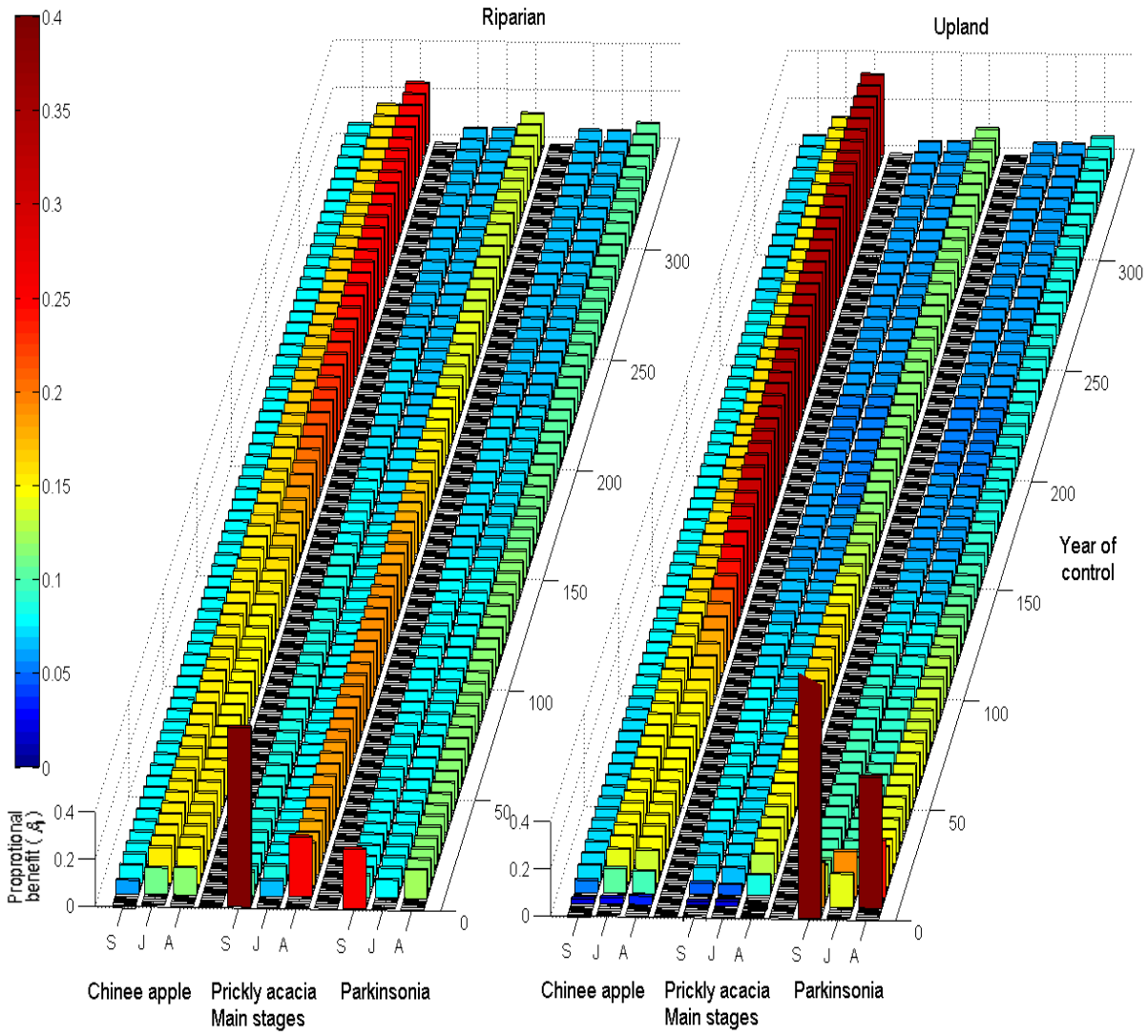


Figure 4-13: The proportional benefit (β_L) from targeting different main life stages of chinee apple, prickly acacia, and parkinsonia in riparian and upland zones (with five-year sampling increments). Column values are the amount that each life cycle stage of each species contributes to the damage of multispecies infestations. The adult life stages include fecundity. Main life-stages are seeds (*S*), juveniles (*J*), and adults (*A*).

4.3 DISCUSSION

Graziers face the arduous task of weighing up the cost and benefits of weed management strategies in a vast heterogeneous system that offers low economic returns and limited available resources. This decision making process is made more difficult by the complexities of multispecies infestations. The relationship between species within an infestation is complex and the population structures are constantly changing. This analysis provides useful information to assist identifying which species (Objective 1) and life stages (Objective 2) to target for the greatest impact on the infestation.

Matrix models are often perceived as the poorer cousin of more detailed individual based population models. As Winkler (2006, p.429) states, "… matrix models which in their strict form are not suited to the investigation of phenomena such as coexistence, competitive exclusion, or succession." However multispecies matrix models have been used in forestry for many years for coexisting, competitive populations with succession processes (Zhao *et al.*, 2005; Lin *et al.*, 1996; Lu & Buongiorno, 1993; Solomon *et al.*, 1986). These forestry models have been adapted into a density-dependent SPM modelling framework for multispecies woody-weed infestations. The model requires relatively few ecological parameters in contrast to individual based models (where many of the parameters are currently unknown) to gain insight into the dynamics and management of these complex weed infestations from an ecological perspective. Chapter 3 provides ecological information on single-species woody weed infestations, including population growth rates, elasticities of life cycle stages, time to reach steady-state, and population structure. However, multispecies woody weed infestations are a common feature in the Australian rangelands and it is useful to determine whether management should change in response to changing species composition.

Results show that eigenvalue analysis only provides information about the dominant species within the multispecies matrix. In essence, this ignores the theoretical benefits of integrated weed management (IWM), which targets different species within the infestation simultaneously. It was therefore concluded that eigenvalue elasticity analysis for multispecies infestations is less effective than an

alternative benefit analysis. The proportional benefit analysis is based on future changes in production losses (damages) with respect to controlling different species and their life stages using the same matrix population model. Both the elasticity and proportional benefit analyses produced similar ranking of the life stages and magnitudinal impacts for single-species infestations. But the benefit analysis is more clearly aligned to the ultimate objective of managing multispecies infestations in a production system.

Growth and dynamics of multispecies infestations

Figure 4-5 and Figure 4-7 modelling results are consistent with Nobel and Slatyer (1980)'s observations:

Post-disturbance (post-control) the population structure and population dynamics is based on plants and propagules (seeds) that have survived the disturbance.

There is then a growth pulse, especially by the faster growing species that were previously outcompeted. The removal of all adults resulted in both prickly acacia and parkinsonia (both of which were declining) increasing their total area faster that chinee apple (which was dominant).

Following the growth pulse, the population will again tend towards steady-state (where the area is essentially fully occupied) and growth rates again decline. In Figure 4-5 and Figure 4-7, all the adults were targeted at different time periods (year 20, 100, and 250) and in all cases this was followed by a pulse recruitment of various species. Cochard and Jackes (2005) observed a similar phenomenon in a parkinsonia infestation, where the removal of a single large plant results in a mass recruitment of seedlings in the newly available area.

In the long run, chinee apple (which can live for ~ 80 years,) will dominate over prickly acacia which can live for ~50 years. Prickly acacia in turn dominates over parkinsonia which lives up to ~30 years. Even though chinee apple is slower growing than the other two species, it can dominate an area, as

once it occupies that area it locks up that space for the longest time. Moreover, as the plant grows it produces more seeds, allowing its propagules to occupy other areas that become available through natural succession processes.

There is additional information that Figure 4-3, Figure 4-5 and Figure 4-7 provide. Multispecies infestations will initially occupy an area as fast as the fastest growing species within the infestation. Therefore, they will tend to occupy areas faster than single-species infestations. These faster growing plants will tend to be the pioneer species, whilst the slower growing and long-lived plants will tend to be the successional species. Additionally, post-disturbance multispecies infestation will tend to recover faster, again due to the faster growing species within the infestation. In fact, these faster growing species can benefit from the control event which may reduce competition from longer-lived species. Both the faster occupancy and recovery rates of multispecies infestations have serious implications for weed management strategies. Firstly, the temporal benefit of controlling them is less than for single species infestations; secondly, weed managers must be more vigilant in monitoring areas that are susceptible to multispecies infestation. This is particularly true as the benefits of controlling the area post-establishment are low.

If it is uneconomical to control single-species infestations of the fast growing weed, then it is most likely that controlling any multispecies infestations with this fast growing weed will be uneconomical. It can therefore be assumed that the maximum benefit from managing multispecies infestations will be no greater than the benefit of managing a monoculture of the fastest growing species. Moreover, the cost of control may be higher for multispecies infestation as more resources may be required and efficacy may be lower. Chain-pulling provides an example. It is one of the most effective methods of control for adult prickly acacia plants (which are faster growing than chinee apple). However, chain-pulling is ineffective on chinee apple. Additionally, more effort may be required for pulling the larger chinee apple plants. Therefore, if for argument's sake it is uneconomical to control prickly acacia, it will also be uneconomical to control any multispecies infestation with prickly acacia in it, including a prickly acacia and chinee apple infestation.

Although multispecies infestations have faster occupancy and recovery rates, the growth rates of individual species within the infestation are often slower (Figure 4-5 and Figure 4-7). In Chapter 3 it is estimated to take singles species infestations of parkinsonia, prickly acacia, and chinee apple approximately 17, 23 and 70 years respectively to reach steady-state in riparian and 27, 37, and 150 years in upland zones. With the presence of other weeds it takes these weeds approximately 12, 100, and 330 years respectively to reach their maximum occupancy[10] in riparian zones and 20, 160, and 270 years in upland zones (Figure 4-3(a) and (b)). Although the presence of other competing weeds delays the impact of individual species, the aggregate damage is just as great.

In Figure 4-3, the multispecies infestation grows exponentially and then asymptotes within the first 50 years. At this point, it appears that the infestation is at steady-state, as damage is constant. However, chinee apple continues to increase its dominance within the infestations, resulting in both the total area occupied and weed damage increasing from $200< t <300$. This is due to this species having denser infestations at steady-state than the other two species, leading to greater damage (Figure 3-4). This modelling framework can be used to investigate the use of replacement plant species that may either have positive attributes or less damaging attributes (lower steady-state damage) than the species already there. Another point of interest for these case study species is that the introduction of prickly acacia or parkinsonia will have little effect on steady-state infestations of chinee apple, unless there is a disturbance where the population will recover more quickly. That is, the introduction of the additional species is only of concern if weed management is actively undertaken.

[10] When multispecies infestations reach steady-state, the area occupied and damage to pasture production is relatively stable. Individual species within the infestation may continue to increase or decrease their total area occupied but they may not reach the same level of occupancy as single species steady-state infestations. That is, in Figure 4-5 and Figure 4-7, the maximum area occupied by parkinsonia in a multispecies infestation is about half that for singles species infestations.

Integrated Weed Management

The results in Figure 4-5 and Figure 4-7 clearly show why integrated weed management (IWM) is important, as targeting of single species within multispecies infestations (as suggested in the elasticity analysis) offers less benefits. The simulations provided in this chapter highlight the benefits of controlling coexisting species. Although the proportional benefit analysis does not provide the optimal IWM policy, it ranks the species and life stages that have the greatest impact on the infestation. This information can be related to the efficacy of different control strategies to determine the true net efficacy (of both the kill rate and recovery from different methods of control) on multispecies infestations. The optimal economic IWM solution must consider the cost and efficacy of different methods of weed control as well as the financial returns from managing the infestation.

Whilst seeds have a low ranking once the infestation is established (after the first ten years) (Figure 4-13), they should still be considered within IWM strategies, as seeds are the only vector through which woody weeds can colonise new areas. Additionally, the removal of large adults will reduce fecundity and may reduce the spread rates into other areas.

This modelling framework can be further expanded for specific weed control methods (including their efficacy and cost) and risk. Both the efficacy and cost of control can be incorporated into the modelling framework through the expansion of Eq. 4.8. The population models presented here and in Chapter 3 are deterministic. To obtain a probabilistic analysis requires the addition of stochastic processes, such the effect of climatic conditions on the different species and their life stages. The inclusion of stochastic process also provides additional information to assist in the risk assessment of managing single and multispecies infestations within heterogeneous rangelands systems.

4.4 SUMMARY

A density-dependent multispecies SPM model was developed to explore the control and population dynamics of multispecies woody-weed infestations. The main objectives of this research were to rank the species and life stages with respect to impacts on these infestations and to investigate if this added complexity affects management decisions. Initially, an eigenvalue elasticity analysis was undertaken but this failed to identify any species other than the dominant species (and their life stages) at time *t*. However, controlling just one species within the infestation is inconsistent with the underlying premise of integrated weed management (IWM), which endeavours to manage different species simultaneously.

Various simulations were run to investigate the dynamics of the infestations and the effect of targeting different species and life stages. The results indicate that the elasticity analysis contributes little to weed management policies of multispecies infestations. An alternative analysis was developed that used the proportional benefit of controlling each life stage of each species within multispecies infestations. The multispecies infestation simulations and the proportional analyses showed that controlling any species or life stage will have an effect on the population. Moreover, there is no one species or life-stage that results in a major impact on the infestation. Generally, the greatest impact on multispecies infestations is through targeting the adults, then the juveniles and lastly the seeds. This is in line with commonly adopted weed management practices, which often use a single method of control to simultaneously target a main life-stage (i.e. adults) of all the species within the infestation. Whilst these results provide ecological information about IWM of multispecies woody weed infestations, they do not indicate the optimal economic IWM strategy.

This analysis indicates that there are less temporal benefits from managing multispecies woody-weed infestation than single species infestation. Additionally, control costs are likely to be higher and the efficacies lower for multispecies infestations, than singles species infestations. Therefore, if a single species weed cannot be controlled economically, nor can a multispecies infestation containing the same species within it be economically controlled.

5 CONTINUOUS AND EPISODIC RECRUITMENT OF WOODY WEEDS IN AUSTRALIAN RANGELANDS

The recruitment of woody vegetation within rangeland systems is complex. The process of recruitment includes: the dispersal, storage, and germination of seeds, within a heterogeneous landscape. There are several other drivers that effect woody recruitment: competition with established plants (density dependence), climatic events (i.e. droughts and floods), wildfires, and commercial grazing (cattle and sheep) (Higgins *et al.*, 2000; Kriticos *et al.*, 1999a; Brown & Carter, 1998; Watson *et al.*, 1997a; Carter & Cowan, 1993). Although all of these elements can be stochastic in nature, affecting weed recruitment to some degree, not all of them need to be considered by weed managers.

As results are not guaranteed, the management of woody weeds does involve risk. The efficacy of any control event will be dependent on the response of the weed population, which in turn is dependent on its stochastic environments. It needs to be said however, that the inclusion of stochastic processes into an economic optimisation of woody weed management should not be taken lightly. This is because stochasticity restricts the type of optimisation techniques that can be used, increases computing processing time, and adds another level of complexity to the economic solution. Two arguments can be put forward for the exclusion or inclusion of stochastic processes into an economic optimisation framework for woody weed management. It may be assumed that, if the larger portion of the population, being adult, is not greatly affected by climatic conditions, and they cause most of the weed damage, the system can be assumed to be deterministic. On the other hand, environmental conditions post-control can affect the lifespan of weed control benefits. If the mechanical removal of adults, for example, is followed by good climatic conditions for the weed, there will be a higher rate of recruitment. The infestation will recover sooner than normal, offering less temporal benefit from the control event. The opposite is also true for dry, post-control, climatic conditions. This is particularly true for longer lived species, where an additional plant can impact pasture production for many

decades. Therefore, the benefits of weed control are uncertain, and stochasticity may need to be considered by weed managers.

This chapter addresses two key research questions: (1) should stochastic processes be included in future economic optimisation research work for woody-weed management, including species not included in this study, and (2) does episodic recruitment affect weed management decisions, using chinee apple as a case study weed. Another problem is how to include stochastic processes into the density dependent SPM presented in Chapter 3. A number of studies have attempted to simulate the stochastic processes of woody plant recruitment within rangeland systems.

The following sections of this chapter will review the dynamic processes of changes within the Australian rangeland systems by reviewing: the effects of commercial stock, wildfires, drought and rainfall events, with respect to woody weed establishment within the rangeland landscape. This is followed by a review of woody plant recruitment models, to ascertain if they can be used directly or indirectly with the density dependent stage projection matrix developed in Chapter 3.

5.1 DYNAMICS OF RANGELAND SYSTEMS

There have been a number of theories and models developed in an attempt to explain the dynamic processes of Australian rangeland systems, including: commercial grazing, wildfires, and climate (Liedloff & Cook, 2007; Bond, 2005; Scholes & Archer, 1997). Part of the problem of observing rangeland ecosystems lies in long-lived plant populations showing little change from year to year (Radford *et al.*, 2002; Watson *et al.*, 1997a; Shreve, 1929). Therefore, it is not unexpected that many of these theories and models are based on anecdotal observations or one-off sampling of populations to estimate their history and dynamics (Watson *et al.*, 1997a). Until the 1970s, Clementsian succession was generally accepted as the theory for change processes in the Australian rangeland system (Watson *et al.*, 1997a; 1997b; Westoby *et al.*, 1989). This assumes that a given rangeland has a persistent state condition (the climax) where no commercial grazing is present and the rangeland rejuvenates linearly

to a point known as ‘the climax’. Grazing moves the system away from this climax in a linear fashion. An equilibrium can be achieved, therefore, between the vegetative state of the rangeland system and stocking rates for a given time period. This model was later adapted to incorporate variations in rainfall, by hypothesising that drought conditions are similar to grazing pressure in their effect on the vegetative state of the system. Therefore, a balance between the succession of the rangeland system and grazing pressure under drought conditions can be achieved to maintain a steady equilibrium of the vegetation (Westoby *et al.*, 1989). Since then, event-driven models of change have gained wider acceptance than gradual change processes of Clementsian succession (Watson, 1997; Watson *et al.*, 1997b).

5.1.1 Introduction of commercial stock

Brown and Carter (1998), using aerial photography, concluded that the encroachment of prickly acacia in the northern Australian rangelands is episodic. They determined that above average rainfall is necessary, but not sufficient, for significant prickly acacia expansion. Kriticos *et al*. (1999a) also supported the premise that the transition from sheep to cattle grazing has resulted in a rapid expansion of prickly acacia in its range within the Australian rangelands. It was hypothesised by Brown and Carter (1998) that this expansion can only occur when three key elements are present: (1) above average rainfall, (2) seed-producing adults, and (3) a highly effective dispersal agent such as cattle. They concluded that the prickly acacia infestation was episodic but primarily caused by a shift from sheep to cattle grazing rather than an increase in the frequency of abnormally wet seasons.

Prickly acacia pods are readily consumed by commercial stock, such as sheep and cattle. Although both types of stock graze similar habitat, cattle disperse seeds further. In cattle, up to 80 % of seeds survive the passage through the gut and are excreted within dung pats. In contrast, less than one per cent of seeds are viable after sheep eat the seed pods (Tiver *et al.*, 2001; Kriticos *et al.*, 1999a; Harvey, 1981). Passage through the gut also scarifies the seeds, increasing the probability of germination by 21 to 39 % (Tiver *et al.*, 2001; Kriticos *et al.*, 1999a; Barker *et al.*, 1996; Harvey, 1981). Seed dispersal

patterns of prickly acacia are correlated with cattle and sheep present in the landscape (Radford *et al.*, 2002; Kriticos *et al.*, 1999a; Mackey, 1998; Harvey, 1981). Sheep may destroy more seeds and are less effective at dispersal (Kriticos *et al.*, 1999a; Brown & Carter, 1998). However, Radford *et al.* (2002) found similar seedling numbers, tree mortalities, and growth vigour on both cattle and sheep properties. Other factors may therefore influence seedling establishment, including the browsing impact from stocking rates (Radford *et al.*, 2002) and competition with nearby prickly acacia seedlings (Kriticos *et al.*, 1999a).

Even if the transition from sheep to cattle is considered an event driver, it is not episodic. This transition can be considered a once-off event, as many of the properties persist with cattle grazing. Of the three elements suggested by Brown and Carter (1998), it is only the first (above average rainfall) which is truly an episodic driver, as the second and the third (the presence of seed producing adults and a dispersal agent) are ever-present in infested areas, representing once-off shifts or shocks to the system rather than episodic event drivers.

5.1.2 Rangeland wildfire events

Wildfire influences the composition of Australian rangelands (Noble, 1986) and drives the population dynamics of woody vegetation (Liedloff & Cook, 2007; Dyer & Smith, 2003; Carter *et al.*, 1991; Noble, 1986). Higgins *et al*. (2000, p.213) stated that: "Recruitment is controlled by rainfall, which limits seedling establishment, and fire, which prevents recruitment into adult size classes." Frequency of wildfires in the Australian rangelands is irregular, due to the variability of rainfall and the resulting grass fuel loads (Noble, 1986). Adding to the episodic nature of wildfire is the ignition source. In western New South Wales, lightning was responsible for over 75 % of wildfire ignitions from 1974 to 1975 (Noble *et al.*, 1997). However, land management practices have impeded the natural occurrence of wildfires and their influences on the vegetation dynamics of the rangelands.

Although fire is a key element in the dynamics of Australian rangelands, Liedloff and Cook (2007) showed that the effects of fire on tree population structure is secondary to rainfall events. Even though

wildfire is an episodic driver, it is excluded from the proposed episodic model for the following reasons. Wildfire predominately affects woody seedlings (Noble *et al.*, 1997) and as most woody seedlings will perish during an average dry season, the occasional wildfire will have little impact on woody-weed population dynamics. Wildfire may have a greater impact shortly after an episodic recruitment of woody germinants, killing off most of the juvenile woody cohorts. The impact of burning as a control method will be investigated in Chapter 7. If a wildfire does occur then it can be considered part of the management scheme. If it occurs when there is no episodic recruitment, it will have little effect, as most of the seedlings are killed during the dry season.

5.1.3 Drought events

Watson *et al.* (1997a) investigated the effects of episodic drought events on long-lived woody plants and concluded, like Gardiner (1986), that mortality rates of adults remained relatively constant, even during periods of drought. Watson *et al*. (1997a; 1997b) showed that droughts rarely result in mass mortalities among adult long-lived species within the Australian rangelands. It was found that woody populations decline by less than 20 % over periods of up to 100 years, whereas some short-lived vegetative species can become extinct from an area over the same time period (Bowers *et al.*, 1995). As yet, woody weeds such as chinee apple, prickly acacia, and parkinsonia have not been investigated to determine drought-stress mortality on adult plants within Australian rangelands. However, observations have indicated that, once established, adult plants are physically robust and can withstand significant drought stress (Lawes, 2004). In fact, they have remained a problem within the landscape despite periods of drought, possibly because they possess similar demographic characteristics to native woody vegetation, as they too are long lived, deep rooted, and can survive dry seasons.

5.1.4 Rainfall events

Climate is the primary driver of events in the Australian rangeland system (Lawes, Abbott, and Whiteman in press; Harrington *et al.*, 1984b). Whilst temperature is relatively reliable, rainfall is

highly variable and unpredictable in most rangeland habitats (Harrington *et al.*, 1984b). In northern Australia there are annual climatic events called wet and dry seasons, which result in seasonal recruitment and cessation of plants (Furley, 2006; Harrington *et al.*, 1984b). However, episodic climatic events also occur, resulting in either above or below average rainfall. In many cases, episodic events resulting in increased recruitment and establishment of woody plant cohorts have a greater impact on their population dynamics than increased mortality (Watson *et al.*, 1997a).

A single establishment of long-lived woody plants can change the rangeland system for decades or centuries (Harrington *et al.*, 1984b). Woody plants are slow growing and rangelands are both vast and heterogeneous. Consequently, it may be some time after the episodic event that cohort establishment is recognised (Harrington *et al.*, 1984b). Therefore, rangeland systems operate on very long time scales and the impacts of events last well beyond that of our memories or observations of the initial event (Harrington *et al.*, 1984b). These systems have been occupied by settlers for little over a hundred years, therefore insufficient time has passed to collate data and test hypotheses that capture the impacts of these episodic processes (Harrington *et al.*, 1984b). Episodic climatic events principally affect the germination and survival of seedlings and drought mortalities (Higgins *et al.*, 2000; Kriticos *et al.*, 1999a; Watson *et al.*, 1997a).

Temporal climatic conditions for the germination and establishment of long-lived woody plants, such as chinee apple, prickly acacia, and parkinsonia in the Australian rangelands, tend to be both seasonal and episodic (Jurena & Archer, 2003; Radford *et al.*, 2002; Kriticos *et al.*, 1999a; Brown & Carter, 1998; Watson *et al.*, 1997a). Episodic recruitment and establishment of woody plants involves two components. In the first instance, the climatic conditions propagate a mass germination of seeds. In the second, climate conditions must continue to sustain seedlings and ensure they survive long enough to establish themselves to endure future dry periods (Lawes, 2004).

5.1.4.1 Germination of seedlings

Germination of woody plant seedlings is relatively frequent, especially in northern Australian rangelands systems where germination coincides with wet seasons (Kriticos *et al.*, 1999a). Therefore, germination can be considered a serial event of seasonal occurrence (Kriticos *et al.*, 1999a; Watson *et al.*, 1997a). However, under normal rainfall conditions not all seeds within the seedbank will germinate. Occasionally, when there is above average wet-season rainfall, a higher proportion of the seedbank will germinate (Lawes, 2004; Kriticos *et al.*, 1999a; Grice, 1998; Brown & Carter, 1998; Watson *et al.*, 1997b). The magnitude of episodic recruitment during the wet season can be related to the amount of rainfall.

5.1.4.2 Establishment of seedlings

Seedlings often germinate during moist conditions in the wet season. However, seedlings must also survive a dry season; they rarely do so. Occasionally though, climatic conditions allow seedlings to survive. Once established, they become quite resilient to dry climates and are long lived (Liedloff & Cook, 2007; Wellington, 1989; Harrington *et al.*, 1984b). Many woody seedlings will fail to survive their first dry season (assuming average rainfalls). Parkinsonia is a good example. Seedlings in a glasshouse trial died within 3 to 5 weeks under dry-season conditions (Lawes, Abbott, and Whiteman in press). However, rangeland climate and terrain are heterogeneous and it is possible for some woody seedlings to survive their first year. Seedlings that germinate and survive average rainfalls for both the wet and dry seasons can be considered a serial event, which will be called “background recruitment” (Brown & Archer, 1999; Watson *et al.*, 1997a).

Many woody weeds including chinee apple, prickly acacia, and parkinsonia are deep rooted perennials. Although grasses and woody plants compete for soil moisture in the upper zone (Scholes & Archer, 1997; Leroux *et al.*, 1995), deep-rooted plants can access deeper soil moisture during drier periods. Further investigation into competition between grasses and woody plants has been undertaken in

tropical (Brown *et al.*, 1998; Knoop & Walker, 1985), subtropical (Brown & Archer, 1990), savannas and temperate grasslands (Booth *et al.*, 2006; Danner & Knapp, 2001; Bragg *et al.*, 1993). However, two-layered soil competition does not explain the establishment of seedlings in the upper soil zone, being dominated by grasses (Jurena & Archer, 2003). Further research indicates that seedlings significantly increase their chance of survival by rapidly extending their tap-root through the upper soil zone in their first year, where competition with grasses for resources is greatest, towards the lower moisture zone (Jurena & Archer, 2003; Danner & Knapp, 2001; Brown *et al.*, 1998; Brown & Archer, 1990). Drought tolerance of a woody plant markedly increases once it has endured its first dry season (Brown & Carter, 1998; Brown & McIvor, 1993).

5.1.5 Further evidence of episodic events in the rangelands

Radford *et al.* (2002) investigated seedling establishment, mortality, tree growth and vigour of prickly acacia in the Mitchell grasslands, near Hughenden and Julia Creek, in northern Australia (from 1996 to 1998). Although this study did not specifically look at the episodic events of climatic conditions, the results can give an insight into episodic effects. Radford *et al.*, (2002, p.266) concluded that:

> *If rainfall is the trigger for invasion/expansion events, such events must be extreme, because relatively high rainfall (>600 mm in the 1996-1997 wet season at both Garomna and Allaru, compared with an average annual rainfall of approximately 470 mm) during this study did not lead to significant growth, seedling establishment or reduction in mortality rates.*

Although the rainfall for the 1997 wet season was above average, the following dry season was particularly harsh, with no rainfall recorded by the Bureau of Meteorology at the Richmond Postal Office (Lat 20^{O} 44'S, Long 143^{O} 09'E) for five months (during the dry season) (BoM, 2008). What needs to be considered is not only the effect of the wet season on germination but also the germinants survival through the following dry season.

Radford *et al.* (2002) also concluded that, based on the climatic conditions experienced during the study, populations of prickly acacia will continue to decline in these regions. However, the fact that this species was sparsely distributed within the landscape less than 40 years ago and now occupies large areas of the landscape (Brown & Carter, 1998), clearly demonstrates that it has the ability to

increase its frequency and abundance in this landscape. Field observations in other areas also indicate that high tree densities of prickly acacia are possible in both riparian and upland zones and in some areas are increasing in density (Radford *et al.*, 2002; Pratt & Knight, 1971). However, Radford *et al*. (2002) were uncertain if these results typified the long-term population dynamics of prickly acacia.

5.2 REVIEWING PLANT RECRUITMENT MODELS

Watson *et al.* (1997a), Liedloff and Cook (2007), Higgins *et al*. (2000), and Lawes (2004) all attempted to capture the episodic processes of long-lived woody germinants in rangeland systems, with varying degrees of success. All authors used rainfall as the primary driver for recruitment events, although each study postulated different recruitment processes.

5.2.1 Watson's episodic population model

Watson *et al.* (1997a) sought to model the continuous and episodic recruitment of two Australian rangeland long-lived shrubs, *Eremophila maitlandii* and *E. forrestii*, based on the results of a long-term sheep grazing study (1983 to 1993) in the Western Australian rangelands. Stage projection matrices were used for the simulation. Density dependence was not integrated into this population model. Population structure was divided into recruits, short, medium, and tall adults. There were two main components of woody vegetation recruitment in this model: one being annual background recruitment and the other being event-driven (rainfall) germination. The model was based on 48–year cycles, with various combinations of episodic mortality and recruitment events. The frequency of episodic events ranged between one and four mortality episodes and two and eight recruitment episodes per 48 years. The magnitude of the episodic recruitment was constant (i.e. not dictated by the rainfall). The model used two primary drivers of the system: background processes and long-term climatic episodic events.

5.2.2 Liedloff and Cook's episodic population model

Liedloff and Cook (2007) developed a process-based tree population dynamics model (Flames), which included fires and rainfall variability. The model operated on daily time steps, tracing individual trees (eucalypts) in a mixed-species population per hectare to simulate landscape processes. The model used soil properties, plant water usage and daily rainfall data, as well as stochastic processes based on the variable effects of tree-fire interactions. It was assumed that most seedlings (99 %) perished due to seasonal droughts (Liedloff & Cook, 2007; Wellington, 1989) and no burnt seedlings survived. Density-dependence was based on total basal area (TBA) per hectare. The simulated results showed that increased average rainfall increased TBA, as did the decrease in the variance in monthly rainfall. Although Liedloff and Cook (2007) acknowledged the impacts seasonal droughts have on seedlings, the stochastic processes of their model were only based on the effects rainfall variation had on fuel loads for fire. The model did not explicitly consider the episodic recruitment and establishment of woody plant cohorts due to favourable climatic conditions.

5.2.3 Higgins' episodic population model

Higgins *et al*. (2000) developed a grass-tree interaction model, where the probability of germination and establishment of woody plants was primarily driven by rainfall. Relationships between rainfall and germinant establishment and survival were multifaceted. Firstly, it was assumed that seedlings did not survive the wet season if drought conditions persisted for more than 30 days. Secondly, increased rainfall resulted in greater grass biomass, which resulted in increased competition for resources and decreased woody seedling establishment. Finally, fire intensity, which was also a function of rainfall and grass biomass, affected germinant survival rates. The model not only considered seasonal conditions for germinant recruitment and establishment but also included the effect of annual rainfall.

The primary driver, rainfall, had two components (1) stochastic variability in the rangeland system and (2) a long-term cycle (20 years). However, the model did not explicitly consider drought effects on

seedling survival during the dry season. Moreover, Higgins *et al.* (2000, p.226) concluded that the answer to understanding the dynamics of plant communities in savannas:

> *Lies in stepping back from the details of fine-scale interactions between grasses and trees and observe the longer term effects of disturbance, life history (Noble & Slatyer, 1980) and regeneration (Grubb, 1977) on demography.*

Higgins *et al.* (2000) supported the premise that episodic rainfalls were a significant contributing factor in the establishment and survival of woody germinants in the Australian rangelands, resulting in sudden increases in woody vegetation followed by long periods of gradual decline. However, similar to the Watson *et al.* (1997a) model, it was not a truly episodic simulation, as the event driver for recruitment occurred with predicted periodic frequencies.

5.2.4 Lawes' episodic germination and survival model

Lawes (2004) developed a model to predict frequency of seedling germination and establishment of parkinsonia based on rainfall data from Charters Towers, Alice Springs, and Darwin between 1950 and 2000. The results were based on a glasshouse experiment using sand, loam, and cracking clay soils which broadly represent the Australian rangeland soil types. The model was based on daily soil moisture. Assumptions of this model were (Lawes, 2004, p.2):

1. Germination occurs if soil moisture reaches 80 % of water holding capacity.
2. Seedlings will survive the following four weeks if the soil remains moist every day of this period.
3. Seedlings will survive for four months after germination if there is at least 166 mm of rain during this period.

If all three conditions are not met, it is assumed the seedlings will not germinate and survive the four month period. These three conditions will be referred to as Lawes' Rules of germination and establishment. Rule 1 is clearly related to the germination of seedlings, whereas Rules 2 and 3 relate to the establishment of plants.

5.2.5 Rainfall events for Charters Towers – northern Australia

To give an indication of seasonal trends and variability of rainfall in the northern Australian rangelands, Figure 5-1, provides the actual and mean monthly rainfalls for the Charters Towers district from 1889 to 2007 (BoM, 2008). There are clear annual periods of high rainfall called the wet season and drier annual periods called the dry season. This figure also gives some insight into the episodic nature of rainfall events. For example, in 1998 there was an above average rainfall for the wet season followed by a very wet dry season, giving the opportunity for seeds to germinate and establish themselves. In contrast, the 2004 wet season had an average rainfall, but was followed by a xeric dry season, giving seedlings little chance for establishment.

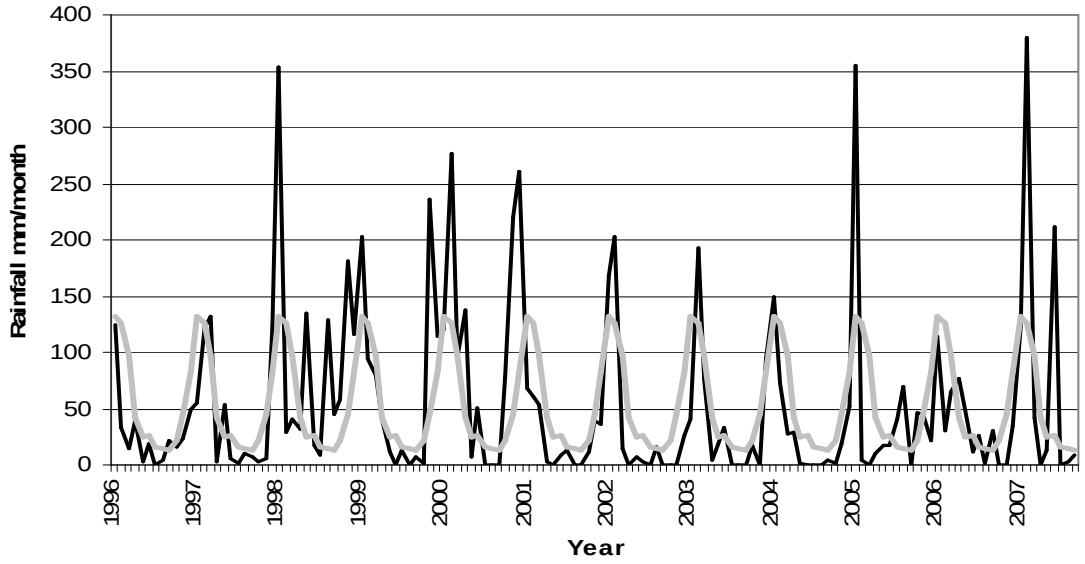


Figure 5-1. Monthly (—) and average monthly rainfall (—) recorded in Charters Towers between 1889 and 2007, extracted from the SILO Data Drill database (BoM, 2008).

5.2.6 Evaluating Lawes' Rules of woody recruitment

The following section will use Lawes' Rules, for woody-weed germination and establishment in conjunction with the historical rainfall and evaporative data from the Charters Towers region between 1889 and 2007 (BoM, 2008). The formulae used in a spreadsheet are presented in Appendix 1: Calculations of Lawes' Rules. The additional information used from Lawes (2004) includes the assumptions that soil is 200 mm deep and holds up to 42 mm of water that evaporates at a rate of 30%

of daily recorded evaporation. The daily results from using Lawes Rules are grouped into months, to estimate the probability of seedlings germinating and surviving the next six months. These results are given in (Figure 5-2). There is a clear relationship between the probability of Lawes' Rules being satisfied and the average monthly rainfall. Results presented in (Figure 5-2) are similar to those of Lawes (2004), which were based on rainfall for the same area but only from 1950 to 2000. From the data presented in Figure 5-2, it is assumed that the wet season is from November to March and the dry season is from April to October. Based on Lawes' Rules and the historic rainfall data of the Charters Towers region (1889–2007) there is a 28 to 75 % chance of parkinsonia seeds germinating in the wet season and the seedlings surviving for four months. The probability of germination and survival decreases during the dry season to between 2 and 15 %.

The probability of germination and survival using Lawes' Rules is increased if future months have higher rainfalls. August and September, which have the lowest average rainfall, have a higher probability of germination and survival than June and July, even though their average rainfall is higher. Although the data in Figure 5-2 have different scales, they can give an indication of trends. The months from August to December all have probabilities of germination and survival above the average monthly rainfall line, as future months have high rainfalls, whereas the months from January to July are all closer to, if not below, the average monthly rainfall line. This indicates that rainfall in future months will affect the survival probability of seedlings germinated in the current month. Lawes' Rules can be broken down into two key requirements: (1) germination, which is a function of the rainfall in the current period and (2) survival, which is a function of rainfall in future periods.

Lawes' Rules need not be met in every month in order for the infestation to survive or even expand. The predicted seedling establishment rate for the various months includes the probability of germination but this is not necessary for the survival of the already semi-established plants. For example, in 117 years of rainfall data from Charters Towers (1884 – 2001), there was not one year in which Lawes' conditions were met for all 12 months (Figure 5-3). Yet infestations of parkinsonia, along with a number of other weeds, are increasing in this region (Lawes *et al.*, 2003).

Clearly, there is a strong relationship between the frequency that Lawes' Rules occur (Figure 5-3) and the magnitude of annual rainfall (Figure 5-1). Above average rainfall in 1998 and 2000, for example,

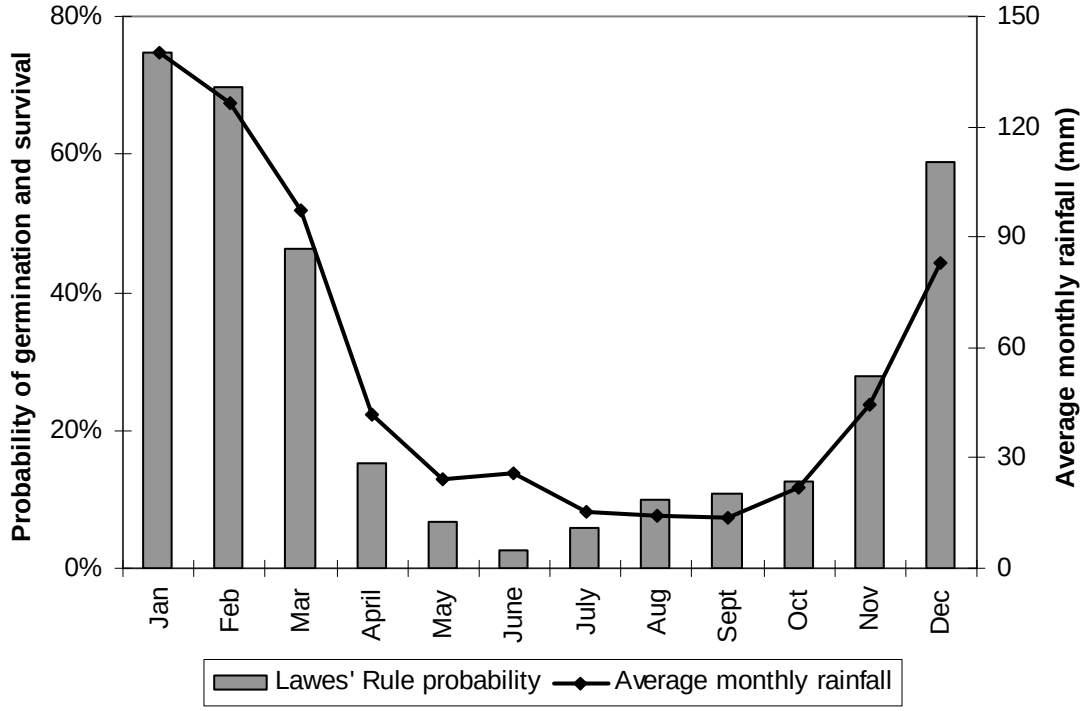


Figure 5-2. Probability of Lawes' Rules for the germination and survival of parkinsonia occurring in Charters Towers; solid line indicates monthly rainfall (—), based on rainfall and evaporation data from 1885 to 2007 (BoM, 2008).

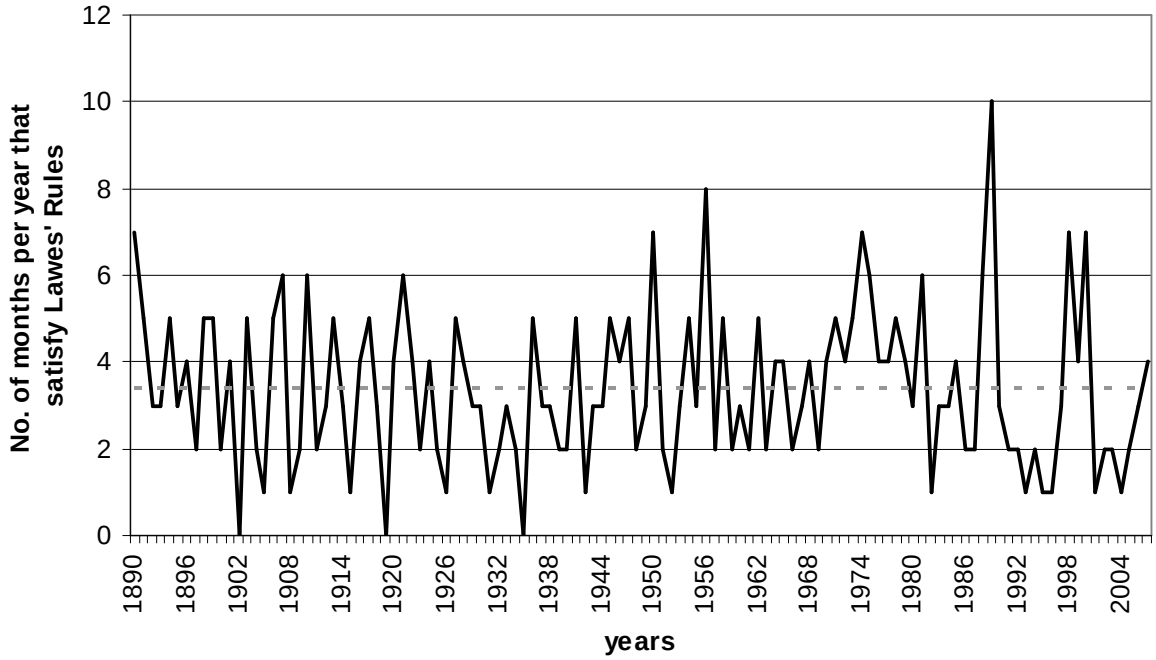


Figure 5-3. The number of months per year that Lawes' Rules are satisfied for the germination and survival of parkinsonia in Charters Towers, based on rainfall and evaporation data from 1885 to 2007 (BoM, 2008). The dashed line indicates that the average number of months per year that Lawes' Rules are satisfied.

resulted in Lawes' Rules being satisfied for seven months in each year, whereas the rules were only satisfied for one month during drier years (e.g., 2001 and 2004). The other point of interest is the distribution of Lawes' Rules, which is somewhat evenly around the average frequency (see dotted line in Figure 5-3). It can be assumed that episodic recruitment of seedlings would have a more skewed distribution, as the likelihood of large recruitment is infrequent. Neither Figure 5-2 nor Figure 5-3 indicate the number of times Lawes' Rules had been met within a month, which would also contribute to the magnitude of the episodic recruitment. Figure 5-4 shows that the occurrence of Lawes' Rules being met per-year increases as the magnitude of rainfall increases. Additionally, the distribution of germination and survival, using Lawes' Rules is now skewed. Another point of interest is that, even when there is below average rainfall, Lawes' Rules can still be satisfied. This indicates background recruitment processes in every year, as hypothesised by Watson *et al.* (1997a).

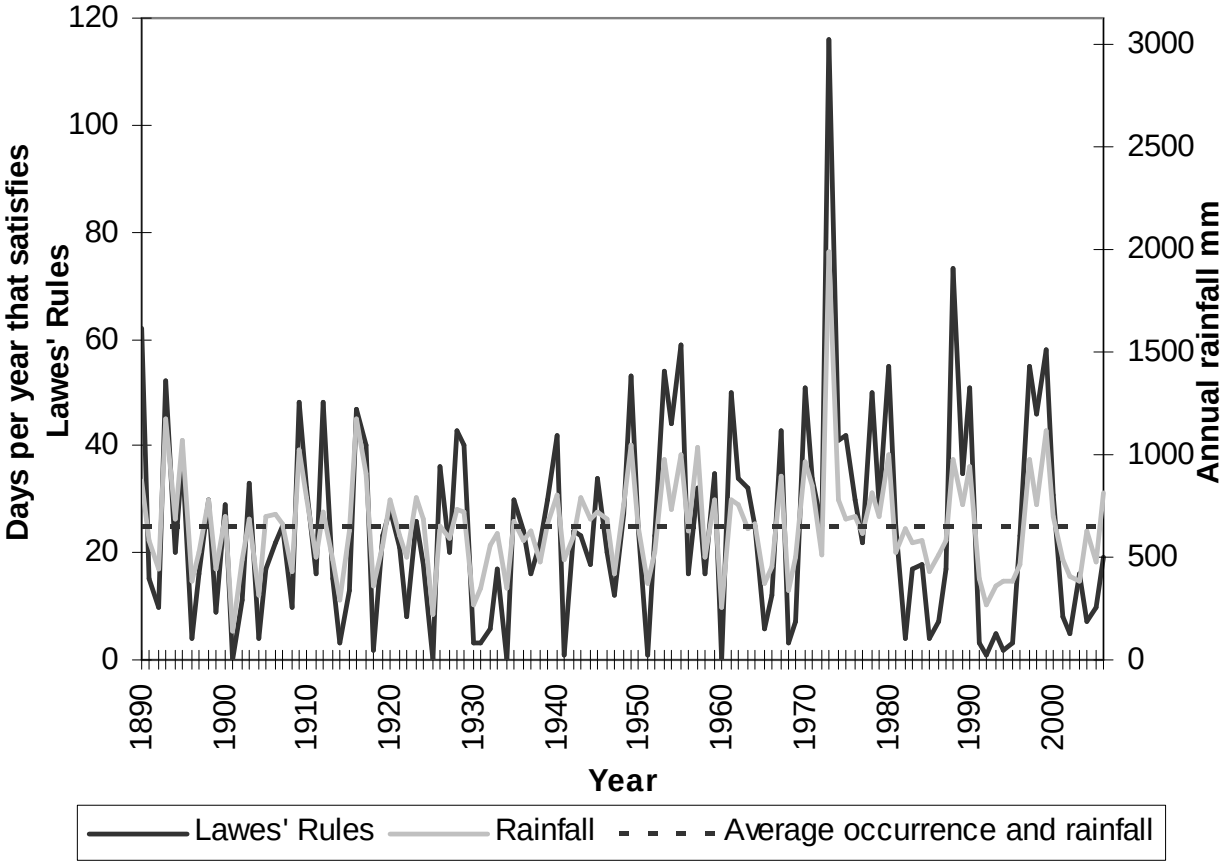


Figure 5-4. Frequency per year of Lawes' Rule for the germination and survival of parkinsonia occurring per year, compared with annual rainfall in Charters Towers, based on rainfall and evaporation data from 1885 to 2007 (BoM, 2008).

The shortfall of Lawes' Rules is that they do not categorically determine if germinants survive their first dry season, which is when mortality rates are highest. Although Lawes Rules require the seedling to survive for four months, this could be in the beginning of the wet season and the plant could still perish in the following dry season, as most woody-weed seedlings do. Therefore, Lawes' Rules are necessary but not sufficient to determine germination and survival of an individual into the following wet season, when they are established enough to be resilient to dry climates (Liedloff & Cook, 2007; Wellington, 1989; Harrington *et al.*, 1984b). Figures 5-2, 5-3, and 5-4 indicate that using monthly or seasonal rainfall may result in a more elegant model, for the establishment and recruitment of woody-weed seedlings, than that provided by Lawes' Rules. This new approach of modelling woody-weed recruitment within rangeland systems will also include the likelihood of seedlings surviving the first dry-season, which is not considered in Lawes' Rules.

5.3 METHODOLOGY OF A SIMPLE STOCHASTIC RECRUITMENT MODEL

There have been various definitions of what constitutes an episodic woody plant recruitment event (see Watson *et al.* (1997a), Watson (1997); Brown and Carter (1998), Kriticos (1999a), and Lawes (2004)). Although varying in definition, they all use the premise that an episodic recruitment event of woody plant seedlings is reliant on above average rainfall. Therefore, germination and establishment that occurs with average (or below) rainfalls are considered background recruitment (Brown & Archer, 1999; Watson *et al.*, 1997a). Any recruitment which is greater than background (average) recruitment is deemed episodic recruitment. The greater the number of weed recruits that survive the year (a wet and dry season), the greater the magnitude of the episodic recruitment event.

Cook and Heerdegan (2001) suggested that rangeland plant communities are more sensitive to the seasonal distribution of rainfall than to total annual rainfall. The wetter the wet season, the greater the probability of Lawes' Rule for germination being met and the wetter the dry season, the greater the probability that Lawes' Rules for establishment will be met. Therefore, for an episodic recruitment

event to occur, above average rainfall is required for both the wet and dry seasons. The greater the amount of rainfall within these seasons, the greater will be the magnitude of the episodic recruitment event.

It is assumed that, once a woody germinant survives its first dry season, it has significantly increased its chances of reaching adulthood (Liedloff & Cook, 2007; Wellington, 1989; Harrington *et al.*, 1984b). Therefore, only one year's rainfall (a wet and dry season) will be included in episodic recruitment model.

5.3.1 Modelling woody-weed recruitment

Within the northern Australian rangelands, the wet season is from November to March and the dry season is from April to October. The magnitude of the episodic climatic event (ψ_t) at time (t) is given by:

$$\psi_t = \frac{W_t D_t}{\bar{W}\bar{D}} \tag{5.1}$$

W_t and D_t is the rainfall for wet and dry seasons, respectively; and $\bar{W}$ and $\bar{D}$ is the average wet and dry season rainfalls, respectively. When $D_t = \bar{D}$ and $W_t = \bar{W}$, $\psi_t \leq 1.0$ = 1.0, and no episodic recruitment effect occurs. If $\psi_t > 1$, there is an episodic event. The greater the value of ψ_t, the greater the episodic magnitude. If $\psi_t < 1$, there will be fewer cohorts germinating and surviving their first dry season. The two seasons can also balance one another. For example, if there was a hydric wet season followed by xeric dry season, then there will more cohorts germinating $\left(W_t / \bar{W} > 1\right)$ but a lower proportion of cohorts surviving the dry season $\left(D_t / \bar{D} < 1\right)$. Conversely, if there was a xeric wet season followed by a hydric dry season, there will be few germinants but a larger proportion of them will survive their first dry season. As there are no negative variables in Eq 5.1 the magnitude of recruitment episodic events will be $0 \leq \psi_t < \infty$. Therefore, some plants may germinate and survive

every year, even when the rainfall for both the wet and dry seasons are below the average, and these will be regarded as background processes independent of episodic events. This provides a model for not only the occurrence of an episodic event but also the magnitude of events for recruiting woody germinants, relative to the long-term average germination rate for that location.

5.3.2 Contrasting model results

The episodic recruitment event model was applied to the rainfall data of Charters Towers (1890–2007) (Figure 5-5). The average episodic magnitude was $\overline{\psi} = 1.0287$, and resulted in a skewed distribution (Figure 5-6), with the majority of episodic magnitudes below the average ($\overline{\psi} = 1.0287$). As with Lawes' (2004) results, there will be some germination in most years owing to background recruitment processes (Watson *et al.*, 1997a). Another way of interpreting the model is when ψ_t = 0.5, only half the average number of seedlings will germinate and survive their first dry season. Likewise, when ψ_t = 4.0, there will be four times the average number of seedlings germinating and surviving their first dry season. As $\psi_t > 1.0$, it will be considered an episodic recruitment event. In the early 1970s there was a mass recruitment of prickly acacia in the northern Australian rangelands (Radford *et al.*, 2002), this was captured by the model, estimating 1974 with $\psi_t > 4.0$. The model results also indicated that in 1890 there was a similar recruitment event, but no evidence could be found in the literature of such an event, due to it being so long ago. As expected for episodic factors, the results are skewed with 62 % of the years having no episodic recruitment events ($\psi_t \leq 1.0$) and 81 % of the years having little episodic recruitment effect $\psi_t \leq 1.5$ (Figure 5-6).

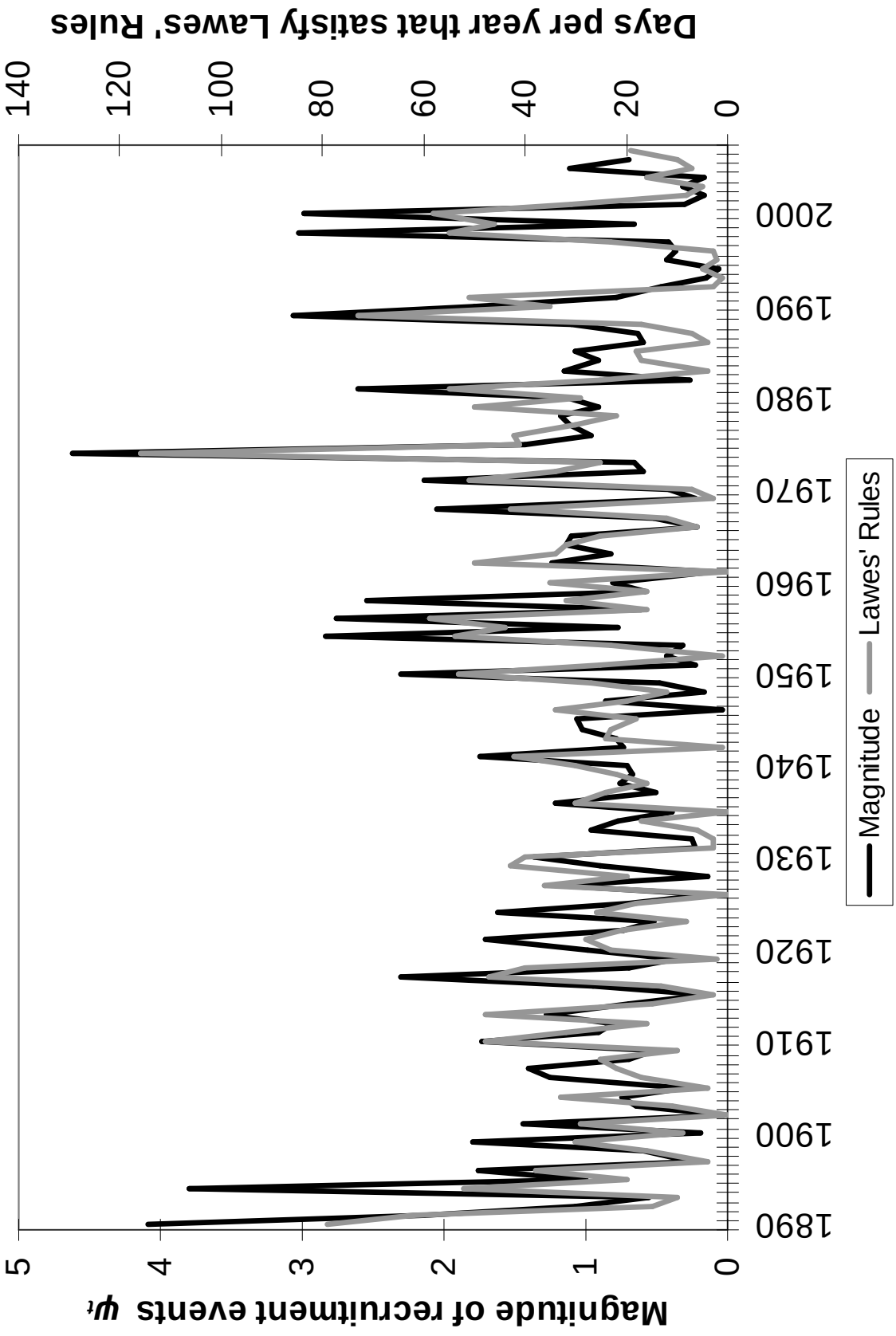


Figure 5-5. Results of applying both the episodic recruitment model and Lawes' Rules based on Charters Towers rainfall and evaporation data from 1885 to 2007 (BoM, 2008).

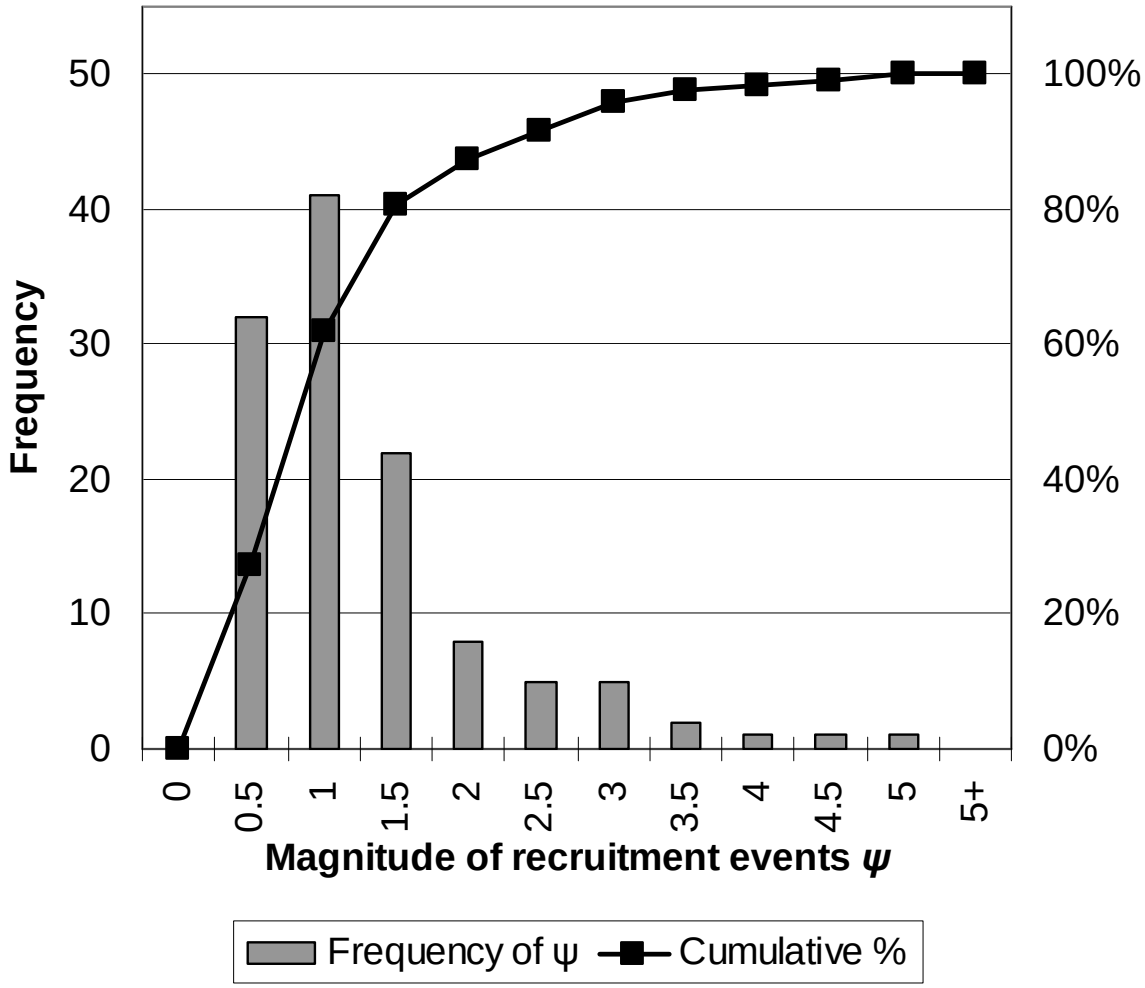


Figure 5-6. Histogram of frequencies of recruitment magnitudes and the cumulative percentage curve, based on Charters Towers rainfall and evaporation data from 1885 to 2007 (BoM, 2008).

5.3.3 Modelling episodic recruitment

Episodic events primarily affect juveniles (Lawes, 2004; Kriticos *et al.*, 1999a; Grice, 1998; Brown & Carter, 1998; Watson *et al.*, 1997b) and have little effect on long-lived adult plant survival (Watson *et al.*, 1997a). Therefore, the modelling of episodic events will be aimed at the number of germinants that survive until t+1. It is assumed that an episodic recruitment will affect all species of woody seedlings in a similar manner. That is, if an episodic recruitment event occurred and $\psi_t = 2.0$ then there will be twice as many germinants of all woody species than there otherwise would have been in that year. However, due to the time it takes for juveniles to reach adulthood and the longevity of the different species, an episodic recruitment event will affect the population dynamics of the different species to varying degrees. Additionally, due to the density-dependent component of the population model, the

episodic recruitment event will have less effect on a steady-state population than on new and recovering infestations. For example, if weed management removed all the adults from the population in the year prior to an episodic recruitment event, there will be less competition. This results in an increase of the surviving cohorts. Conversely, if management of adults occurred prior to a xeric year, most of the seedlings will die and benefits of the weed management will be enhanced. With this in mind, a land manager may decide to target a population a year or two after the episodic event, with the knowledge that the population has increased its potential to have an effect on pasture production.

Serial correlation of the historical magnitudes of recruitment events was tested with the Durbin-Watson test using SHAZAM 10.0® (2004) software on the outputs in Figure 5-5. This resulted in DW = 2.0001, indicating no serial correlations existed for recruitment between the years. Rainfall data from the Charters Towers district gives 114 observations and episodic probabilities. As there is no serial correlation of annual rainfall event, a Monte Carlo random sampling algorithm can used to select the annul rainfall events from the Charters Towers rainfall data from 1885 to 2007 (BoM, 2008). This permits the modelling recruitment simulations to extend the simulations beyond 114 years of historic data.

Only a small portion of the seedbank is assumed to germinate in any given year (being less than five per cent), and the largest episodic recruitment magnitude (ψ_t) is 4.6. Therefore, it is safe to assume that the number of germinants modelled in any year will not exceed the number of available seeds. The episodic population model is given as

$$\mathbf{x}_{t+1} = (\mathbf{H}_t \mathbf{x}_t) \circ \Upsilon_t \tag{5.2}$$

where x_t is the population vector, $\mathbf{H}_t$ is the density-dependent SPM (developed in Chapter 3), and Υ_t is a vector (equal in length to the population vector) of ones except for the third element which represents the germination and survival of seedlings, and takes a random value (ψ_t) from data in Figure 5-5.

5.4 RESULTS FROM PROPOSED RECRUITMENT MODEL

To contrast the impacts of episodic recruitments and subsequent damages of different woody weeds to grazing production in the Australian rangelands, both episodic and deterministic simulations were run for chinee apple, prickly acacia, and parkinsonia in riparian and upland zones for new and steady-state infestations (Figure 5-7). Stochastic simulations were generated using the stochastic density-dependent SPM model Eq 5.2. Deterministic simulations used a similar model without the stochastic element ($\mathbf{x}_{t+1} = \mathbf{H}_t \mathbf{x}_t$) as presented in Chapter 3. Damage to pasture production was calculated with the rectangular hyperbola function from Chapter 3 (Cousens, 1985a). For parameter values see Chapter 3.

New and steady-state infestations can be representative of the edges and the centre of an infestation in a similar location and time. Episodic recruitment events were randomly selected from those presented in Figure 5-5, which are based on the rainfall data of the Charters Towers region. All the simulations for new infestations started with 2000 seeds per hectare.

Although only the earlier years of the models are presented in Figure 5-7, the actual simulation was run for 2000 years. Episodic damages drifted around the deterministic steady-state damage line. The model captures perception of long-lived rangeland plant populations being in constant decline, punctuated with episodic recruitment events (Radford *et al.*, 2002). On average, there are 35 % more declining years than increasing years. To derive the ratio between increasing and decreasing damage (*D*) over time, the following ratio function was used, $(D_t > D_{t+1} / D_t < D_{t+1}) - 1$. The damage from these simulations indicate that chinee apple, prickly acacia, and parkinsonia had 38%, 32%, and 23% more declining years than increasing years within riparian zones, with 44%, 45%, and 31% in upland zones, respectively. Drifting of damage was due to three components of the model working simultaneously: (1) a decline in the population due to the limited recruitment opportunities, (2) episodic recruitment due to favourable climatic conditions, and (3) the dampening effect of these two opposing forces due to density-dependence.

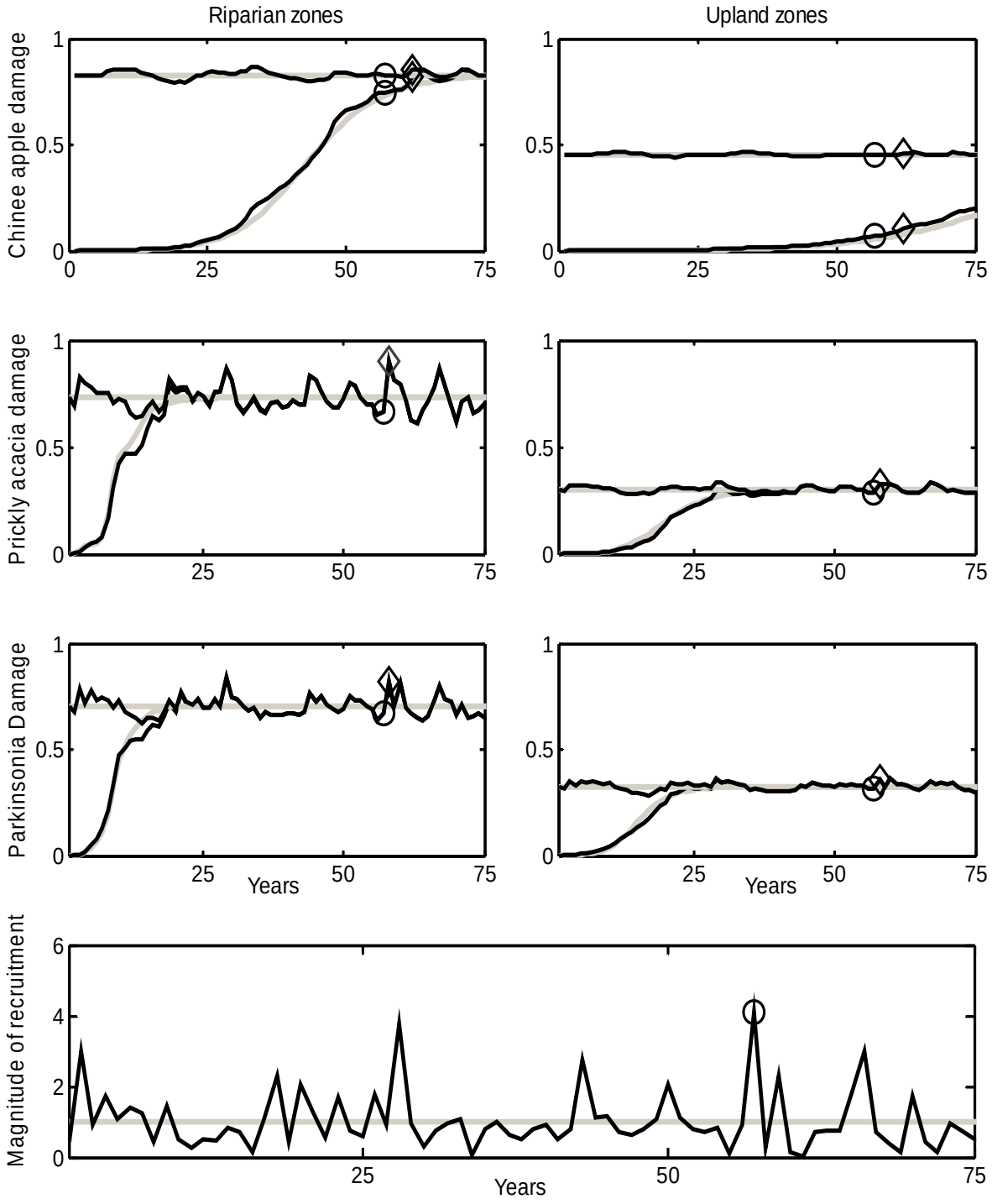


Figure 5-7: Episodic (——) and deterministic (——) population simulations of new (lower initial damages) and steady-state (higher initial damages) infestations of chinee apple, prickly acacia and parkinsonia in riparian and upland zones. All simulations are based on the same randomly selected rainfall events from the Charters Towers district. The magnitude of episodic recruitment events used for the simulations is given in the lower chart. To compare the effects of episodic events, the recruitment event in year 57 (○), is the same as the 1974 rainfall (recruitment) event, the resulting growth pulse for the different infestations is indicated by (◇).

An episodic recruitment event in year 57, highlighted by circles in Figure 5-7, resulted in a peak in chinee apple, prickly acacia, and parkinsonia damages in years 62, 58, and 58 respectively, highlighted by diamonds. Damages of recruitment events varied, depending on the location and plant species. Although the timing of impacts from episodic recruitment events was similar in riparian and upland zones, the variance of weed damage from recruitment was higher for all three species in riparian zones. Based on 2000 iterations of randomly selected rainfall data and 2000 year simulations, the coefficient of variation (CV) for damage from chinee apple, prickly acacia and parkinsonia was 3.8, 9.2, and 7.5% in riparian zones and 3.9, 5.7, and 6.6% in upland zones, respectively. Chinee apple, the longest lived species, had the lowest overall CV. All the simulations in Figure 5-7 started with 2000 seeds per hectare.

By increasing the initial seed populations, the temporal impact of the infestation will occur sooner, with the probability of establishment being higher. Figure 5-8 shows the estimated damages from the recruitment events of the different weed species in riparian and upland zones 50 years after the introduction of seeds into an area for given quantities of seed. To generate the damage distribution, 1000 iterations were run for each initial quantity of seeds using a Monte Carlo selection criterion to obtain different random values (ψ_t) for seedling recruitment. The number of seeds introduced to the system were increased exponentially, starting with one and doubling sequentially to just over a million seeds ha^{-1} (i.e, 1, 2, 4, 8,…, 1 048 576). Whilst the damages from chinee apple at year 50 are correlated to the quantity of initial seeds, the same cannot be said for prickly acacia or parkinsonia in either zone (Figure 5-8). It is only when the initial chinee apple seed population exceeds 100 000 and 1000 000 seed ha^{-1}, for riparian and upland zones respectively, that additional seeds do not affect grazing production.

The above simulations assume that the modelled hectare is indicative of an average hectare and consequently the results are continuous, i.e., a portion of a tree can exist. For example, if the model at a

given time has 0.5 adults, then this can be taken as one adult per two hectares. As the model results are continuous, eradication is unlikely. For example, weed management will tend to drive the weed population density to zero, which will result in the different life stage tending towards ∞^{-1}. However, as the modelled results are continuous, a portion of a seed or a plant will continue through the life-cycle stages and the population will always tend to an asymptote over time. In reality, eradication and non-establishment is possible. It will be assumed that at 5 years after the initial introduction of seeds, a minimum population is required for the infestation to be considered established. That is, one of the following must exist in order for establishment to have occurred: >1 seed ha^{-1} or in any life-cycle stage there are at least >0.1 juvenile or >0.01 adult ha^{-1}. This represents at least one seed per hectare and/or one juvenile per 10 hectares, and/or one adult per 100 hectares.

Based on the 2000 model iterations, the probability of prickly acacia and parkinsonia establishing is insensitive to the initial number of seeds in riparian zones (based on the given criteria of establishment), as 11 seeds ha^{-1} is sufficient to result in a 100 % probability of establishment (Figure 5-9). In the upland zone, this increases to 100 seeds ha^{-1}. In contrast, chinee apple is more sensitive to initial seed quantities requiring 300 and 500 seed ha^{-1} for certain establishment in riparian and upland zones, respectively.

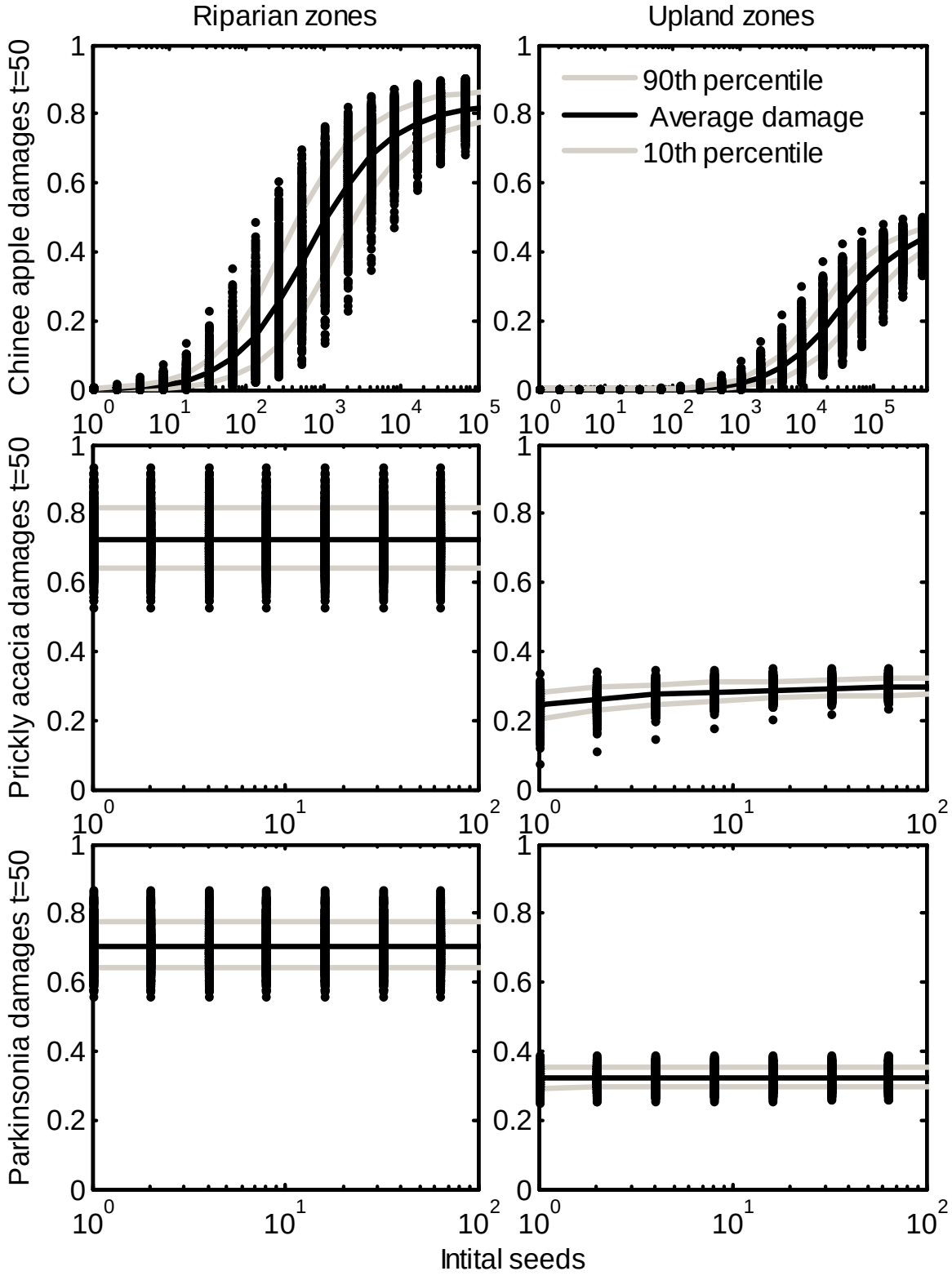


Figure 5-8: Relationship between the number of initial seeds and the damage incurred 50 years later (●) for chinee apple, prickly acacia, and parkinsonia in riparian and upland zones in the Charters Towers district from randomly selected recruitment events. The 10 and 90 percentiles are given as (——) and deterministic recruitment as (——).

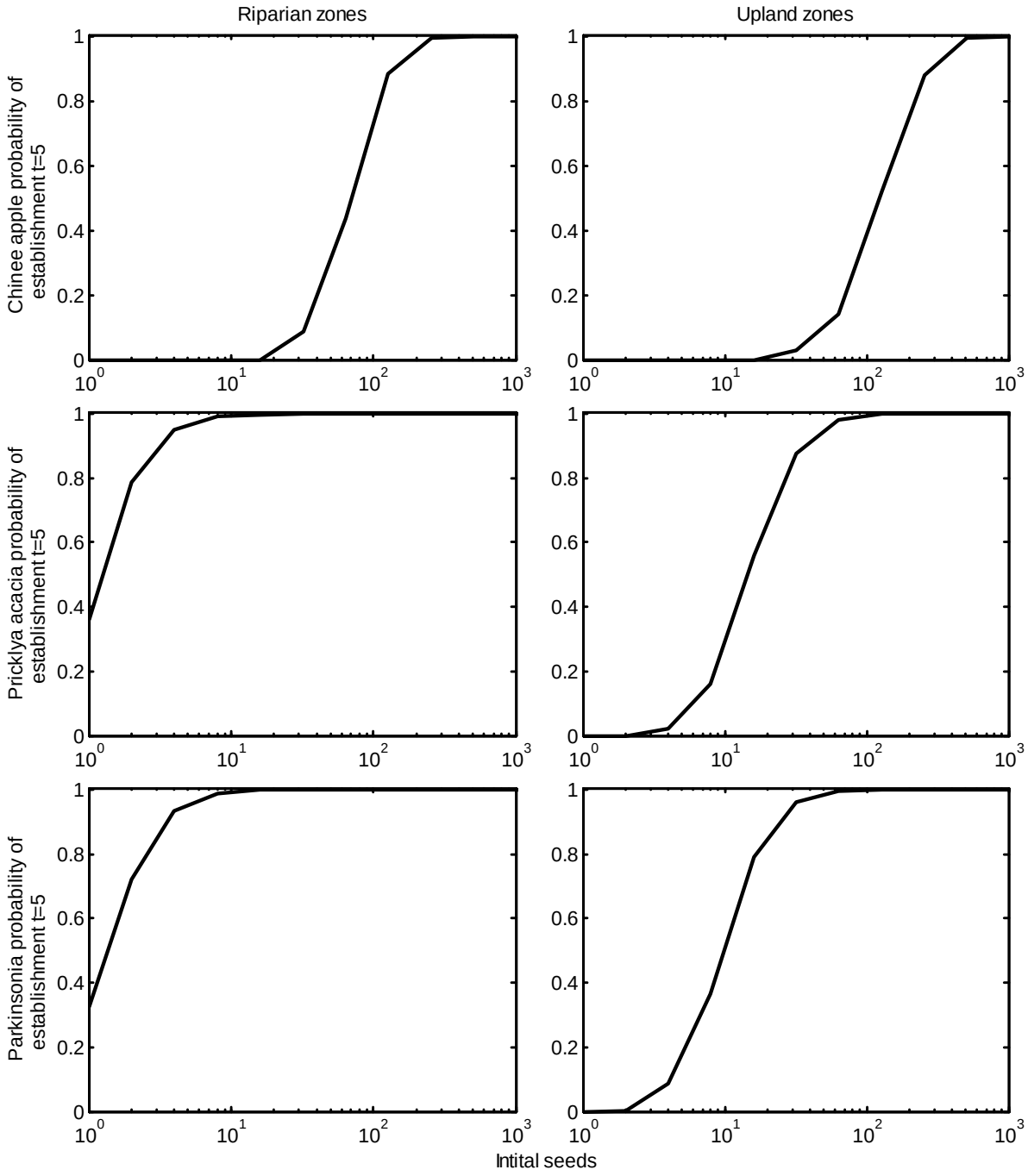


Figure 5-9. Probabilities of weed establishment for given initial quantities of seeds of chinee apple, prickly acacia, and parkinsonia in riparian and upland areas. Data based on the given criteria and randomly selected rainfall data from the Charters Towers district.

5.5 AN EVALUATION OF MODELLED RESULTS

The episodic population model of Watson *et al.* (1997a) showed that background recruitment accounts for 15 to 75 % of recruitment. As with their model, the episodic recruitment model presented here also captures background recruitment in all years (except year 40) (Figure 5-7) and all episodic recruitment magnitudes of $\psi_t \leq 1.0$ can be considered as background recruitment. Although episodic events can result in spectacular population explosions (Kriticos *et al.*, 1999a), they should not be targeted blindly for management purposes at the expense of ignoring continuous or background recruitment processes (Watson *et al.*, 1997a), which also affect grazing production.

Changes in population densities of long-lived woody plants are slow and will not change greatly between years (Radford *et al.*, 2002; Watson *et al.*, 1997a). This is reflected in the simulated results where it takes decades for populations to reach steady-state. Just as it takes years for the pulse of an episodic event to affect pasture production, it will also take many years for the impact of the shock to diminish Figure 5-7. Although the damage to grazing production is relatively constant when the population reaches steady-state, the population structure continues to change. As woody vegetation populations are dependent on episodic recruitment, populations will have fewer juveniles for most years. In the absence of juveniles, there will be less competition for natural resources. Therefore, the steady-state population with episodic population dynamics will have a higher proportion of mature plants than the model with determinist population dynamics. Additionally, there is a feedback loop; with more adults there are fewer opportunities for juveniles to establish themselves.

For the deterministic simulation, the damages from the steady-state population remained constant over time. Consequently, damages were unchanged over time. With the introduction of stochastic processes, however, the steady-state damages changed over time. On average, there were 35 % more years with declining damages than years with increasing damages due to episodic recruitment events. As Higgins *et al.* (2000) highlighted, the rate of change in population density (damage) tends to be greater when populations are increasing than when they are decreasing in density (damages) (Figure 5-7).

Figure 5-7 indicates that, with an initial seed population of 2000 seeds ha^{-1}, a riparian chinee apple infestation requires approximately 75 years for damage to reach steady-state. However, with favourable climatic conditions and associated episodic recruitment events, the same level of damage can occur in less than 50 years (Figure 5-8). These temporal changes to production damage will have management implications. Additionally, by increasing the initial seed population, the impacts of damages will occur sooner. As an example, increasing the initial seed population from 2000 to 80 000 for the same infestation, results in steady-state damage being in year 50 instead of year 75. Therefore, weed management may focus on retarding seed immigration into the system by quarantining cattle between fields (Grice, 1998). Results also showed that longer lived species tend to be more sensitive to the initial quantity of seeds, as they take longer to start producing seeds. During the non-seed producing stage, when an infestation is in the establishment stage, long-lived woody species are more susceptible to unfavourable environmental impacts than at any other time. As longer-lived weeds generally have longer periods of non-seed production, they are more susceptible to unfavourable environmental impacts than their shorter-lived counterparts.

Simulated damages from episodic recruitment occur sooner and are greater for shorter lived woody species than their longer lived counterparts (Figure 5-7). Additionally, short lived woody weeds also have the capacity to recover quickly as they require only a few viable seeds to re-establish themselves (Figure 5-8). However, the elasticity analysis in Chapter 3 (Figure 3-7) shows shorter lived woody weeds are more sensitive to the removal of juvenile plants. This includes non-recruitment due to unfavourable episodic climatic conditions, or fire management which targets seedlings. Therefore, further analysis is required for fire management of weeds with respect to species longevity, effects of episodic recruitment, optimal time of management with respect to episodic recruitment, frequency and intensity of fire, and benefits and cost of fire management.

Impacts from isolated, extreme, climatic events, such as episodic recruitments, have serious implications on plant populations and their structure, as well as impacts on grazing production (Harrington *et al.*, 1984b). However, the sequence and magnitude of these events cannot be predicted.

In fact, even when an episodic recruitment has occurred, it may take some years to recognise it, due to the vastness and heterogeneity of the rangeland system, coupled with the slow growth rates of long-lived woody vegetation (Harrington *et al.*, 1984b). Although this model may help estimate the probability of damage caused by woody weeds in future episodic events, its greatest benefit may come from predicting future damages from past episodic recruitment events and the timing and magnitude of those damages. This is in line with Harrington *et al.* (1984b, p.10) stating: "management in the rangelands depends not so much on predicting what will happen in the future, as adjusting to what has happened in the past."

Both Higgins *et al.* (2000) and Watson *et al.* (1997a) did not account for density-dependence in their models. Rather, they relied on the episodic recruitment of germinants and the gradual declines of populations over time. As Radford *et al.* (2002, p.266) had observed: "Higgins et al. (2000)'s… stochastic conceptual model for tree–grass coexistence was proposed, that suggested there is no optimal density of trees in savanna landscapes." Both models assumed that the frequency and magnitude of episodic recruitment events was constant over time. However, caution is required with this logic when considering compounding recruitment of consecutive years, as has occurred in the early 1970s. Not accounting for density-dependence can result in the population's physical size and requirements exceeding the space and resources available in an area. As with any resource, there are limitations to expansion and exploitation.

In the model presented here, density-dependence has been accounted for with the following outcomes: when population density is low, germinants face less competition for space and resources, therefore weed cohorts gain more from episodic recruitment events than when population densities are high. For any given recruitment magnitude (ψ_t) (assuming there are copious seeds) there will be a greater number of germinants for 'new' or 'recently managed' infestations than for steady-state infestations. Similar to Higgins *et al.* (2000) and Watson *et al.* (1997a), the model still results in episodic recruitments of germinants, followed by gradual decline in the population over time. However, there are upper limits to the space and resources available to populations, as there are for damages on pasture

production. With consecutive years of recruitment, a steady-state population may increase in size and cause more damage. Due to density-dependence, the population will tend back to the normal steady-state level of damages. Figure 5-7 demonstrates this, where damages can be higher than the average (deterministic) steady-state damage and then tends to decrease.

Watson *et al.* (1997a) used long-term recruitment frequencies. Although there was no serial correlation between sequential years, there may be long-term cycles, such as El Niño and La Niña temperature fluctuations, which affect rainfall events. These climatic events may result in long-term pulses of recruitment. For a steady-state population, the impact of a single mass recruitment is diminished within a decade. Therefore, a mass recruitment pulse from any long term trend can be treated as a series of individual episodic recruitment events. Even if there was quick succession of recruitment events over a period of time, their impact would be dampened by the population's density-dependence.

The stochastic driver presented here is simpler than those used by Higgins *et al.* (2000), Watson *et al.* (1997a), Lawes (2004), or Liedloff and Cook (2007), as it does not consider soil moisture, soil types or water stress by particular species. It can, however, determine the timing and magnitude of woody weed damage to pasture production and can be used to explore the impact management options can have on the population structure from an economic perspective. It is also easy to parameterise from survey data, provided a reasonable estimate of average germination can be obtained, overcoming one of the major limitations associated with season-dependent population dynamics models.

5.6 SUMMARY

The recruitment of woody vegetation within rangeland systems is complex, and the process of recruitment is driven by many stochastic elements (Higgins *et al.*, 2000; Kriticos *et al.*, 1999a; Brown & Carter, 1998; Watson *et al.*, 1997a; Carter & Cowan, 1993). Previous studies have attempted to simulate the processes of woody plant recruitment, to explain the dynamics of rangeland systems. These stochastic processes can increase the risk of woody-weed management, as benefits are not guaranteed. However, including stochastic processes into an economic optimisation of woody-weed management should not be taken lightly, as it restricts the economic models that can be used, increases computing time, and adds complexity to the economic solution.

This chapter reviewed the dynamic processes of change within the Australian rangeland systems and concluded that the stochasticity of the wet and dry seasons had the greatest affect on woody seedling recruitment. This was followed by a review of some woody plant recruitment models (Lawes, 2004; Higgins *et al.*, 2000; Watson *et al.*, 1997a). It was determined that seasonal and episodic rainfalls are the dominant drivers of the system. The recruitment model proposed by Lawes (2004) established germination and establishment rules based on daily rainfall and evaporative information. Although the conception of the model is well suited, the model was incompatible with the population model developed in previous chapters, which operated on annual time steps. Based on the research from Watson (1997a), recruitment was divided into background and episodic recruitment. A more elegant model was then developed based on wet and dry season rainfalls. The two models were contrasted based on Charters Towers rainfall and evaporation data from 1885 to 2007 (BoM, 2008). As rainfall events in the region have a skewed distribution, it was possible to simulate episodic recruitment events. The results from the two models were similar.

As the annual rainfall events in Charters Towers are not auto-correlated, it was possible to randomly select annual rainfalls for the stochastic simulator. The model was then used to investigate new and steady state stochastic populations of chinee apple, prickly acacia, and parkinsonia in upland and

riparian zones. Similar to field observations by Radford *et al.* (2002), the populations decreased in more years than they increased. The effects of initial seed densities were also investigated to estimate the probability of the infestation establishing and the damage over a 50 year period. By comparing determinist and stochastic simulation, it does not appear that stochasticity greatly effects woody-weed management decisions.

The aim of this chapter is not to establish a superior woody plant recruitment model to those that have already been proposed by Watson *et al.* (1997a), Liedloff and Cook (2007), Higgins *et al*. (2000), and Lawes (2004). Rather, the objective is to fit an episodic recruitment model to the deterministic density dependent SPM developed in Chapter 3, to test affects of stochasticity on the temporal damage from woody weed infestations. It is currently unknown if stochasticity affects weed-management decisions and if stochasticity should even be included in the optimisation of woody-weed control, although the results presented in Figure 5-7 indicate that the stochastic model has similar levels of weed damage to the determinist model. The answer to either of these questions is still unknown, until the economic model is optimised in Chapter 7, with measures of the variance (risk) of weed management decisions.

6 DEVELOPMENT OF A METHOD FOR OPTIMISING WOODY-WEED MANAGEMENT

This chapter presents the development of a dynamic optimisation model that incorporates stochastic environmental events. The model is further expanded to investigate the effects of biocontrol on management decisions. The parameter values and the simulation results are presented in Chapter 7. To date, ecological population and economic optimisation models have not been combined to establish management policies for woody weeds in the northern Australian rangelands. Historically, many bioeconomic models have sought to combine these modelling disciplines, but have lacked biological realism (as highlighted by Deacon *et al*. (1998)). Some of these biological simplifications include deterministic modelling, concentrating on single species dynamics (Bulte & van Kooten, 1999), along with over-simplified population dynamics. The woody-weed decision model presented here seeks to derive optimal management decisions, whilst retaining ecological processes and economic robustness.

Several economic frameworks have been developed to help determine optimal weed control decisions, each having varying degrees of complexity and success. These frameworks use modelling theories such as: benefit-cost analyses, threshold theory, optimal control (OC), dynamic programming (DP), and stochastic dynamic programming (SDP), listed in increasing levels of complexity (Jones & Cacho, 2000). Economic optimisation models for weed control decisions have tended to be for annual weeds in cropping systems (Jones *et al.*, 2006; Pannell *et al.*, 2004; Jones & Medd, 2000; Pandey & Medd, 1990; Pannell, 1990a; Taylor & Burt, 1984). In contrast to the optimal application rates of control sought in cropping systems, woody-weed management solutions typically seek to determine if management is warranted, when it should be applied, and the method of control to use.

Economic optimisation techniques have not been applied to woody weeds in the Australian rangeland system. A study by Odom *et al.* (2003) derived optimal management polices for another long lived woody-weed, scotch broom (*Cytisus scoparius*, L.) (with a life expectancy of 20–25 years (Hosking *et*

al., 1998)), within Australian National Parks. To derive their solution, the population was divided into the number of seeds and plants. However, it may be prudent to also consider the size of individual plants within the population, as size not only affects the efficacy of different management strategies but also their economic and environmental impacts.

Both OC and DP are reviewed for optimal management strategies for woody weeds within the Australian rangelands. The model developed in this chapter seeks to establish an economic framework for the optimal management of chinee apple (*Ziziphus mauritiana*), prickly acacia (*Acacia nilotica*), and parkinsonia (*Parkinsonia aculeata*). These case studies are used to develop a multidimensional dynamic decision model for the control of woody weeds in the Australian rangelands by using bioeconomic models.

6.1 ECONOMIC MODELS OF WEED MANAGEMENT

Weed control affects current and future levels of infestation (Pandey & Medd, 1991). Static weed management decisions fail to account for the carryover effect of weed management through time (Jones, 2003). When temporal dynamic processes are accounted for, the optimal level of control tends to increase and weed densities decrease. Therefore, control may make economic sense even when the cost of treatment exceeds the benefits in the current period, due to future benefits from current management decisions.

Weeds can, in fact, be viewed as a renewable resource stock that needs to be managed through time, with ongoing costs and benefits (Jones & Medd, 2000). Beef-cattle grazing is the dominant economic use of northern Australian rangelands (Macleod & McIvor, 2004) and the primary economic gain from weed management is the decrease in weed density and the subsequent gain in pasture production. Weed density represents the stock of this resource, although the stock has a negative economic value to graziers. Weed management can be considered an investment in resource extraction, involving implementation costs which results in future benefits. The benefits of a single application of weed

control will diminish over time as the weed infestation tends to grow towards steady-state (assuming non-eradication), being the maximum level of damage to pasture production. Weed management can be considered a periodic reinvestment in extraction of benefits from reduced weed stock levels.

Fishery and forestry literature has shown how OC and DP techniques can be used to determine optimal economic levels of extraction of such renewable resources (Conrad, 1999; Conrad & Clark, 1995; Clark, 1990). Likewise, these optimisation techniques can be used to determine the optimal level of weed control in order to achieve the highest benefits over time (Hastings *et al.*, 2006; Odom *et al.*, 2003; Eiswerth & Johnson, 2002; Higgins *et al.*, 1997; Jones & Medd, 1997; Pandey & Medd, 1991).

For the purpose of this study it has been assumed that: (1) seeds are evenly distributed within a representative hectare, (2) an area does not impede on the productivity of neighbouring areas, i.e. a weed-free hectare will have the same level of productivity regardless of its neighbours' conditions, (3) there are no economies-of-scale for weed management, where the marginal-cost of managing an additional hectare is constant, (4) all prices are assumed to be constant over time, and (5) each hectare can be managed independently of its neighbour. In reality, neighbouring areas will often have similar weed population structures and therefore weed management decisions will be similar for much larger patches than a single hectare.

6.1.1 Weed growth and control model

SPM models developed in Chapter 3 will be used to estimate the effects of control decisions on both current and future profits. Only the basic algebraic model will be represented here; for further model details refer to Chapter 3. The temporal transition of the weed population can be represented as:

$$\mathbf{x}_{t+1} = (\mathbf{H}_t \mathbf{x}_t) - (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_t}) \tag{6.1}$$

where $\mathbf{x}_t$ is a vector representing the weed population in different life-cycle stages; $\mathbf{H}_t$ is a density-dependent SPM that changes over time t (years); u_t is the control decision, represented by an element

from a vector $\mathbf{U}$ of possible management actions; and the mortality rate for each life-cycle stage in response to the control decision is represented by an efficacy vector $\boldsymbol{\varphi}$, from a matrix $\boldsymbol{\Phi}$ representing the efficacies of possible management actions. The values of the elements within $\boldsymbol{\varphi}$ are between zero and one. They represent the mortality rate, above natural mortality rates, due to control for each respective life-cycle stage. Management of the infestation occurs at the end of the annual time step, but prior to the following season (t+1).

Damage to pasture production due to weeds is based on Cousens' (1985a) rectangular hyperbola damage function, as used in Chapter 3, with the addition of the control decision u_t.

$$D_t = \frac{\psi(\mathbf{x}_t - (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_t}))}{1 + \frac{\psi(\mathbf{x}_t - (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_t}))}{\tau}} \tag{6.2}$$

where D_t is the proportion of pasture production lost per hectare at time t due to weeds, τ is the maximum proportion of pasture production loss as weed density approaches infinity, ψ is a damage vector index for the average amount of pasture production lost per weed in each life stage, when weed density is near zero. Damage to pasture production can be reduced by controlling the infestation. The financial return (net benefit) in any time period is:

$$B_t = \pi_{wf}\left(1 - D_t\left\{\mathbf{x}_t, u_t\right\}\right) - C_{u_t} \tag{6.3}$$

where π_{wf} is the weed-free grazing gross margin, and C_t is the cost of control in time t.

6.1.2 Optimal control

There have been numerous studies applying OC strategies for management decisions of invasive species (Hastings *et al.*, 2006; Eiswerth & Johnson, 2002). The solution of the OC model is a level of weed control used in each time period to maximise the present value of investment over the planning horizon T. The objective function is:

$$Max\ NPV = \underset{(u_t)}{} \sum_{t=1}^{T-1} B_t\{\mathbf{x}_t, u_t\}\beta^t + \beta^T S\{\mathbf{x}_T\} \tag{6.4}$$

subject to

$$\mathbf{x}_{t+1} - \mathbf{x}_t = G\{\mathbf{x}_t, u_t\} \tag{6.5}$$

where $B_t\{\mathbf{x}_t, u_t\}$ defines net benefits at time *t* as a function of the weed population (**x**) and the weed control variable (u). $S(\mathbf{x}_T)$ is the terminal value of the weed population at the end (*T*). β is the discount factor $(1+r)^{-1}$ for the given discount rate *r*. The change in the weed population from one time period to the next depends on the state of the infestation in the current time period and the weed control undertaken. This is represented by the equation of motion Eq.6.5. Applying Pontyagrin's Maximum Principle, the first-order condition of control is:

$$\beta_t \frac{\partial B\{\bullet\}}{\partial u_t} + \lambda_{t+1} \frac{\partial G\{\bullet\}}{\partial u_t} = 0 \qquad t = 0,1,...,T-1 \tag{6.6}$$

The first term in this equation is the net discounted marginal benefit of weed control in time *t* and the second term is the marginal change in the weed infestation with respect to a marginal change in weed control. Setting either λ_{t+1} or $\partial G / \partial u_t$ to zero will cause the marginal future benefits of management to be zero. The costate variable λ_{t+1} is akin to a Lagrange multiplier. With some algebraic manipulation $\lambda_{t+1} = \beta_t \frac{\partial B\{\bullet\}}{\partial G\{\bullet\}}$ measures the discounted marginal benefit with respect to a marginal change in weed growth.

6.1.2.1 Assessing optimal control for woody-weed management

One of the key assumptions of OC is that the equation of motion is continuously differentiable with respect to the state (x_t) and control (u_t) variables. Although it is theoretically possible to differentiate this woody-weed growth function with respect to x_t, it is essentially impossible to execute due to the

complexity of the model. For example, Figure 6-1 illustrates how a new woody-weed infestation (α_1) may have a similar level of damage to grazing production, as a recently managed woody-weed infestation (β_1). However, each has a different population structure and will respond differently to the same method of control. Therefore each will have different growth functions, which will require differentiation. Assuming chinee apple has ten life-cycle stages, the model would have ten state variables, which will imply the existence of ten sets of partial derivatives and the corresponding cross-derivatives. Additionally, the rectangular-hyperbola density-dependent growth function and damage function are complex, making the solution very difficult to obtain.

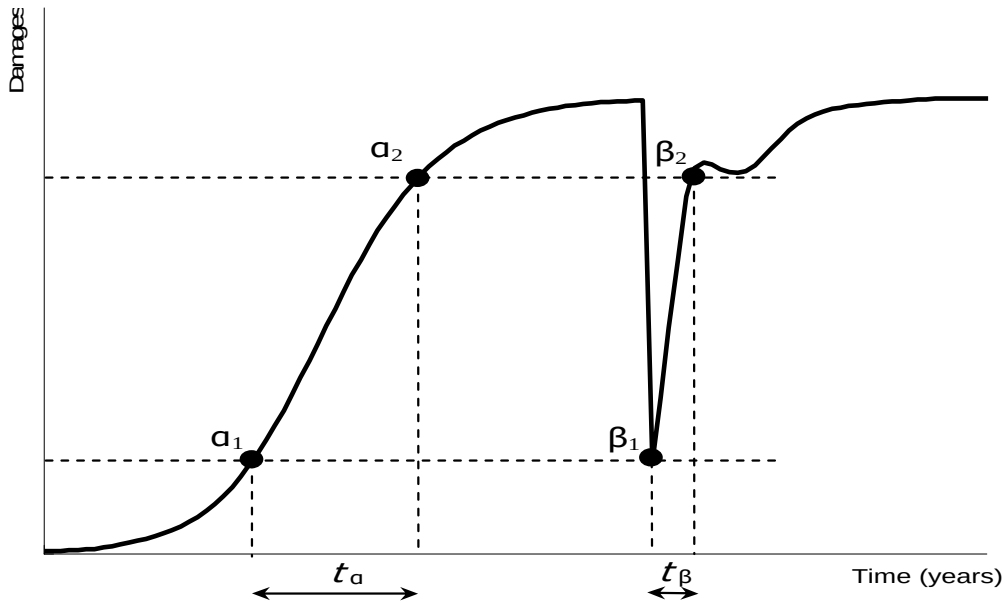


Figure 6-1: An illustration of how an infestation of woody weeds may change over time. Although the initial states α_1 and β_1 have similar damages, their growth rates to α_2 and β_2 are significantly different due to different initial population structures.

Another problem with OC in this instance is the key assumption of the control variable u_t being continuous differentiable. Many woody-weed control practices have dichotomous application rates. For example, burning or mechanical removal of plants do not have degrees of application; they are either applied or not. Therefore, these methods of control are non-continuously differentiable and OC cannot be used in this instance. However, DP can overcome these issues as it is not dependent on differentiability.

6.1.3 Deterministic dynamic programming model

An alternative to OC is DP, which is a computationally effective method for solving maximisation problems and does not rely on the equation of motion being continuously differentiable (Kennedy, 1988). Examples of DP being used to derive optimal weed control strategies include Eiswerth and van Kooten (2002), Jones and Medd (2000), Pandey and Medd (1991), Taylor and Burt (1984) and Fisher and Lee (1981).

Dynamic programming has been used to determine optimal management strategies for a number of different invasive species. Harper *et al.*, (1994) developed a framework for flexible thresholds which accounted for changing economic and production environments and stochastic pest populations. Higgins *et al.*, (1997) built a dynamic ecological-economic model to optimise the services of natural ecosystems with different management regimes. Pandey & Medd (1991) developed a stochastic multi-period decision model for managing wild oats and Jones and Medd (1997) created a population dynamic framework to investigate the integrated weed management of wild oats with a range of control options. Cacho (2006) investigated eradication versus containment decisions for an aquatic biological invader based on a biological spread function while Odom *et al.* (2003) looked at the management of a woody-weed in natural ecosystems with combinations of different control methods and given financial budgets.

The use of DP is of particular significance when control measures are stepwise giving rise to several thresholds (Pannell, 1988a; Auld & Tisdell, 1987). This can occur when applications of control are dichotomous (apply or do not apply), as is the case with managing woody weeds. In essence, DP offers the benefit of threshold theory, i.e., control (u) is used if an infestation is greater than a given level – for a number of different control measures, based on the temporal effects of the population dynamics, and efficacy of control measures. This research will use a similar threshold approaches but will expand on the research by Odom *et al.* (2005; 2003) in determining optimal decision rules based on the state of a woody-weed infestation.

The DP framework groups all the possible population structures into 'states' and divides the planning horizon into periods or 'stages' (Pandey & Medd, 1991). By using backward recursion, the optimal solution for each state, in any given stage, can then be obtained. State variables need to be defined in a way that captures all necessary information that pertains to the current decision problem, in order to achieve a condition of Markovian independence (Nemhauser, 1966). The DP representation from Eq. 6.4 and 6.5 is:

$$V_t\{\mathbf{x}_t\} = \underset{u_t}{Max}\left[B_t\{\mathbf{x}_t, u_t\} + \beta V_{t+1}\{\mathbf{x}_{t+1}\}\right]; \qquad t = T,...,1 \tag{6.7}$$

subject to

$$\mathbf{x}_{t+1} - \mathbf{x}_t = G\{\mathbf{x}_t, u_t\} \tag{6.8}$$

where V_t is the optimal return in the current period. The problem is solved backwards, from $t = T$ to $t = 1$. The recursive equation Eq.6.7 provides the optimal decision policy for any given state. The solution of the model results in an optimal policy $\mathbf{U}^*\{\mathbf{X}\}$.

Using the optimal policy and the equation of motion, optimal trajectories of control measures and weed densities can be derived. The length of the time horizon can be either finite or infinite. One method for solving infinite time horizon problems is to cease the recursion when the temporal change in net present values is less than a predetermined tolerance, $(V_t - V_{t+1}) < Tolerance$ (Kennedy, 1986). An alternative method is to set the timeline well beyond the planning horizon of policy makers, where it is then perceived to be infinite. Under this premise, if the timeline is set at 400 years, this allows for the impact of any management control decision to be fully absorbed by woody-weed populations. However, due to discounting this is essentially the same as an infinite planning horizon. The DP analysis performed in this research was undertaken in MATLAB 7.0® (2007). The algorithm for this process is presented in Figure 6-2:

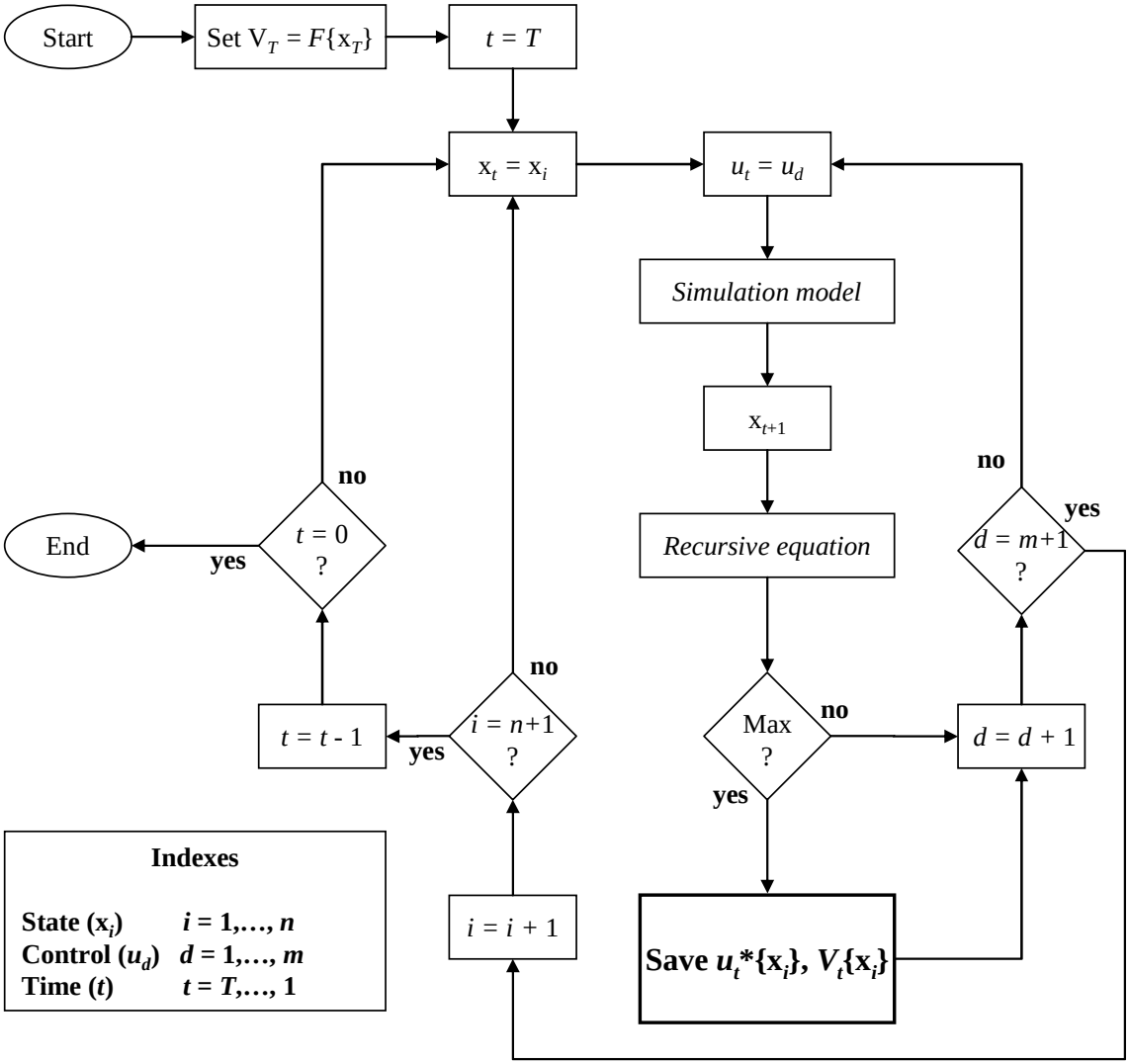


Figure 6-2: Algorithm for dynamic programming (DP)

Steps to solve dynamic programming problem (Kennedy, 1986):

1. Establish a set of fixed states $\mathbf{x} \in \mathbf{X}$ and a fixed control set $u \in \mathbf{U}$, with n and m elements respectively. The time horizon is set at $T = 400$.

2. Set terminal value $V_T = F\{\mathbf{x}_T\}$: this is generally zero for weed management.

3. Solve by backward recursion for all $\mathbf{x}_t \in \mathbf{X}$.

4. Decrease time counter and return to step 3 until $t = 0$ or where convergence is obtained.

5. Obtain the optimal (state-contingent) decision rule set $\mathbf{U}^*\{\mathbf{X}\}$.

6. Use the decision rule to derive the optimal path $\mathbf{x}_t^*$ for the initial state $\mathbf{x}_0$.

6.1.4 Stochastic dynamic programming model

As highlighted by Pannell (1988a), there have been several attempts to include risk and uncertainty when determining optimal thresholds[11] (Jones *et al.*, 2006; Harper *et al.*, 1994; Pandey & Medd, 1991). Deen *et al.* (1993), Pannell (1990b) and Auld and Tisdell (1987) all highlighted that uncertainty in weed management leads to decreased weed control and higher weed density thresholds. Therefore, inclusion of risk into the modelling framework is imperative to ensure weed management decisions are optimal. One of the primary benefits of DP over OC is its ability to capture stochastic processes within its framework.

Risk is introduced into the model as a random variable for stochastic recruitment, as developed in Chapter 5, with its values are selected by Monte-Carlo sampling. Climatic conditions will affect weed-free pasture production and the subsequent profitability of weed management, as will changes in grazing gross margins. It will be assumed that these variables are known at the time of control. Growth and control within this model start with a weed population and followed by an order of events numerically described as:

$$\cdots \underbrace{\text{Stochastic growth}}_{\text{Time interval } t} \rightarrow \text{Control} \rightarrow \underbrace{\text{Stochastic growth}}_{\text{Time interval } t+1} \rightarrow \cdots$$

The SPM model with weed control is expanded to include stochastic weed growth due to climatic conditions. Stochasticity is included in the model as a Hadamard product:

$$\mathbf{x}_{t+1} = (\mathbf{H}_t \mathbf{x}_t) \circ \boldsymbol{\Upsilon}_{t+1} - (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_t}) \tag{6.9}$$

where $\boldsymbol{\Upsilon}_t$ is the stochastic recruitment and establishment of germinants at time t based on climatic conditions, as developed in Chapter 5. $\boldsymbol{\Upsilon}$ is a vector of ones except for the third element, which is the

[11] See **Appendix 3** for a theoretical evaluation of risk on weed management decisions.

stochastic recruitment and establishment of seedlings. As indicated in Eq.6.9, the transition of the population and the stochastic recruitment occur simultaneously.

The decision rule is now based on the current state and the probabilities of going into other states. As with the deterministic DP model, the state of the infestation $(\mathbf{x}_t)$ is known before selecting a control value (u_t), resulting in known current benefit $B\{\mathbf{x}_t, u_t\}$. However, with stochastic influences the future states of the weed population are unknown, $\mathbf{x}_{t+1} = f\{\mathbf{x}_t, u_t, \boldsymbol{\Upsilon}_{t+1}\}$, as are future rewards $R_{t+1} = \pi_{wf}\left(1 - D_{t+1}\{\mathbf{x}_t, u_t, \boldsymbol{\Upsilon}_{t+1}\}\right) - C_{u_t}$. The expectation operator of episodic recruitment $\boldsymbol{\Upsilon}_t$ has known probabilities that are assumed to be independently and identically distributed (*iid*), resulting in a random walk stochastic process. The equation of motion is replaced with a transition probability function $P_{ij}\{u_d\}$ from state *i* to state *j*, given the decision *d* for control u_d. Let the Markovian probability matrix $\mathbf{P}(u_d) \in \Re^n$ denote the state transition probabilities when policy u_d is followed. The recursive equation with stochastic recruitment is:

$$V_t\{\mathbf{x}_t\} = \underset{u_t}{Max}\left[B_t\{\mathbf{x}_t, u_t\} + \beta \sum_{j=1}^{n} \mathbf{P}_{ij}\{u_d\}\ V_{t+1}\{\mathbf{x}_j\} \right]; \quad t = T, ..., 1 \tag{6.10}$$

subject to

$$\mathbf{P}_{ij}(u_d) = pr(\mathbf{x}_{t+1} = \mathbf{x}_j \mid \mathbf{x}_t = \mathbf{x}_i,\ u_t = u_d) \tag{6.11}$$

All other variables are as explained in Eq.6.7. As indicated in the SDP algorithm (Figure 6-3), first the transition probability matrix and the reward matrix for a fixed state set $\mathbf{X}$ and control set $\mathbf{U}$ are defined. Then the optimal control decision $\mathbf{U}^*$ is determined for all possible weed population states in $\mathbf{X}$ using the recursive equation (Figure 6-4).

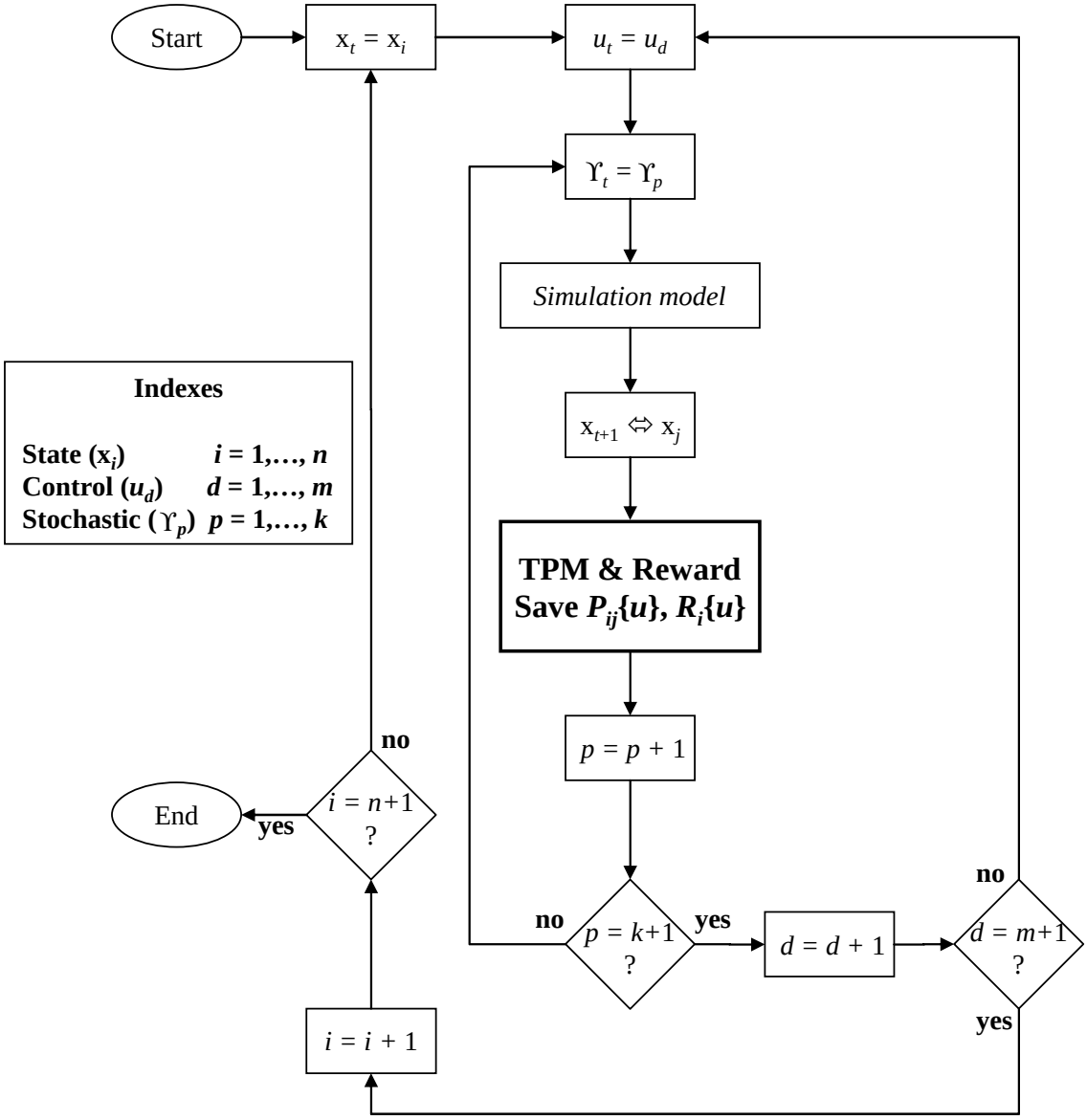


Figure 6-3: Programming algorithm to generate a transition probability matrix (TPM), and reward vector.

To derive the transition probability and reward matrices adapted from Kennedy (Kennedy, 1986):

1. Set dimensions (n, m) and initialise state set (**X**) and control set (**U**).

2. Create an array of transition probability matrices $\mathbf{P}\{n, n, m\}$ and a reward matrix $\mathbf{R}\{n, m\}$ with zero values.

3. Solve simulation model: $\mathbf{x}_j = \mathbf{x}_i + f\{\mathbf{x}_i, u, \boldsymbol{\Upsilon}_p\}$, for $\boldsymbol{\Upsilon} = \boldsymbol{\Upsilon}_1, ..., \boldsymbol{\Upsilon}_k$ where k is the number of Monte Carlo simulations, with each $\boldsymbol{\Upsilon}_i$ drawn randomly from a lognormal distribution.

4. Use results from 3 to estimate $\mathbf{P}_{ij}\{u\}$ given that $\sum_j \mathbf{P}_{ij}\{u\} = 1$.

5. Calculate the reward $\mathbf{R}_i = \pi_{wf}\left(1 - D_i\{\mathbf{x}_i, u_d\}\right) - C_d$.

6. Repeat steps 3-5 for $\mathbf{x}_1, \mathbf{x}_2, ...\mathbf{x}_n$ to fill up the rows of **P** and **R**.

7. Repeat step 6 with $u_d = u_1, \ldots, u_m$ to fill up the 3rd dimension of **P** and the columns of **R**.

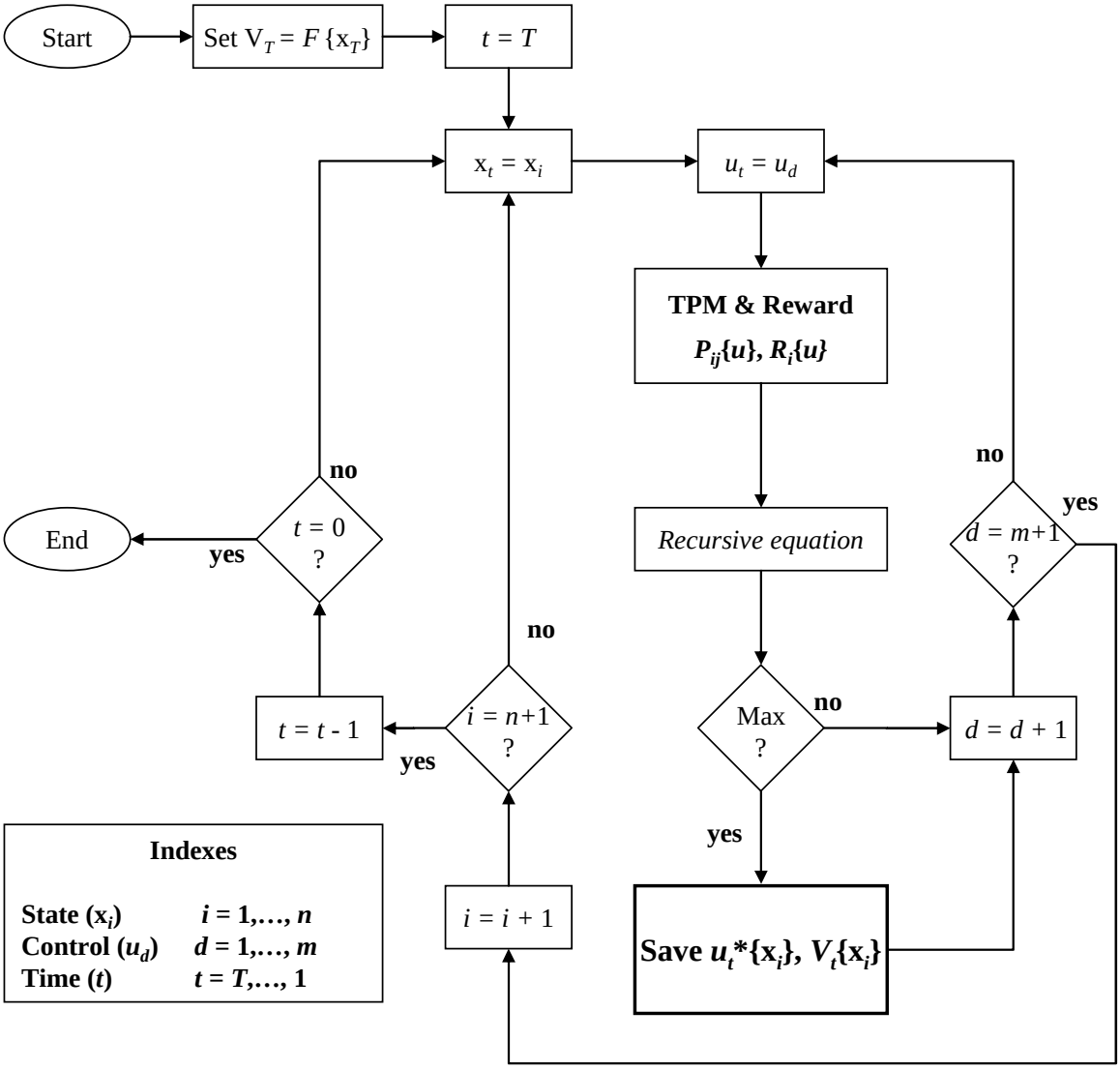


Figure 6-4: Programming algorithm for stochastic dynamic programming (SDP)

Steps to solve the SDP problem:

1. To perform backward recursion set terminal value $V_T = F\{\mathbf{x}_T\}$ and set $t = T$.

2. Apply the recursive equation Eq.6.10 for all $\mathbf{x}_t \in \mathbf{X}$ and $u_t \in \mathbf{U}$, select optimal rule set $\mathbf{U}^*\{\mathbf{X}\}$.

3. Decrease time counter and return to 2 until $t = 0$ or convergence is achieved.

4. Apply the optimal (state-contingent) decision rule $u_t^*\{\mathbf{x}_t\}$ to derive the optimal state transition probability matrix $\mathbf{P}_{ij}^* = \mathbf{P}_{ij}\{u^*\{\mathbf{x}_i\}\}$.

5. Derive the optimal path $\mathbf{x}_t^*$ by performing Monte Carlo simulations for any initial state $\mathbf{x}_0$ and applying the optimal decision rule.

Note that $\mathbf{P}_{ij}$* is the probability of going from state *i* to state *j* in the following period, given the optimal control decision u^* is applied for $\mathbf{x}_i$.

6.1.5 Curse of dimensionality

A major drawback of DP is the 'curse of dimensionality' (Shugart & Smith, 1996). An exponential increase in computation, results from every additional state variable added to the model, resulting in greater demand for computing processing power and memory requirements (Jones & Cacho, 2000). DP models for weed management problems have typically used the number of seeds or total number of individuals in a population as a state variable, with states often ordinally ranked with respect to density. This method is justified for annual weed infestations, as every individual has to go through a seed stage per cycle. Seeds are used as a surrogate indicator for the density of the whole infestation. This assumption cannot be used for long-lived woody weeds. Individual woody weeds can exist in one of many life-cycle stages, and for many annual cycles. Additionally, individuals in different life-cycle stages will have different effects on pasture production. Even the efficacy of different management strategies will depend on the life-cycle stage of the individuals being managed. This means the state of the weed population must be described by the state of its life-cycle stages. This results in a large number of possible combinations of states which are not ordinally ranked. For example, if there are ten life stages and each stage can have ten states, then the population can be in one of 10^{10} states. As DP uses backward recursion for all possible state transitions (Kennedy, 1988), the solution to a deterministic problem with a timeline of 25 years and with four control variables requires 10^{10} x 10^{10}x 4 x 25 = 10^{22} iterations. In reality, land managers will not base their control strategies on such a fine

scale. Management decisions are more likely to be based on the density of seeds, juveniles or adults, as different management strategies typically target these groups of life stages. With three main life-cycle stages, each having ten possible states, the total number of states is reduced to 10^3. Now the numerical problem becomes manageable requiring only 10^8 iterations. This is a decrease of 10^{14} iterations. Or, put another way, if the reduced problem takes four hours to solve, it will take around 45 billion years to solve the unreduced problem. This example was based on a deterministic problem, if stochasticity is included the 'curse of dimensionality', compounded with additional iterations and a very large transition probability matrix.

6.1.5.1 Reduced state stochastic dynamic programming model

Simply reducing the DP and the population models from ten life cycles to three will result in losing vital ecological information, therefore the number of states within the DP model must be reduced in a way that retains the vital population information of the full model. What is needed is a parsimonious probability transition matrix, capturing the probable transition of one state to another. The matrix would be based on the dynamics of the full stochastic population model and the possible control actions, along with a reduced number of state variables for the SDP. This process can be used for both deterministic and stochastic DP models. Let the reduced state variable be denoted by $\mathbf{z}$, a function of $\mathbf{x}$. The recursive equation Eq.6.10 becomes:

$$V_t\{z_t\{\mathrm{x}_t\}\} = \underset{u_t}{Max}\left[B_t\{z_t\{\mathrm{x}_t\}, u_t\} + \beta \sum_{j=1}^{n} p_{ij}\{u_d\}\ V_{t+1}\{z_j\{\mathrm{x}_j\}\} \right]; \qquad t = T,...,1 \tag{6.12}$$

subject to

$$P_{ij}\{u_d\} = pr\left\{\mathrm{z}_{t+1} = \mathrm{z}_j \mid \mathrm{z}_t\{\mathrm{x}_t\} = \mathrm{z}_i\{\mathrm{x}_i\},\ u_t = u_d\right\} \tag{6.13}$$

To derive the state variable $\mathbf{z}$ requires two additional steps. First a '*sample set*' is derived of various possible states over time ($\mathbf{x}_t$) using the SPM model Eq.6.9. To capture the population dynamics of a new infestation, as well as for infestations recovering from different management scenarios, the sample population was started from many different initial populations. The different methods of control were

then applied on the different states for a set time period, or allowed to reach steady-state; whichever occurred first. Once these simulations were run, the population states $\mathbf{x}_t$ were condensed into $\mathbf{z}_t$ states, which only has three life stages: seeds, juveniles, and adults. The first life stage in **z** involves a simple summation of the seedbank and new seeds. However, smaller and larger plants will have different impacts on pasture production. Therefore, a summation of juvenile and adult life stages will not accurately reflect the true impact on pasture production, the efficacy of weed management, or fecundity. The damage function accounts for the damage to pasture production based on the area occupied by the individuals in each life-cycle stage. Using this premise, a juvenile state is based on the total area occupied by all the plants in their life stages. The same process is used for adult plants. This ensures that smaller adults will have less impact on pasture production and produce less seeds than larger adults. In summary, the new state variable z_t captures the total number of seeds (including those by different sized adults), and the total area occupied by juveniles and adults expressed as the aggregate damage by each.

The next step is to map the relationships between the state variables $\mathbf{x}_i$ and $\mathbf{z}_i$, through grouping of $\mathbf{x}_t$ into $\mathbf{z}_t$ values. The values of these variables are created prior to using the SDP model and are stored in vectors **X** and **Z**, which are used as look-up tables whilst solving the problem. The algorithm for this process is presented in Appendix 2: The algorithm for the SDP model.

6.1.6 Introducing biological control

To date, no effective seed-eating biocontrol agent has been found for chinee apple. However, the SDP model can be used to investigate the financial benefits of a hypothetical biological-control agent, prior to identifying a suitable biocontrol agent. This determines if it is even financially viable, by comparing *NPV*s with and without biocontrol. This process could be a first step when considering the use of biocontrol. The model takes into account the potential financial returns for biocontrol, given the current weed-free grazing gross-margins, different levels of biocontrol, efficacies, and implementation costs. The first step is the inclusion of the biocontrol agent into the stochastic population framework. It is

assumed that all weed controls will have a greater impact on the weed than on the biocontrol agent; therefore, the proportion of biocontrol agents to weeds will be the same or higher. That is, biocontrol efficacy will not decrease post control.

The stochastic SPM model Eq.6.9 is expanded to include biocontrol as defined in Chapter 3 (Eq.**3.9**), and is presented as:

$$\mathbf{x}_{t+1} = \left((\mathbf{H}_t^{bio} \mathbf{x}_t) \circ \boldsymbol{\Upsilon}_t \right) - (\mathbf{x}_t \circ \boldsymbol{\varphi}_{u_t,bio}) \tag{6.14}$$

The transition of the population from one period to the next with biocontrol is captured within the transition matrix **H**. The decision and efficacy of biocontrol have been kept separate from other control decisions, as biocontrol is regarded as a once-off (permanent decision) that does not diminish over time (unlike other weed control decisions). To distinguish the optimal weed control decision with biocontrol, '*bio*' has been added to the u_t subscript[12]. Now the biocontrol-SDP model for woody weed decision rules is:

$$V_t\{\mathbf{z}_t\{\mathbf{x}_t\}\} = \underset{u_{t,bio}}{Max}\left[B_t\{\mathbf{z}_t\{\mathbf{x}_t\}, u_{t,bio}\} + \beta \sum_{j=1}^{n} \mathbf{P}_{ij}\left\{u_{d,bio}\right\}\, V_{t+1}\{\mathbf{z}_j\{\mathbf{x}_j\}\} \right]; \qquad t = T,...,1 \tag{6.15}$$

subject to

$$\mathbf{P}_{ij}^{d}\left\{u_{d,bio}\right\} = pr\left\{\mathbf{z}_{t+1} = \mathbf{z}_j \mid \mathbf{z}_t\left\{\mathbf{x}_t\right\} = \mathbf{z}_i\left\{\mathbf{x}_i\right\},\ u_{t,bio} = u_{d,bio}\right\} \tag{6.16}$$

There are now two sets of optimal weed control decision rules $\mathbf{U}^*\{\mathbf{X}\}$ and $\mathrm{U}_{bio}^*\{\mathbf{X}\}$. The former is conditional on weed-free grazing gross-margins, the cost and efficacy of different methods of control, climatic conditions, and the dynamics of the woody weed infestation. The latter is also conditional on the efficacy of biocontrol.

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[12] For more detail into the modelling of biocontrol within the population matrix model please refer back to 3.1.7 Modelling biocontrol of woody weed.

6.2 SUMMARY

An economic SDP framework has been developed to help determine optimal control decisions for woody-weed management in the Australian rangelands. The framework is designed to capture the stochasticity of the system and complex population dynamics, whilst significantly reducing the 'curse of dimensionality' commonly encountered in DP models. This was achieved through the parsimonious grouping of possible populations on the dynamics of the full stochastic population model ($\mathbf{x}_i$) to a reduced number of variable for the same state ($\mathbf{z}_i$), which must weigh up the cost of losing valuable population dynamics information against the curse of dimensionally. Too many $\mathbf{z}_i$ states will result in the curse of dimensionality, whilst too few $\mathbf{z}_i$ groups will result in the simulated population remaining in the same state over time, i.e. the change in the simulated population is insufficient for the population to move into another state within a time period, making management decisions meaningless. With the right parsimony in truncating $\mathbf{x}_i$ into $\mathbf{z}_i$ it is possible to capture the finer interactions between individual weeds and their relevant life-cycle stage, whilst allowing the control decision to be made on a coarser scale.

Weed management threshold decisions are often multidimensional (Pannell, 1990a). The modelling framework developed here not only considers the number of plants in the infestation, but also the population structure, its impact on pasture production, variation in weed-control efficacy, the cost of management, and weed-free grazing gross margins. This framework provides a state-contingent based management tool; i.e., given all the known information, if the weed population state is **x**, then management decisions $u^*\{\mathbf{x}\}$ should be undertaken. This decision does not consider whether management has or will be undertaken in the future. It is in the true sense Markovian (Pandey & Medd, 1991): management decisions are made independently at each stage in time. However, the population state variable portrays the nature of the system (with an implied memory of past events) and transfers all relevant information for the decision problem at that stage. In this way the model maintains a condition of Markovian independence (Nemhauser, 1966).

Modelling and experimental methods are required to help predetermine the benefits of a potential biocontrol agent (van Klinken, 2006; Raghu *et al.*, 2006). In contrast to other weed control decisions, biocontrol is essentially irreversible. Once implemented, the decision for biocontrol is difficult to reverse, whereas the effects of other types of control will diminish over time. To overcome this in the SDP model, biocontrol needs to be treated separately from other control methods. Additionally, the SDP model does not optimise the biocontrol decision. Rather, it optimises the control decisions $\mathbf{U}^*\{\mathbf{X}\}$ given that biocontrol is being used. To determine if using biocontrol is optimal, the model needs to be run with and without biocontrol, and the respective *NPV*s compared. If the *NPV* with biocontrol is higher than without, then the use of biocontrol is optimal, otherwise it is sub-optimal. The application of this modelling framework is undertaken in the following chapter.

7 OPTIMISED INTEGRATED WOODY-WEED MANAGEMENT

This chapter will address five key policy issues facing many Australian rangeland graziers in managing woody-weed infestations. It is currently unknown if controlling woody weed infestations within the Australian rangelands is economically viable and, if it is, the optimal integrated weed management (IWM) policy is unknown. The primary research objectives of this chapter are:

- To determine the threshold to start managing woody-weeds.
- To estimate the optimal strategy for controlling woody weed infestations.
- To analyse how management strategies are affected by changes in:
  - the state of the infestation.
  - the cost of weed management.
  - the efficacy of management (including biocontrol) with weed-free grazing gross margins
- To evaluate the potential benefit of biocontrol with IWM strategies.
- To determine if the risk of weed management due stochastic woody-weed recruitment needs to be considered.

There are four main woody-weed control options (u): (1) no-control[13], (2) burning, (3) chemical treatment, and (4) mechanical removal (blade ploughing). It is unlikely that any single method of weed control is an optimal economic weed management strategy (Monjardino *et al.*, 2005). IWM of woody-weeds often involves different methods of control being applied in different years; different controls are rarely used in the same year. This is partially due to the population dynamics of woody-weed infestations operating on a long time-horizon, resulting in the temporal window of management being

[13] Ignoring an infestation is not the same as no-control. The term ignoring is reserved for when an infestation does not receive any type of weed control over time, whereas no-control is the action taken in years between other types of controls.

longer than that for short-lived plants. This is illustrated by the fact it takes around five years for a chinee apple seedling to reach maturity and produce seeds, whereas short-lived annual weeds can geminate and set seeds within months, resulting in shorter temporal management windows. This means that shorter lived weeds require more intensive and frequent controls.

There are many factors that influence an IWM strategy and subsequently economic benefits of weed control (Vitelli & Pitt, 2006). Some of these factors include:

1. the available methods of weed control
2. efficacy of weed control methods
3. cost of weed control methods
4. weed-free grazing gross-margins
5. weed impact on grazing production
6. environmental characteristics
7. discount rate (time value of money)
8. population structure and density
9. plant demographics, including:
    a. size of plants
    b. potential lifespan
    c. natural mortality rates,
    d. reproductive age (size)
    e. size of the seedbank
    f. germination rates
    g. fecundity rates
    h. seed longevity

All of these elements are captured within the stochastic dynamic programming (SDP) framework developed in Chapter 6; used here to derive the optimal economic IWM policy (**U***).

Biological control (biocontrol) is another potential method of weed control, although none of the three case study species have been successfully controlled by biological agents (van Klinken, 2006; Grice, 2002; Radford *et al.*, 2001a). Biocontrol (such as seed eaters) can be used in isolation or in conjunction with other treatments (van Klinken, 2006). Unlike other methods of weed control, the biological decision should be considered irreversible once implemented, as it is difficult to reverse or remove. Therefore, to investigate the potential of biocontrol, the SDP model is run with and without a theoretical biocontrol agent. If the net present value (*NPV*) increases with biocontrol, then it should be part of the optimal package. Otherwise, biocontrol is a sub-optimal option.

As described in Chapter 6, the optimal control decision of the SDP model is contingent on the state of the infestation, which is described in terms of the woody-weed population structure and size (and the resulting damage to pasture production). This chapter brings together the analytical tools developed in previous chapters: the density-dependent SPM in Chapter 3, the stochastic recruitment model in Chapter 5, and the stochastic dynamic programming (SDP) framework developed in Chapter 6. Results presented in this chapter are based on the management of chinee apple infestations in an upland zone[14]. Due to the curse-of-dimensionality with SDP models (Bellman, 1957), the optimal weed control decision is only derived for single-species infestations. The SDP technique was implemented in MATLAB (2007).

[14] The other case study weeds (in both zones) are not presented in this chapter, as this additional information will be cumbersome to present, due to the complexity of the model and amount of information that will be provided on each species in each zone. The author intends to publish this information in journal articles at a later date.

7.1 CALIBRATING THE SDP

The cost of controlling an infestation ($\mathbf{C}_u$) is dependent on its density. The denser a weed infestation, the more effort and materials are required to manage it. However, there are also fixed weed management costs that are independent of weed density, such as transporting equipment and labour to the site. The fixed and variable costs of four different chinee apple control options are given in Table 7-1 (Zull and Marshall, *unpubl. data* 2008). C_u for a given state is calculated based on weed density as $\mathbf{C}_u = \mathbf{C}_{u,\text{fixed}} + \mathbf{C}_{u,\text{variable}} * \mathbf{x}_t$.

The efficacy of the four different control options is provided in

Table 7-2. This represents the proportion of the individuals in each life-cycle stage which are removed with the different control methods. These efficacies include the possibility of missing some of the smaller plants that may be hidden in grass or other vegetation and are in addition to natural or climatic mortalities (Zull and Marshall, *unpubl. data* 2008). It is assumed that the application rates for these control measures are fixed, as it is not possible to partially apply these types of controls. Additionally it is assumed that weed control costs and technology remain constant over time. The discount rate for the time value of money is also constant at five per cent.

The state of an infestation is based on the quantity and size of juvenile and adult plants, along with the total number of seeds ha^{-1} (new-seeds + seedbank). Any chinee apple infestation was assumed to be in one of 5280 possible population states (**X**), derived from 11 seed, 24 juvenile, and 20 adult sub-states, (11x24x20=5280). The weed damage to pasture production from each state is derived from the damage of juveniles $D\{J\}$ and adult $D\{A\}$ weeds. Deciding on the number of states for the decision model required weighing up the cost of an additional state (curse-of-dimensionality) against the loss of information caused by coarse states.

State variable values were derived by running multiple population simulations and truncating the population structures as outlined in Chapter 6. It was found that the model was sensitive to the

Table 7-1: Weed control options (u) and costs \$ ha^{-1} (C_u) (Zull and Marshall, *unpubl*. data 2008).

	Control method (u)	Fixed cost[15] \$ per hectare	Variable cost[16] \$ per 1 % of weed density	Max total cost \$ per hectare (steady-state)
u_1	No-control	0	\$0	\$0
u_2	Burning	18	\$0	\$18
u_3	Chemical (poisoning)	15	\$9.85	\$1000
u_4	Mechanical (blade ploughing)	200	\$2	\$400

Table 7-2: The efficacies of different chinee apple management options in upland zones (Zull and Marshall, *unpubl.* data 2008).

		Control method and efficacy			
		No-control (u_1)	Burning (u_2)	Chemical (u_3)	Mechanical (u_4)
Life-cycle stage	New seeds (*NS*)	0	0.70	0	0
	Seedbank (*SB*)	0	0.70	0	0
	Seedlings (J_1)	0	0.48	0.68	0.54
	Small juveniles (J_2)	0	0.29	0.80	0.57
	Medium juveniles (J_3)	0	0.11	0.90	0.70
	Large juveniles (J_4)	0	0.05	0.92	0.75
	Small adults (A_1)	0	0.01	0.89	0.83
	Medium adults (A_2)	0	0.01	0.85	0.87
	Large adults (A_3)	0	0.01	0.83	0.89
	Largest adults (A_4)	0	0.01	0.82	0.90

sampling of $\mathbf{x}_t$ in determining the truncated states $\mathbf{z}_t$. To achieve the best results, the model was started with 2000 seeds and allowed to reach steady-state. Then each type of control (u) was applied to each of the sample years and the model was run for 100 time periods (years) or until the damage had reached steady-state without any further controls, whichever occurred first. The net benefit of control is defined as the difference between the *NPV* of the optimal decision rule **U*** and ignoring the infestation.

[15] Fixed control costs are the amount paid for control that is independent of the infestation density. This is represented as the minimum cost involved in controlling each hectare. For example, if it costs \$3500 to get to an infestation, set up equipment and search it for weeds, and there are 100 hectares to control, then the minimum (fixed) cost is \$35 ha^{-1}.

[16] Variable costs include additional labour, equipment, and/or chemicals used to control denser infestations. Variable values given in **Table 7-1** are based on a portion of the weed density ($\mathbf{x}_t$), where full weed density ($\mathbf{x}_t$) = 1.0.

7.2 RESULTS

7.2.1 Applying control rules U*

Based on Markov chain process principles, this research problem is autonomous, where the control decision is only dependent on the current state and not contingent on previous decisions. The optimal decision rule $\mathbf{U}^*\{\mathbf{X}\}$ is a 'package' of control decisions used to manage any infestation, based on its current state (Odom *et al.*, 2003). The control decision *u** is based on the population state ($\mathbf{x}_t$) which in turn is a function of number and size of juvenile (*J*) and adult (*A*) plants, as well as the number of seeds (*S*), therefore $u_t * \{\mathrm{x}_t\{S_t, J_t, A_t\}\}$.

Preliminary analysis indicated that the control decision appears to be independent of the quantity of seeds present; therefore, a control decision table can be constructed based on the level of damage from juvenile and adult plants (Figure 7-1). Note that the maximum damage to pasture production from juvenile plants (0.032) is far less than that from adults (0.41). Figure 7-1 illustrates the thresholds between treatment types, based on the state of the infestation. An example of applying control decision **U*** against the ecological stochastic population model (Eq. **3.8**) is given in Figure 7-2. The initial infestation represents a steady-state chinee apple population in an upland zone over a 200-year time period. This provides an insight into the temporal weed damage, management decisions, and financial benefits. The simulation is based on π_{wf} = \$80. Climatic conditions were randomly selected using a Monte Carlo selection process (based on Charters Towers historical rainfall), as defined in Chapter 5. The model indicates that controlling chinee apple can increase profit margins, when π_{wf} = \$80 (Figure 7-2(c)). Controlling chinee apple has a long payback period as **U*** required 29 years to breakeven[17] for the given weed costs and grazing returns (Figure 7-2(c)). The solid line in Figure 7-2(a) represents how

[17] The breakeven point is when the *NPV* from **U*** is equal to the *NPV* of ignoring the infestation.

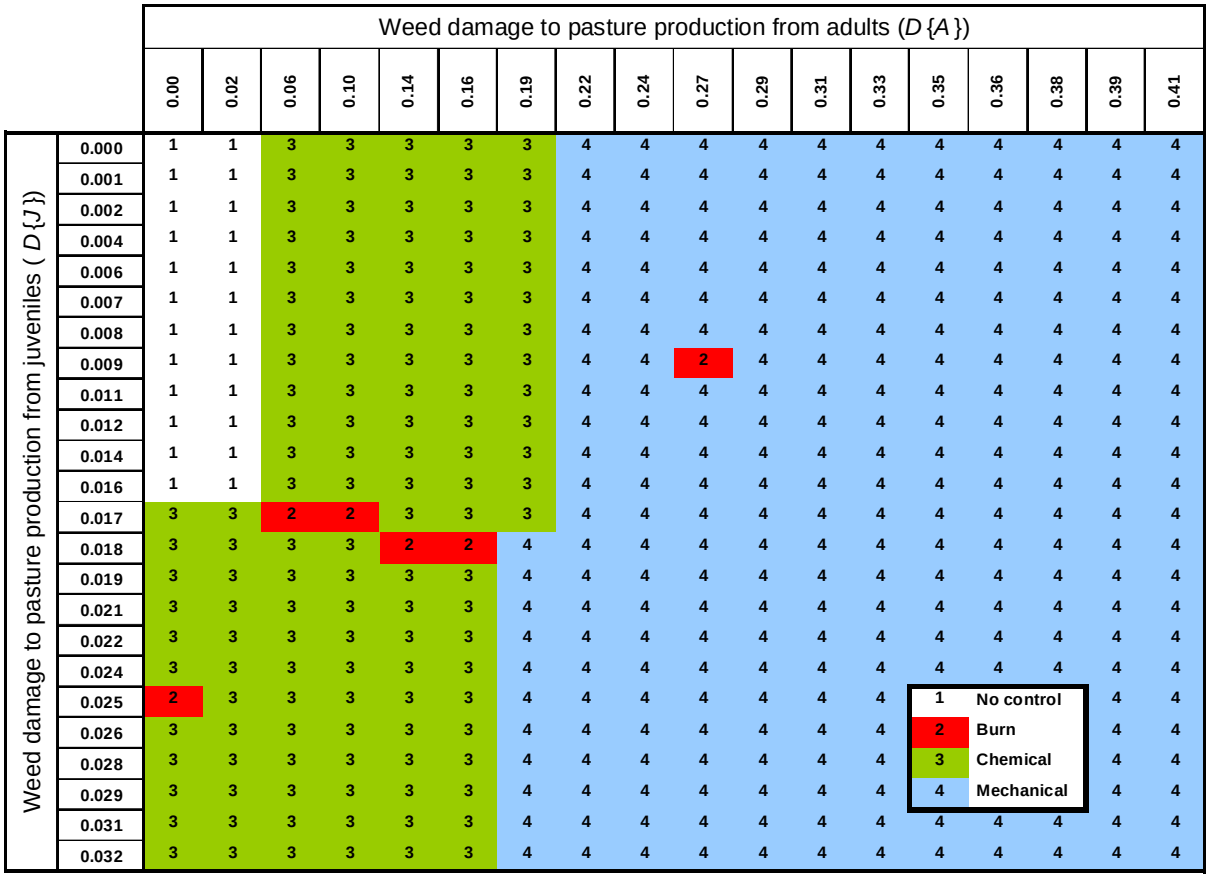


Figure 7-1: Optimal weed control decisions *u with respect to changes in damages to production from juvenile ($D\{J\}$) and adult ($D\{A\}$) plants, seed state set at 190,000 ha^{-1}.**

the damage to pasture production is affected by climatic condition, when **U*** is not applied. Damage to pasture production decreases from about 0.45 down to around 0.05 with **U*** (Figure 7-2(a)). All four weed controls were used at some point to manage the infestation (Figure 7-2(b)). After the initial intensive control phase in the range t<25, chemical control was used periodically (every ~20 years), but the frequency of application was climate dependent. Beyond the 100th year, the *NPV*s remain constant, because future values beyond that time are practically nil due to discounting. Every run of the simulation will result in different temporal weed damage, management decisions, and financial benefits, as the simulation is based on randomly selected climatic conditions. Nonetheless, this single simulation (Figure 7-2) illustrates how the decision rule **U*** can be applied over time after observing the initial state for every time period.

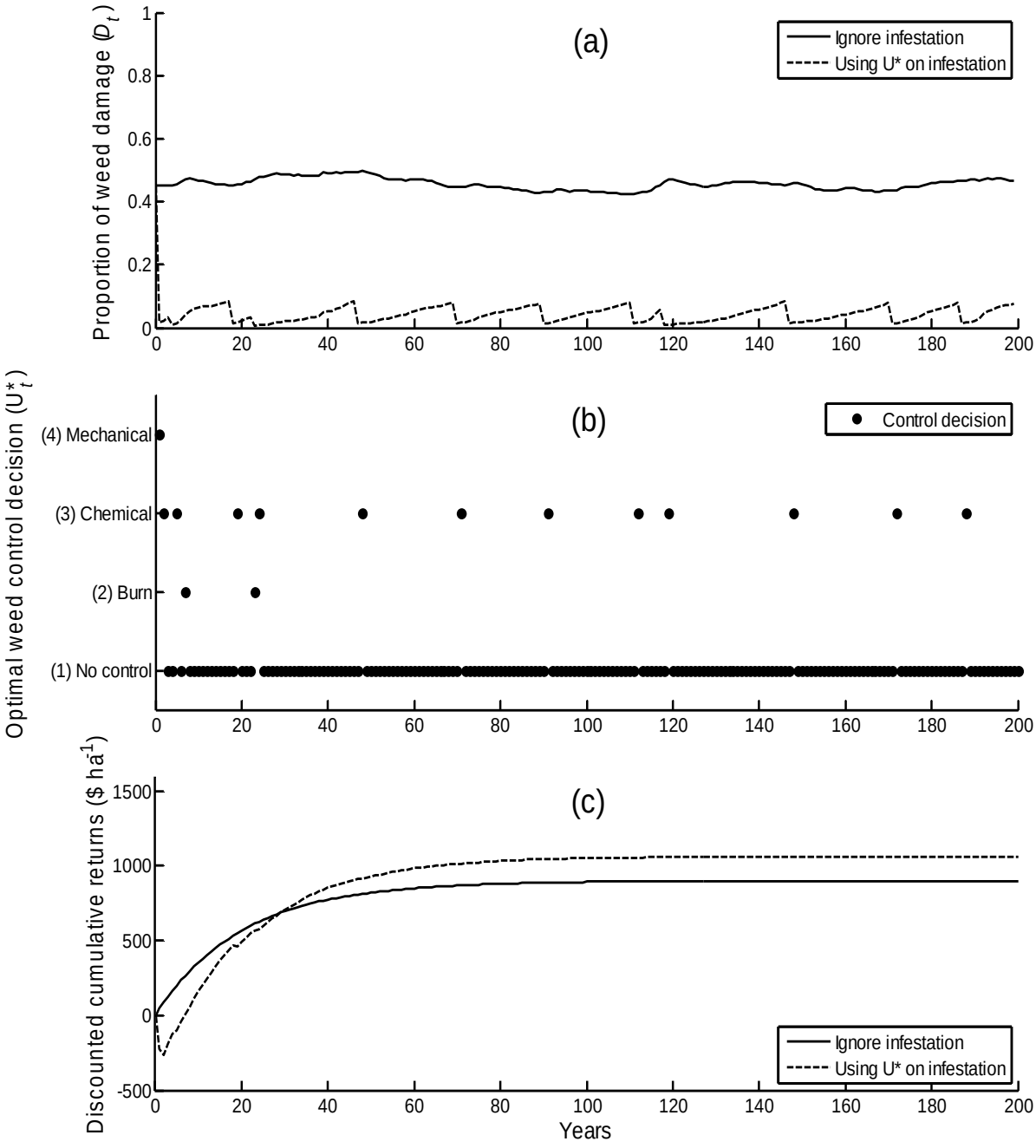


Figure 7-2: A 200-year stochastic simulation of chinee apple in upland zones using optimal control decision U* as compared to ignoring the infestation. Population started from steady-state population, and climatic conditions were randomly selected. Simulation is based on normal control costs (C_u) and weed free grazing gross margins π_{wf} = $80, (a) damage over time from both using U* and ignoring the infestation, (b) the types of control implemented over time under U*, and (c) discounted cumulative profits from controlled and ignored infestations.

7.2.2 Probable benefits of the optimisation rule U*

Future climatic events and their impacts are unknown. Therefore current decisions must be based on the probability of future benefits and costs. To investigate the expected benefits of using the optimal weed control decisions **U*** was applied to a steady-state (upland) chinee apple population. The results are based on a 100–year time period using 4000 iterations. The cumulative distribution of the net benefits of **U*** are given in Figure 7-3. The expected (median) net benefit of **U*** is ~\$165 ha^{-1}, with the *NPV*s ranging between \$–125 and \$208 ha^{-1}. The 10th and 90th percentiles of **U*** are between \$140 and \$179 ha^{-1}, with **U*** resulting in a positive net benefit 94 % of the time.

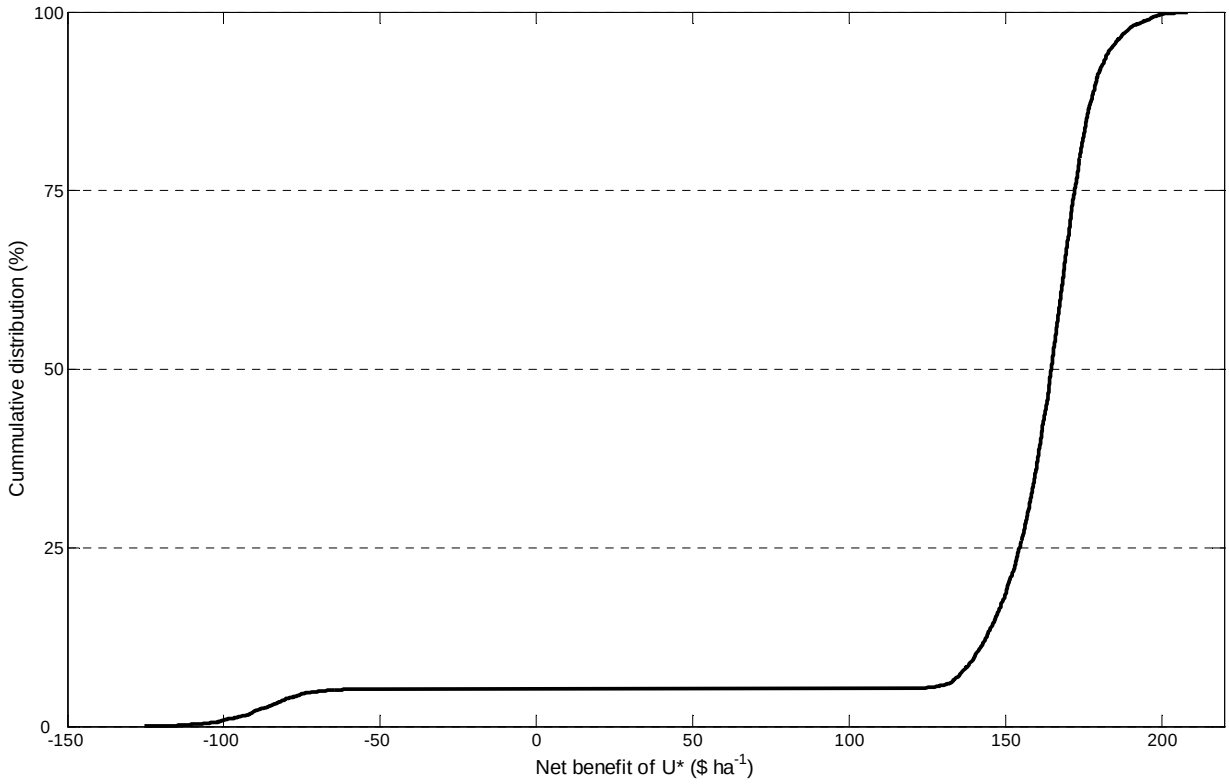


Figure 7-3: Cumulative distribution function of net benefits of U*** for controlling a chinee apple infestation in upland zones, based on normal C_u and π_{wf} = \$80.**

7.2.3 Effects of grazing gross margins

In the previous illustrations (Figure 7-2), the weed-free grazing gross margin (π_{wf}) was fixed at \$80ha^{-1}; now the expected levels of damage and *NPV*s are investigated with respect to changes in π_{wf}. The optimal decision rule **U*** is obtained from the reduced population states (Z) and was applied to the full stochastic population model states ($\mathbf{x}_t$) for a 100–year time period and for 1000 iterations, starting with an upland steady-state chinee apple infestation. The following analysis (Figure 7-4) is based on the full 100,000 population samples for each π_{wf}. Figure 7-4 (a) shows the expected optimal level of annual weed damage for given levels of π_{wf} and normal control costs ($\mathbf{C}_u$). The variance in damage is due to the stochasticity of climatic conditions on weed populations. When π_{wf} = \$80, the annual proportion of damage to pasture production will generally be between 0.02 and 0.08, with a mean of 0.05. This is similar to the levels of damage presented in Figure 7-2(a), with similar upper and lower bounds of damage over time. When π_{wf} < \$34 the optimal decision is to ignore the infestation and damages to pasture production are similar to those shown in Figure 7-2 (a) for the ignored infestation (solid line). The optimal level of damage drops when π_{wf} > \$34, (when weed control starts being applied) and then levels out. The variance of damage initially increases with control and then decreases as π_{wf} increases; this is due to the increased intensity of weed control. Figure 7-4(b) shows the expected optimal annual level of damage from juvenile and adult plants as well as seed-numbers. The level of damage from juveniles is insignificant relative to that of adults, and is the least changed of all three states. Adults represent the majority of the damage to pasture production, and there is a clear relationship between adult density and seed numbers. Figure 7-4 (c) presents the expected (median) total *NPV* using the optimal decision rule **U***, compared with the *NPV*s of ignoring the infestation. Figure 7-4 (d) presents the expected benefit from **U*** calculated as the difference in *NPV* between **U*** and ignoring the infestation. According to these results, control should be applied when π_{wf} > \$65, where the net benefits from **U*** became positive (Figure 7-4 (c)). Control decision **U*** for

$\$34 < \pi_{wf} < \$65$ presents an interesting case, as **U*** results in a net loss (Figure 7-4 (d)) relative to ignoring the infestation. Even though there is a net loss in this interval (Figure 7-4 (d)), the expected total *NPV* from grazing with weed control is still positive (Figure 7-4 (c)). Note, that farmers still generate profits from grazing and weed control, though it is less than from grazing and ignoring the infestation. These two thresholds (π_{wf} = \$34 and π_{wf} = \$65) also result in steep changes in optimal weed damage to pasture production and weed population density (Figure 7-4 (a & b)).

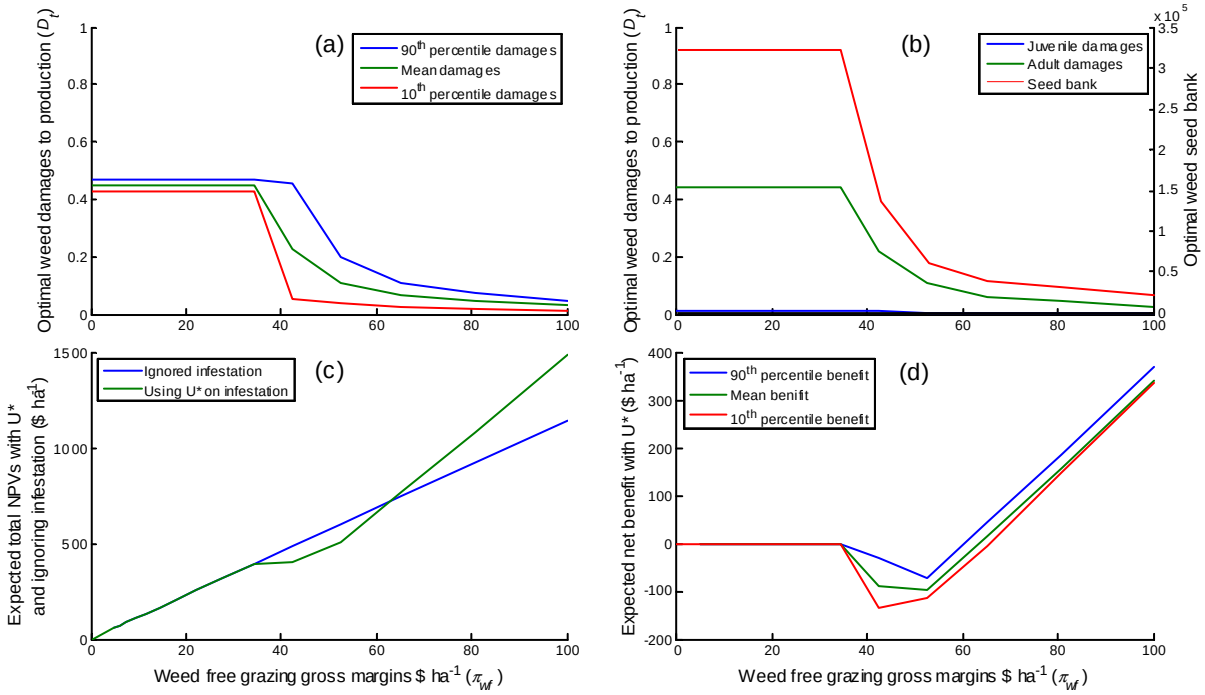


Figure 7-4: Expected damages, *NPV*s, and net benefits of U* using normal control costs (C_u), with respect to changes in weed free grazing gross margins π_{wf}. (a) Range of optimal weed damage from U*, (b) optimal weed damage from juvenile and adult plants, and seed quantities, (c) comparing expected *NPV*s from U* and ignoring the infestation, and (d) range of net benefit of U*.

7.2.4 Effects of weed-management costs

Similar to the analysis of weed-free grazing gross margins (π_{wf}), changes in control costs ($\mathbf{C}_u$) will also affect the level of weed control, levels of weed damage, and benefits from **U***. For each level of weed control costs ($\mathbf{C}_u$) a sample population of 100,000 observations was used. The results in Figure

7-5 have a similar format to those in Figure 7-4, but here reductions in weed management costs ($\mathbf{C}_u$) are represented on the horizontal axis, whilst weed free grazing gross margins are held constant at π_{wf} = \$50. A decrease in $\mathbf{C}_u$ by 20 % represents a 20 % reduction in all weed management costs for all types of control. Figure 7-5 indicates that at the base cost (reduction = 0), **U*** results in a net loss (~ -\$70 ha^{-1}). Figure 7-5 (c & d) shows that $\mathbf{C}_u$ must be decreased by 21 % before there is a net benefit from **U***. When the reduction in $\mathbf{C}_u$ is greater than 45 % the optimal weed damage remains constant (Figure 7-5 (a & b)). That is, the reduction in weed costs does not increase control, but increases the net profitability of **U*** (Figure 7-5 (c)). Note that, the expected *NPV* from ignoring the infestation remains constant, as it is not affected by changes in $\mathbf{C}_u$. When reductions in $\mathbf{C}_u$ are between 0 and 21 %, the expected net loss from **U*** initially increase and then start to decrease (Figure 7-5 (c)). Control decisions for $\mathbf{C}_u$ <21 % will also require further investigation, as **U*** results in net losses. Again, even when **U*** results in the greatest loss (reduction in $\mathbf{C}_u$ of five per cent), total *NPV* of **U*** is still positive (Figure 7-5(d)). The reason for these net losses is explained in the next section of this chapter.

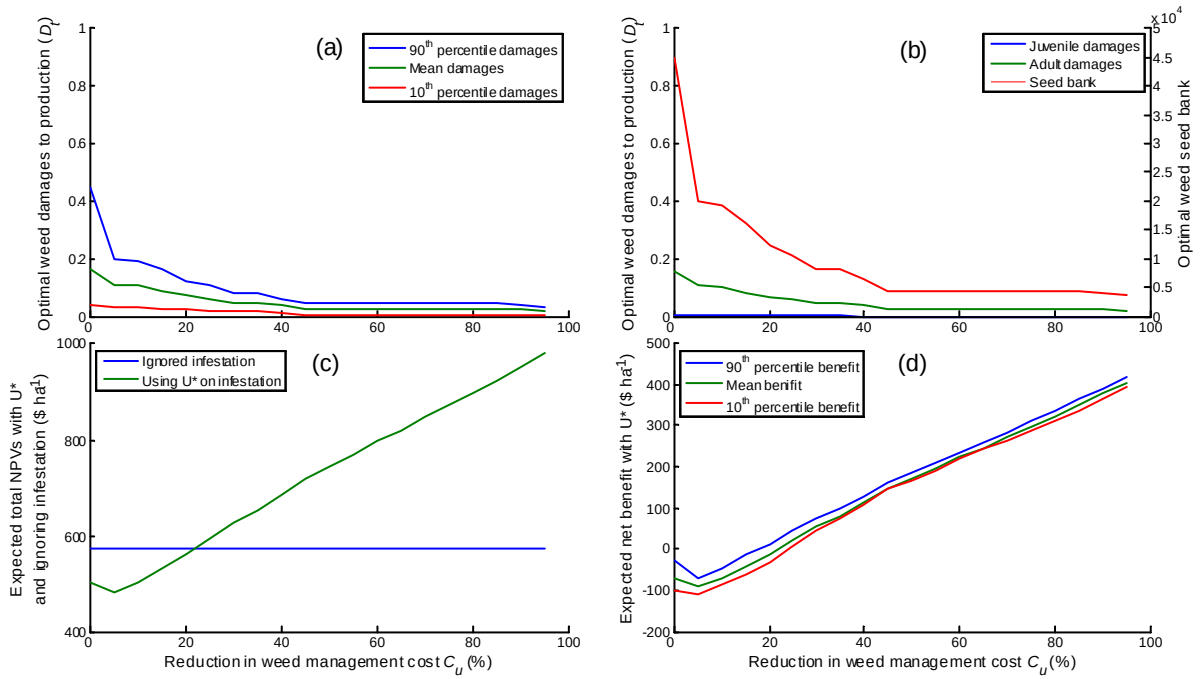


Figure 7-5: Expected damages, *NPV*s, and net benefit of U* for weed free grazing gross margins of π_{wf} = \$50, with respect to reductions in weed control costs ($\mathbf{C}_u$). (a) optimal weed damage with U*, (b) optimal weed damage from juveniles and adults, and seed quantity, (c) comparing expected *NPV* from U* with the *NPV* of ignoring infestation, and (d) net benefits of U*.

7.2.5 Changes in weed management costs and grazing gross margins

To understand the effects of changes in both weed control costs ($\mathbf{C}_u$) and weed-free grazing gross margins (π_{wf}) on the optimal weed control decision $\mathbf{U}^*$, both parameters were changed simultaneously and the SDP was solved. $\mathbf{U}^*$ was applied against the full stochastic population model for 400 iterations, for each combination of $\mathbf{C}_u$ and π_{wf} to derive the optimal level of weed damage and assorted variables. In this case, expected pasture production is reported instead of expected damage (Figure 7-6)[18] for ease of presentation. The lower darker area of the graph, where the pasture production is at ~65 % (z-axis) is where the infestation is ignored.

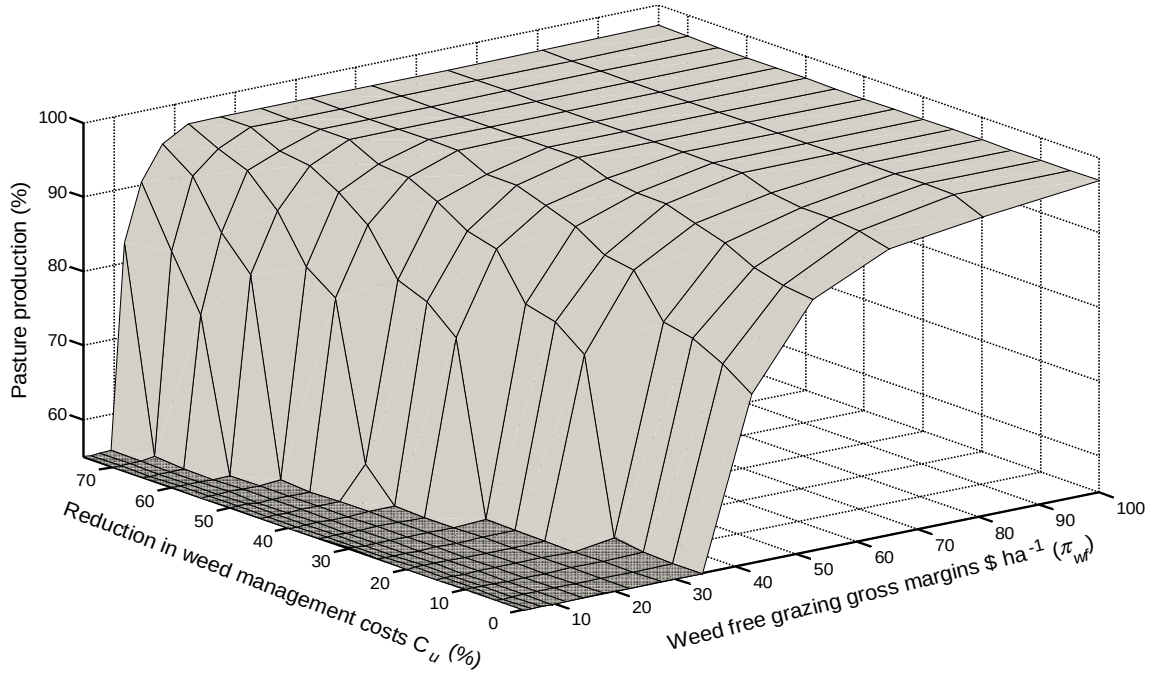


Figure 7-6: Expected pasture production from using $\mathbf{U}^*$ with respect to changing weed control costs (C_u) and weed free grazing gross margins (π_{wf}). The maximum damage to pasture production (45 %) is from combinations of high C_u and low π_{wf}, resulting in the infestation being ignored. Eradication is not sought even for combinations of low C_u and high π_{wf}, with the optimal level of weed damage to pasture production of around five per cent.

[18] The exact relationship is: expected pasture production (%) = 1 – expected damages (%)

Similar to previous analyses, there are two clear thresholds in the application of **U***. Below the lower management threshold, the combinations of $\mathbf{C}_u$ and π_{wf} result in ignoring the infestation as the optimal decision. At this point, the expected level of pasture production is reduced by about 45 %. It is never optimal to attempt eradication, as the expected pasture production is at around 97 %, at high values of cost reduction and weed free gross margins. The net benefit of **U*** is again calculated by comparing the expected *NPV* from **U*** with that of ignoring the infestation Figure 7-7). As before, the net benefits of **U*** are negative over some intervals of $\mathbf{C}_u$ and π_{wf} (Figure 7-8).

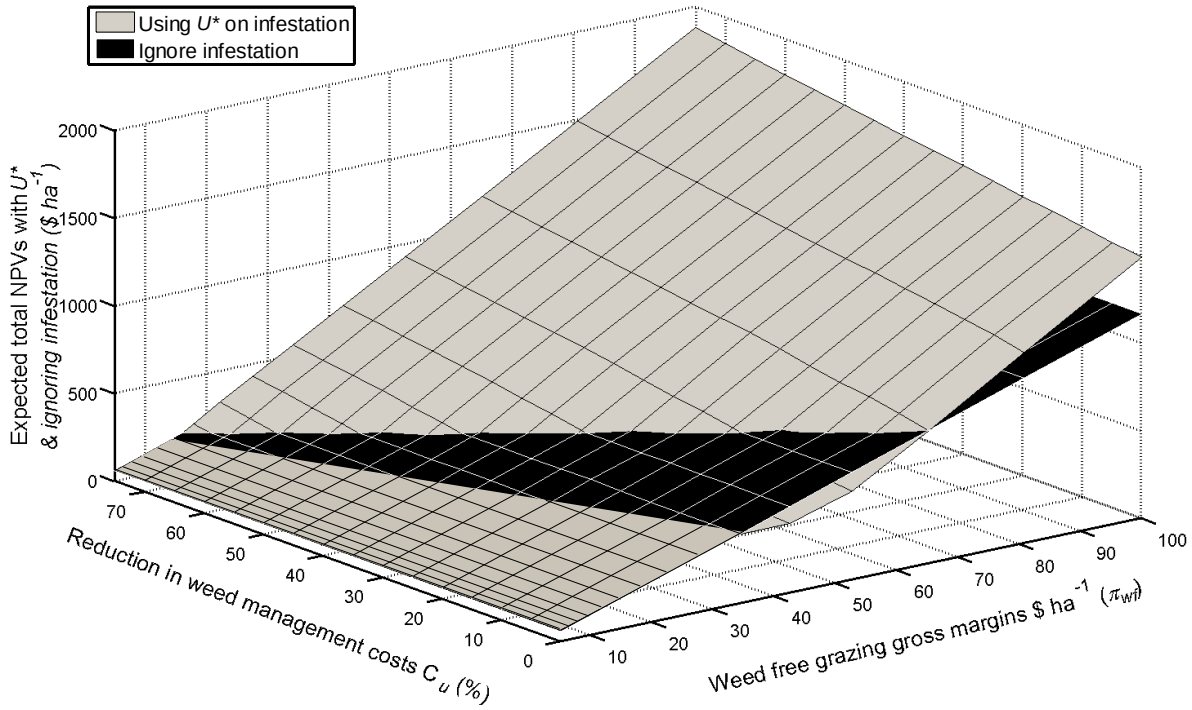


Figure 7-7: Expected total *NPV* from grazing with U*** and ignoring the invasion, with respect to changing** C_u **and** π_{wf} **. The combinations of higher Cu and lower** π_{wf} **result in the same expected total *NPV* using** U*** or ignoring the infestation.**

Optimal economic decisions can have expected net losses. Such cases include mitigating the risk of an adverse condition occurring (Auld *et al.*, 1987). As with any insurance decision it is expected, that insurance will not be needed in any given year, therefore the expected return from purchasing insurance is negative. However, if an adverse event occurred and insurance was required, the insured party would be in a better position than if they were uninsured. Therefore the expected loss from

insurance is warranted. The merit of risk-mitigation of **U*** can be interpreted in a similar context, and is investigated.

Weed-free grazing gross margins are positive (π_{wf} >0), and the damage from woody weeds is never greater than one ($0 \le \psi \le 1$), therefore ignored infestations will always have a total $NPV \ge 0$. Although stochastic climatic conditions can affect woody-weed population dynamics, weeds can never occupy more than 100 % of an area. Therefore, the net losses associated with **U*** are not due to risk-mitigation and the expected benefits of **U*** should have the same or a higher *NPV* than when ignoring the infestation. It can therefore be assumed that the SDP is over-optimistic over a small range of states.

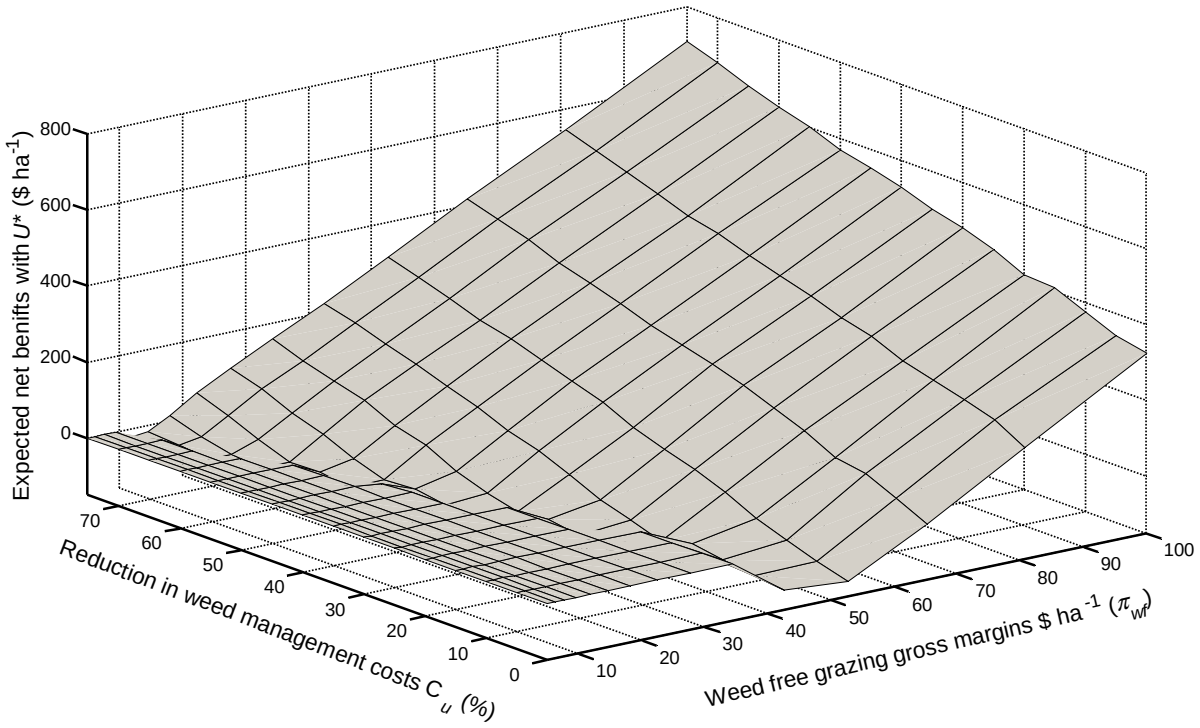


Figure 7-8: Expected net benefits of U* with respect to changing C_u and π_{wf}.

It is unusual to see results similar to Figure 7-8 with SDP models, these results have come about from using **U***, which is based on the reduced population state (**z**), against the full (ecological) stochastic population model. The results of most SDP models are verified against the same states used within the SDP model. In such cases, the model should result in expected net benefits for all states. In this research problem the possible population states are far greater than those that can be used within the

SDP model. One option is to simply reduce the number of states within the SDP and assume that that state of the real population falls into one of these states. In this case the full state definition was used (**x**) instead of the reduced state (**z**). As indicated in Figure 7-8, **U*** can result in net-losses when the benefits are marginal. This problem is due to the truncation of the states being too coarse and the predicted transition between population states within the transition probability function (TPF) (within the SDP) being exaggerated. In reality, an infestation may require two years to go from truncated State-A to State-B. However, the TPF may have estimated that it takes only one year. One solution is to increase the number of states so that the infestation can be placed into a state between State-A and State-B. Time constraints, though, did not allow for the exploration of finer state grids, due to the curse-of-dimensionality (Bellman, 1957). Additionally, the extra computation and burden may not be of any real benefit out in the field, as measuring the state of a population to such accuracy is generally impractical. The best alternative is to accept **U*** only when the net benefit is ≥ 0.

In the remaining analysis the set of optimal weed control decisions **U*** is constrained to those that result in a net benefit ≥ 0. Where it is equal to zero, the infestation is ignored. With this constraint, a single threshold between **U*** and ignoring upland chinee apple infestations can be defined for any given combinations of $\mathbf{C}_u$ and π_{wf} (Figure 7-9). Whilst Figure 7-9 provides a graphical representation of the effects of π_{wf} and $\mathbf{C}_u$, on the net benefits of **U***, it is difficult to determine actual *NPV* values in a three-dimensional plot. For ease of interpreting the results, a pallet was constructed containing the same information as Figure 7-9, but easier it is to read (Figure 7-10).

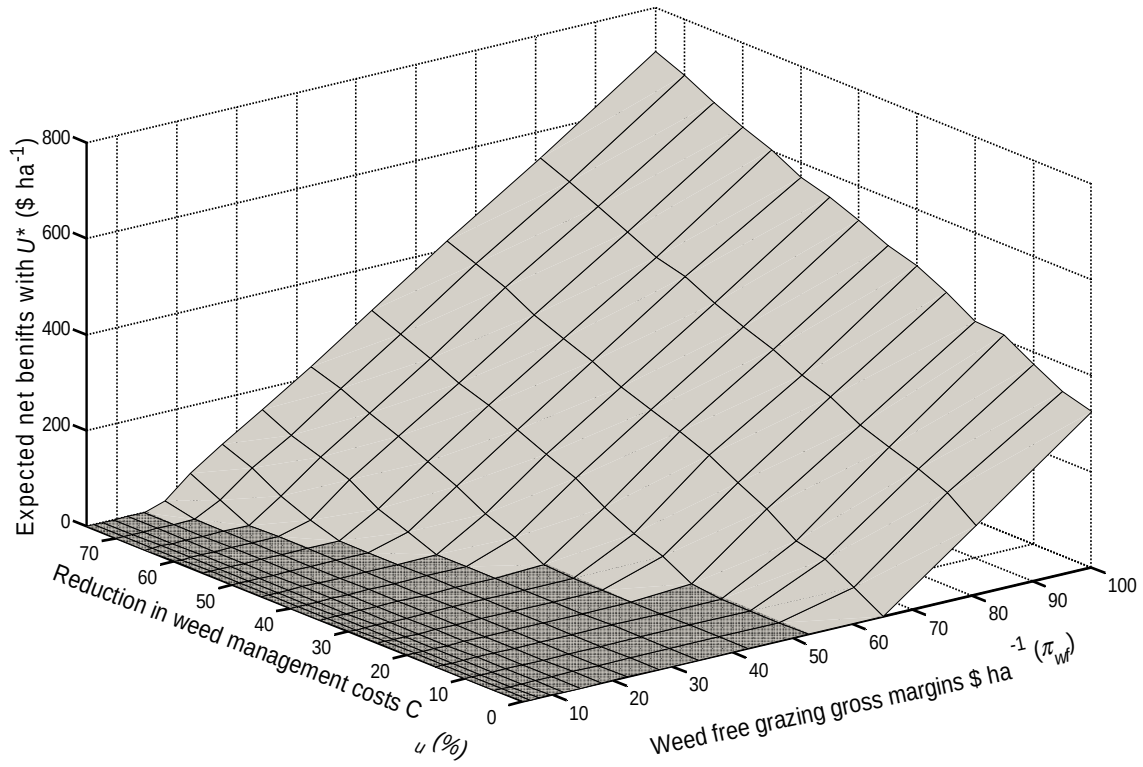


Figure 7-9: Expected *NPV* of the corrected U*, (constraining $NPV \geq 0$), with respect to given weed control costs (C_u) and weed free grazing gross margins (π_{wf}).

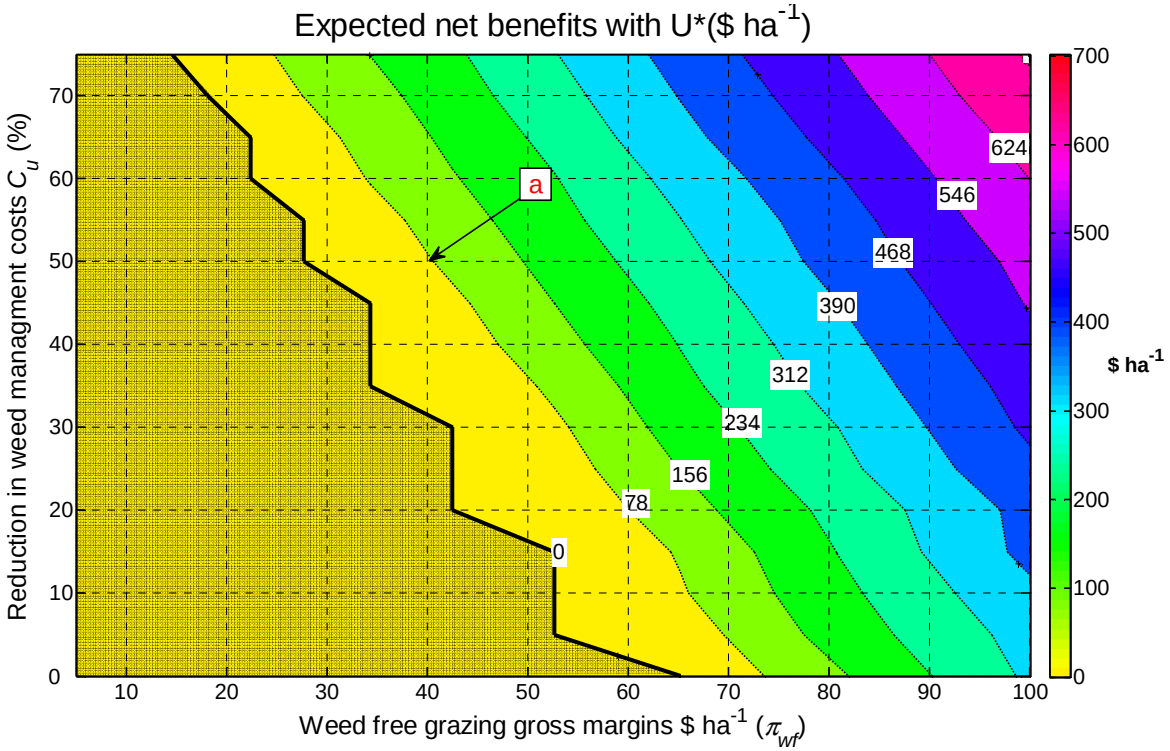


Figure 7-10: Management threshold based on net benefits of U* with respect to weed control costs (C_u) and weed free grazing gross margins (π_{wf}).The bold dark line (zero value) represents the threshold frontier of weed management intervention. Below and to the left of this frontier it is optimal to ignore the infestation.

7.2.6 Biological control

In this section, the biocontrol-SDP model developed in Chapter 6 (Eqs. **6.15** and **6.16**) is used to investigate the merits of biocontrol of woody weeds as part of integrated weed management (IWM) policies. The four weed management strategies are evaluated: (1) ignoring the infestation, (2) solely using biocontrol, (3) using the optimal weed control decisions $\mathbf{U}^*$, and (4) using biocontrol in combination with the optimal weed control decisions $\mathbf{U}^*_{bio}$. This includes no-control, burning, chemical and mechanical removal of plants in addition to biocontrol. The SDP model does not choose if biocontrol should be used. Instead, it chooses the optimal weed control decision in the presence of biocontrol. Therefore both $\mathbf{U}^*$ and $\mathbf{U}^*_{bio}$ are estimated separately and their performance is then compared, to establish if biocontrol is part of the optimal strategy. Figure 7-11 illustrates the effects of using the four different control methods. The assumptions here are similar to those of Figure 7-2, with the same initial infestation, normal weed management costs ($\mathbf{C}_u$), and with free grazing gross-margins of π_{wf} = \$80. The efficacy of the theoretical seed-eating biocontrol agent is set at 0.8 (80 % of new seeds destroyed). Additionally, the setup cost of biocontrol is set at \$10 ha^{-1}, which includes research and release costs. For this case study it will be assumed that there are no detrimental effects to non-targeted species and that any other detrimental ecological effects are minimal. Additionally, the decision to use biocontrol is a single dichotomous decision (yes or no) and is irreversible. The *NPV*s of this particular simulation for ignoring, biocontrol, $\mathbf{U}^*$, and $\mathbf{U}^*_{bio}$ were \$929, \$103, \$1105, and \$1146 ha^{-1}, respectively. All three control strategies resulted in a higher *NPV* than ignoring the infestation, and they all had similar breakeven points (~23 years). These results are contrasted with less effective biocontrol agents in Appendix 4.

Since the weed invasion and control process is stochastic, different simulations will result in different trajectories. It is therefore important to evaluate not only expected values but also probability distributions of weed control strategies (Figure 7-12). Risk can be quantified through the measurement of variance (Peirson *et al.*, 2002). This makes it possible to evaluate the level of risk associated with

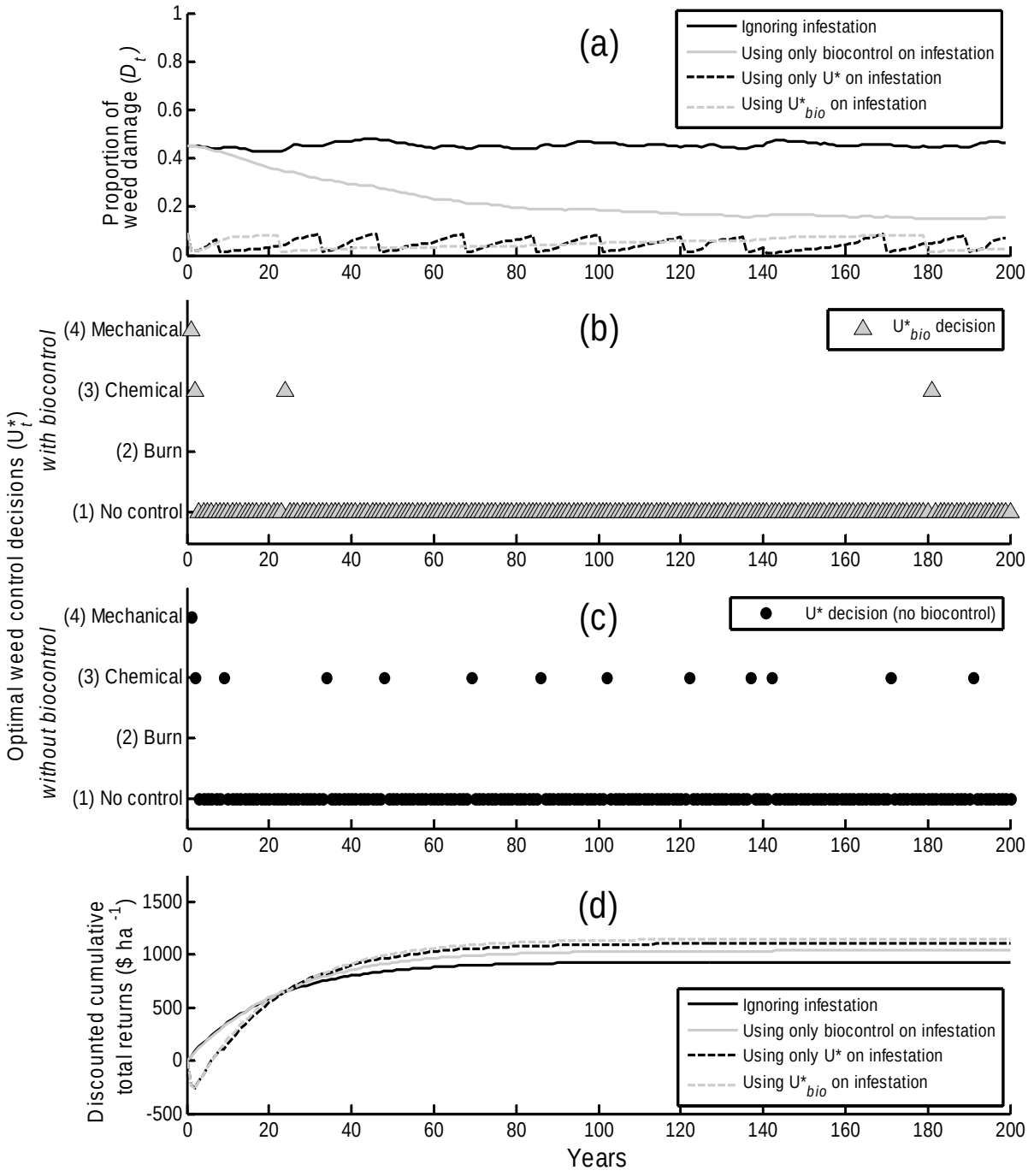


Figure 7-11: A 200–year stochastic simulation of a chinee apple infestation in an upland zone, population started from steady-state, climatic conditions were randomly selected. Simulations are based on normal control costs (C_u), weed free grazing gross margins π_{wf} = \$80, a biocontrol efficacy of 0.8 and a biocontrol setup cost of \$10 ha^{-1}. Results are for (a) damage over time from each management strategy; (b) the controls implemented over time for U^*_{bio}; (c) the controls implemented over time for U^*; and (d) discounted cumulative returns from each management strategy.

different control strategies. The returns from grazing a fully infested area will vary because of climatic conditions and are independent of risk arising from weed management control decisions (Figure 7-12). Using management strategy **U*** results in a higher expected total *NPV* (Figure 7-12'c') than from ignoring the infestation. However, there is an increased risk of having lower returns five per cent of the time than with ignoring the infestation, where the lower distribution of 'c' crosses over 'a' in Figure 7-12. Interestingly, using only biocontrol results in a higher expected total *NPV* than ignoring the infestation. It has a similar variance (level of risk). Moreover, biocontrol resulted in higher returns than **U*** five per cent of the time (see the crossing of lines b and c in Figure 7-12). By including biocontrol into the IWM policy ($\mathbf{U}^*_{bio}$), the expected total *NPV* is greatest and the risk is similar to that of ignoring the infestation. In Figure 7-12, notice that curve 'd' shifts to the right and its lower tail moves downwards compared to curve 'c'. That is, risk of from using $\mathbf{U}^*_{bio}$ for weed management is almost zero as it is similar to 'a', which is driven by climatic conditions.

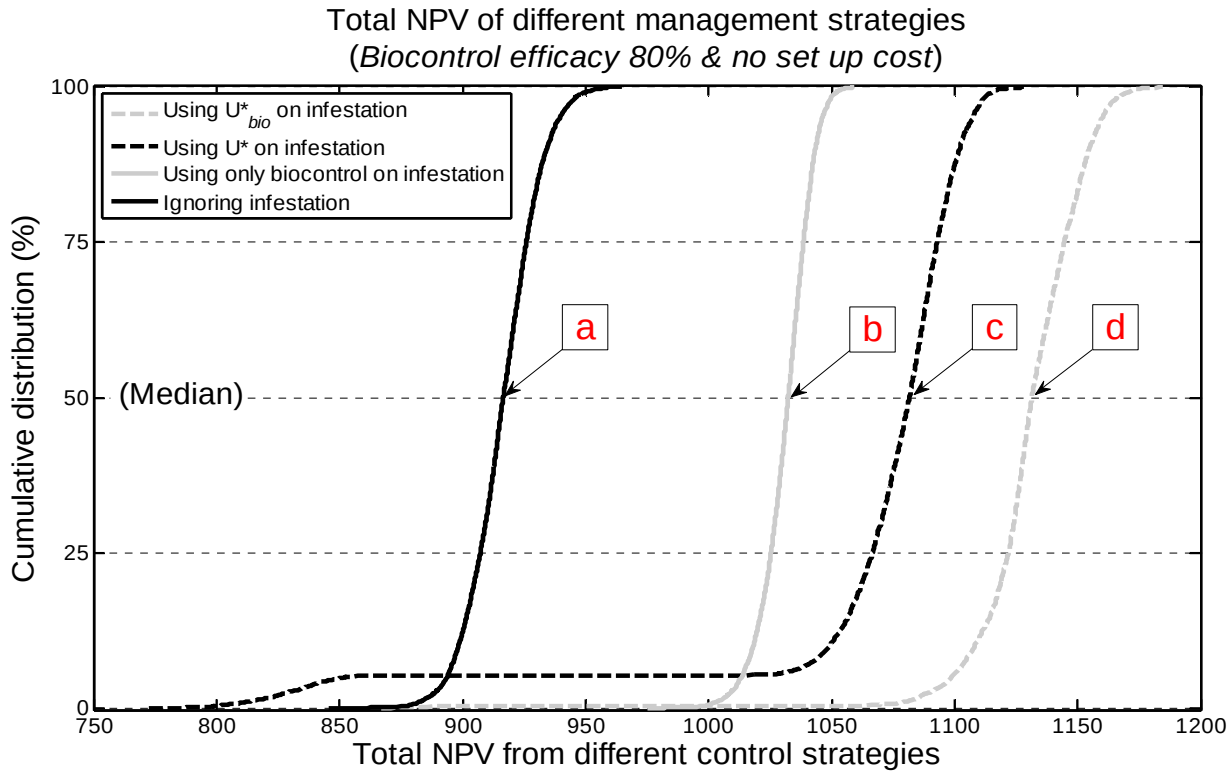


Figure 7-12: Probability distributions of four different weed management strategies. Results are based on normal control costs (C_u), weed-free grazing gross-margins of π_{wf} = 80, and a hypothetical biocontrol agent with an efficacy of 0.8 (with no initial setup costs). The distribution was generated from 5000 stochastic iterations of 200-year simulations of chinee apple in an upland zone. The initial population started from steady-state.

Figure 7-13 shows the change in expected total net benefits of all four control strategies with respect to changes in the biocontrol efficacy. It is evident that increased biocontrol efficacy increases the expected benefits for both biocontrol and $\mathbf{U}^*_{bio}$; however, the two curves do not increase at the same rate $(\eta_1 > \eta_2)$. Although $\mathbf{U}^*_{bio}$ continues to provide the highest return (assuming no initial setup costs), the marginal increase in *NPV* with $\mathbf{U}^*_{bio}$ is lower than with only biocontrol. This reflects the marginal diminishing return to weed control. That is, as the *NPV*s approach the upper limit ($1666 for weed-free grazing over the time period) the marginal cost of weed control increases.

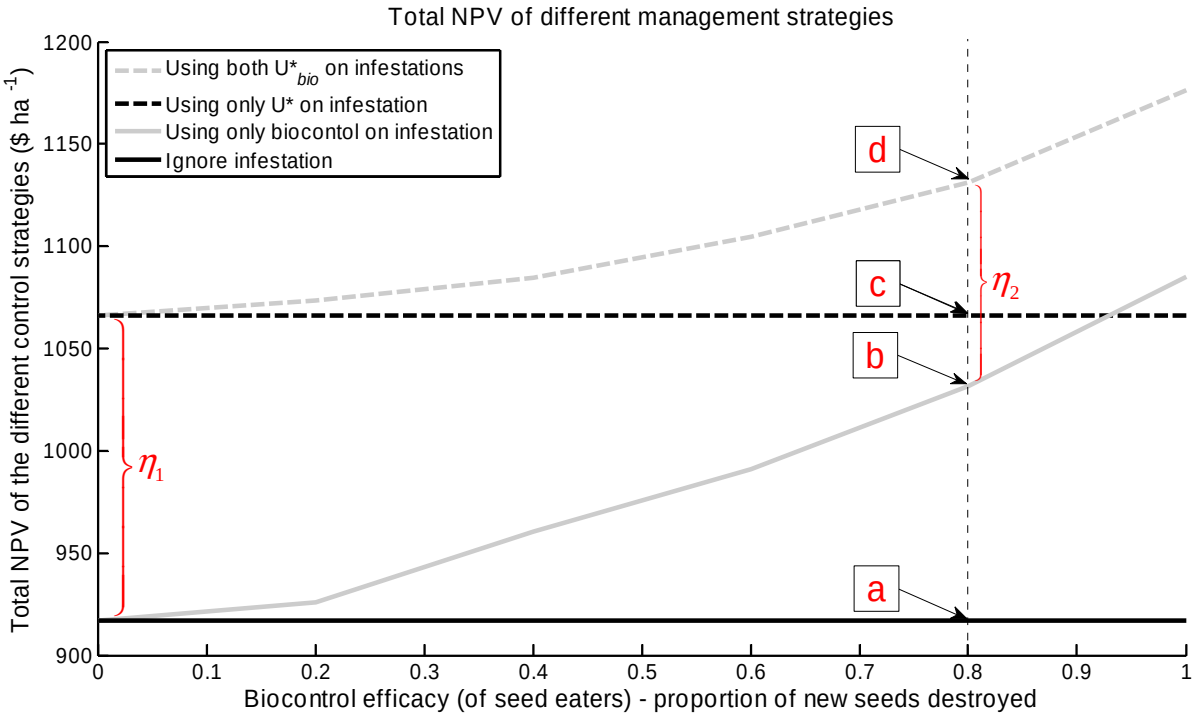


Figure 7-13: Expected (median) total net benefits of four different management strategies. Results are based on normal control costs (C_u), weed-free grazing gross-margins of π_{wf} = 80, and a hypothetical biocontrol agent (without initial setup costs). The expected *NPV*s are generated from 5000 stochastic iterations of 200-year simulations of chinee apple in an upland zone. The initial population started from steady-state.

Note that points a, b, c, and d in Figure 7-13, where the efficacy of biocontrol is 0.8, are the same as those indicated in Figure 7-12. In essence, Figure 7-13 shows the shifting of the expected total net benefit values in Figure 7-12 in response to changes in the biocontrol efficacy. With the inclusion of

biocontrol setup costs, the net benefit of biocontrol and $\mathbf{U}^*_{bio}$ will shift downwards. As the setup cost is incurred when *t*=0, there is no need to discount this value and the shift will be parallel, by the value of the biocontrol setup cost.

Including biocontrol setup costs will result in $\mathbf{U}^*_{bio}$ shifting upwards in Figure 7-13, resulting in $\mathbf{U}^*$ and $\mathbf{U}^*_{bio}$ curves intersecting, this provides a threshold between these two weed managing strategies. That is, if the *NPV* for $\mathbf{U}^*_{bio}$ is greater than for $\mathbf{U}^*$, use $\mathbf{U}^*_{bio}$, otherwise use $\mathbf{U}^*$. This mathematically equates to, use U^*_{bio} if $\left(U^*_{bio}\text{NVP} - \text{setup costs} - U^*\text{NPV}\right) > 0$. This threshold is presented in Figure 7-14; for example, with an initial biocontrol setup cost of \$10 and a biocontrol efficacy of 0.8, the optimal weed control strategy is to use $\mathbf{U}^*_{bio}$ (Figure 7-14 point 'a'). However, if the efficacy is only 0.1, the optimal control strategy would be to use **U*** (Figure 7-14 point 'b'). Additionally, with a biocontrol efficacy of 0.8, the initial setup cost must be less than \$63 ha^{-1} for $\mathbf{U}^*_{bio}$ to be the preferred option.

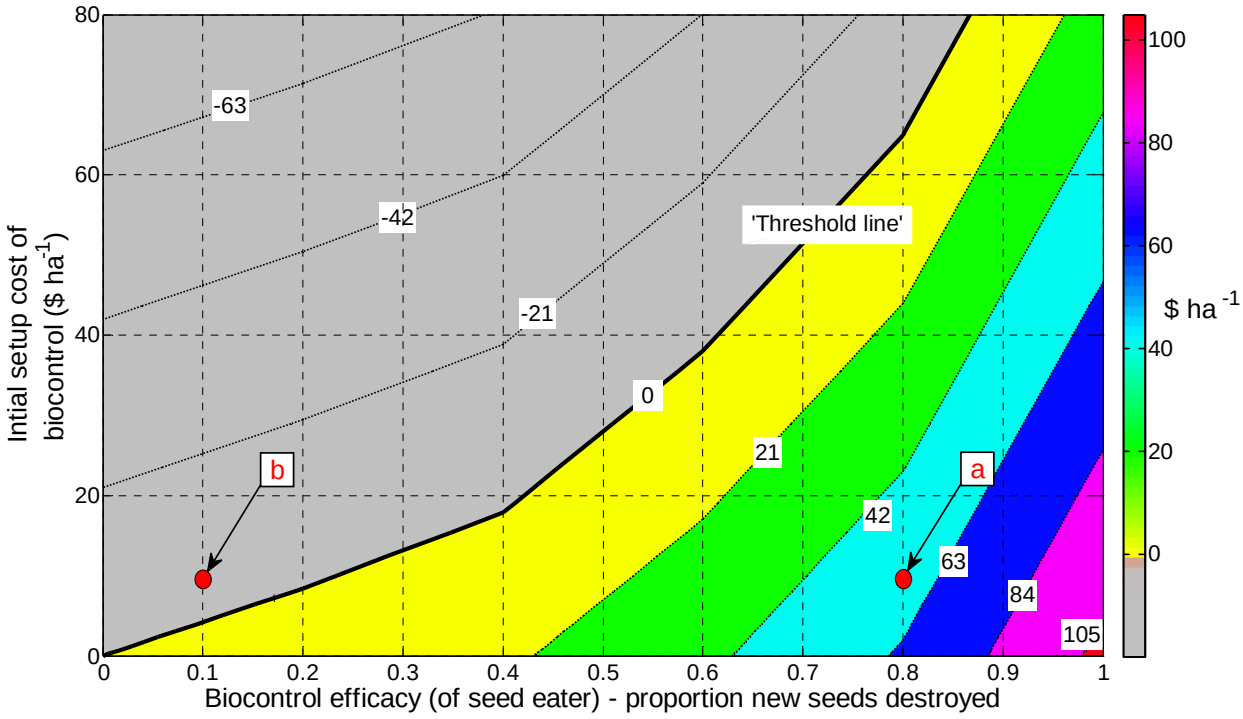


Figure 7-14: Expected net benefit (median) benefits of U^*_{bio} for given combinations of initial biocontrol setup costs and biocontrol efficacies. Results are based on normal control costs (C_u), weed-free grazing gross-margins of π_{wf} = 80, and a hypothetical biocontrol agent.

7.3 DISCUSSION

7.3.1 Thresholds

Pannell (1988a) stated that, instead of adopting a threshold definition with a binary control decision (control or not) based on weed density, a more appropriate strategy would be to ask, 'What is the optimal control strategy given a weed density (and structure)?' This has been labelled an optimal economic threshold (Pannell, 1988a). As the weed population changes, so does the optimal control strategy, and therefore there will be a number of control thresholds. There have been suggestions that the optimal economic solution involves stepping away from threshold theory and use dynamic programming to achieve the optimal long-term solutions (Jones & Medd, 2000). The analysis presented in this chapter is based on an SDP and has in fact provided the long-term optimal thresholds for an IWM strategy of stochastic woody-weed population. Therefore, the two approaches are not incompatible.

In this research, the term "threshold frontier" has been reserved to indicate the set of points beyond which the infestation is managed. All infestations to the left of the threshold frontier are to be ignored (Figure 7-10). However, even if the infestation is managed, it will not be actively treated every year. In fact, beyond the threshold frontier there are many combinations or optimal weed control decisions (i.e. no-control, burning, poisoning, and the mechanical removal of plants). In essence, these represent multiple thresholds where the optimal weed control decision changes between these types of weed controls. For example, if the infestation is to be managed (it is to the right of the threshold frontier) and it may be optimal to use mechanical control, in the following year the infestation will be less dense and no control will be required. Eventually, the infestation goes beyond another optimal economic control threshold, which might then require chemical control. The actual control to be applied for any given state of the infestation is given by the decision rule $\mathbf{U}^*$, as presented in Figure 7-1 and illustrated in Figure 7-2. Additionally, the control decision $\mathbf{U}^*$ will change as weed control costs, control efficacy, and/or weed-free grazing gross-margins change.

7.3.2 Eradication

Grove and Panetta (2002) stated that unless eradication is attempted very early in the invasion process, it is unlikely to be achieved. In the Australian rangelands, discovery and intervention of woody-weed infestations seldom occur early enough to achieve eradication (Grice, 2000). Preliminary analysis indicates that, given normal weed management costs and weed-free grazing gross-margins of \$80 ha^{-1}, eradication is not the optimal option for chinee apple infestations, even when the level of infestation is low. Figure 7-6 also supports this premise, as there is always a level of damage (~ three per cent) even when weed control costs are very low and weed-free gross margins are very high. This is partially due to the temporal impact of the infestation. If a young (new) chinee apple infestation is ignored, it will be many years before it has a noticeable impact on pasture production. Due to the discounting of future values, the benefit of eradicating in years to come is low relative to the eradication costs incurred today, and therefore eradication is not warranted.

7.3.3 Burning

Although Grice (1997) stated that burning is an ineffective management tool for controlling chinee apple, as it only impacts the very young seedlings and the seedbank, the results from the SDP indicate that burning can be an optimal method of control when used in conjunction with other weed management techniques (see Figure 7-1 and Figure 7-2). This is in line with Radford's (2001b) premise that burning can have future benefits, even against fire resistant species. In reality, the question of fire within the Australian rangelands is not simply the difference between "fire" and "nil-fire", but rather the difference between 'non-controlled (wild or bush) fires' and 'controlled fires', as bush fires are inevitable within these systems, but the timing of occurrence is not predictable (Macleod & Noble, 1991). It may, therefore, be optimal to use fire when is has the greatest impact on the infestation, with the added benefit of removing debris. When wild fire does occur, it can be retrospectively incorporated into the IWM strategy, as the fire may change the state of the infestation and change the optimal weed control decision at time *t*. Therefore, fire should not be ignored when managing weeds.

7.3.4 Biocontrol and IWM

To date, the biocontrol of chinee apple has not been attempted (Grice, 2002). It is commonly believed that if a plant population is not seed limited, seed predators will have little impact on the population dynamics of the target species (Kriticos *et al.*, 1999a; Crawley, 1989). However, the model indicates that there are potential benefits from biocontrol, even when the efficacy of seed removal is low (**Figure 3-10** and Figure 7-14), for some combinations of weed-free grazing gross-margins and weed control costs.

Once biocontrol is implemented it will take decades for the infestation to reach a new (lower) steady state (Buckley *et al.*, 2004), which will result in lower weed damage to pasture production. This process can be expedited through the use of other types of weed control within an IWM policy that uses burning, poisoning, and the mechanical removal of plants (Figure 7-11). Whilst it may be true that biocontrol has less effect on non-seed limited populations, these populations can become seed limited by targeting adult (seed producing) plants, and removing significant proportions of the seedbank through burning. The rejuvenation of the weed population is then dependent on smaller adults that naturally produce less seed than their larger counterparts. They will produce even less seed and therefore less offspring in the presence of biocontrol.

Biocontrol not only increases the net benefit of IWM, but it also reduces weed management risk. The main benefits of biocontrol as part of an IWM strategy are the decreased frequency of control (Figure 7-11) and reduced weed management risk (Figure 7-12) in addition to the increased *NPV* (Figure 7-11 (d) and Figure 7-12). This is due to the extended temporal benefits of other control methods, i.e. if $\mathbf{U}^*$ is implement but followed by favourable climatic conditions for weed growth, the infestation may recover faster than expected and the benefits of $\mathbf{U}^*$ are less. However, with biocontrol the benefit of other control methods can last longer, even in years with favourable growth conditions; the infestation will not recover as it normally would.

Figure 3-10 indicates that biocontrol can have an impact on an infestation even with low efficacies but the benefits of biocontrol (seed-eaters) dramatically increase when the efficacy is greater than 0.8. Similarly, Figure 7-13 indicates that (assuming there are no-set up costs) even low biocontrol efficacies can result in financial benefits. These benefits increase exponentially as biocontrol efficacies increase. However, one of the primary drawbacks of using only biocontrol is the time it takes to affect the infestation. Figure 7-11(a) indicates that it takes approximately 200 years for the infestation to reach the new lower steady-state. Figure 7-11(d) shows that, due to discounting, any benefits beyond the 100–year mark have little current financial benefit.

7.3.5 Modelling scale

Weed population models can be used to represent different areas. The scale chosen for a modelling framework is dependent on a number of factors, including the problem at hand, the solution sought, available information, and computing capacity. Here, it has been implicitly assumed that the population is evenly distributed; this includes the seedbank. New-seed distribution, however, is not truly homogenous, as many seeds will be deposited under parent trees or clumped in animal faecal material (Grice, 1996). To reflect this, the model could be run at a smaller scale but this will come at the cost of additional computational burden. The probability of seedling establishment is dependent on a number of factors: first, the probability of the new seed remaining viable, second, climatic and environmental conditions allowing germination, third, surviving competition with other plant, and lastly, climatic conditions that allow a seedling to survive its first dry-season. The method used to model weed control captures the first two and the last factor. The third factor is only partially accounted for, as the model considers competition between seedling and larger plants. However, the greatest threat to seedlings is not competition with neighbouring seedlings but the survival beyond the first dry-season, due to harsh climatic conditions and competition with already established plants (Grice, 2002; 1998).

The model could have been set at a field scale, but the infestation at this scale is often very heterogeneous. The centre (oldest) part of the infestation has a different population structure to the

outer (newer) perimeter. The model developed here, although not specifically spatial in nature, can provide decision rules for the different sub-zones of an infestation. That is, based on $\mathbf{U}^*$, the central (older) population state may warrant mechanical control, whilst the outer (newer) infestation may be better treated with chemical control. It is through this process that, even if the infestation is assumed to be homogenous on the finer scale of an average hectare, the model can partially deal with the heterogeneity at a field scale.

7.3.6 Weed-free grazing gross-margins

Some of the simulations provided here represent weed-free grazing gross-margins per hectare (π_{wf}) greater than those currently achievable from grazing within the Australian rangeland system. The economic-ecological framework developed here is complex enough without the inclusion of decisions at the whole-farm level and optimising farms' production functions. A generic framework was desired for farms within a region, rather than a model for a specific farm and its production function. Therefore, deterministic weed-free grazing gross-margins were used to represent the aggregate output (profit) of farms' production functions. Consequently, it was important to investigate a range of gross-margin values. Current π_{wf} values are estimated to be around \$4.50, \$6.50, and \$13.21 ha^{-1}, for poor, average and good land conditions, respectively (N. D. Macleod 2008, pers. comm.). As all of these values are to the left and below of threshold frontier (using normal control costs) in Figure 7-10, the model indicates that controlling chinee apple infestations is economically suboptimal. Using base parameters and normal control cost, the SDP-model indicates the weed-free grazing gross-margin required by graziers to control the infestation is about \$65 ha^{-1} (being the threshold frontier) to control upland chinee apple infestations.

There can be situations where an external stakeholder is willing to pay a grazier to manage an infestation. An example would be the prevention of spread on to their property. If an external stakeholder wishes a grazier to manage a 100 hectare upland chinee apple infestation and the current

the weed-free grazing gross margins are π_{wf} = \$13, the level of compensation can be calculated as the difference between the weed free grazing gross margin and the control threshold (π_{wf} = \$65), which is (π_{wf} = \$52 ha^{-1}); being the additional weed-free grazing gross-margin required to undertake control. However, the grazier will also benefit from increased pasture production as damage will drop from ~45% down to ~5%; this equates to a benefit of around \$5.20 (\$13 x 0.4). Therefore the total compensation needs to be ~\$46.80 ha^{-1} annually; or \$4680 annually to control the 100 hectare infestation. However, the grazier may also need to be compensated for the additional risk of IWM due to episodic recruitment, for example using the lower 95 percentile of expected returns (Figure 7-3).

An alternative application of this information can be used to derive the optimal IWM policy based on the grazier's willingness to pay. Currently many grazing properties are undertaking woody-weed control under the assumption that it will result in a net financial loss, and weed control is often for stewardship purposes. This research can assist these graziers in reducing their control cost whilst achieving their desired goal through IWM strategies, based on graziers' willingness to pay. Figure 7-4(a) indicates that the greatest marginal benefit of weed control for upland chinee apple infestations is at ~π_{wf} = \$65, which is also the π_{wf} threshold. Any additional control appears to have minimal marginal benefit to the grazier. Using the above example and assuming the grazier is willing to control the infestation at a net loss. If the grazier were to undertake this control without any compensation, they would incur a net-loss of ~\$4680. However, the grazier has the additional benefit of the weed control being tax deductable (with the usual marginal tax rate of 30%). Therefore the grazier will receive a tax reimbursement of ~\$1400 ha^{-1} for controlling the infestation. Therefore, the willingness-to-pay (accepting a net loss) must be greater than ~\$3280 annually or ~32.80 ha^{-1} for **U*** to be applied[19], otherwise the infestation should be ignored.

[19] This is where the marginal benefit of control is the greatest and the increased willingness to pay is assumed to be linear.

Woody weed invasions are stochastic; therefore it is important to understand not only expected values but also probability distributions of weed control strategies. Risk can be quantified through the measurement of variance (Peirson *et al.*, 2002). The returns from grazing a fully infested area will vary because of climatic conditions and are independent of risk arising from weed management control decisions (Figure 7-12). Using management strategy **U*** results in a higher expected total *NPV* than from ignoring the infestation. However, there is a risk of having lower returns than with ignoring the infestation. But this must also be put into perspective. **U*** does have a more skewered distribution than ignoring the infestation. With an expected total NPV of \$1020 ha^{-1}, but could be \$245 less due to climatic conditions. Whereas, the expected total NPV from ignoring the infestation is \$920 ha^{-1}, but could be \$130 less due to climatic condition. However **U*** will only result in a lower total NPV five per cent of the time, and even then it would be less than \$75 ha^{-1}, for ignoring the infestation in similar climatic conditions. From these results the risk of weed management due to the climatic conditions does not appear to be substantial. Additionally, it was found that with the inclusion of biocontrol into the IWM policy ($\mathbf{U}^{*}_{bio}$) had higher expected total *NPV* and risk is similar to that of ignoring the infestation. Therefore, both weed management decision and future economic research work on the optimisation of woody weeds can be solved deterministically.

7.4 SUMMARY

The SDP model developed in Chapter 6 was calibrated and applied in this chapter to a chinee apple infestation. This model is used to derive optimal IWM strategies. Five different weed control options are considered as part of an IWM policy: (1) no-control, (2) burning, (3) chemical treatment, (4) mechanical removal, and (5) seed eating biocontrol agents.

As part of the optimal weed solution the expected NPVs and optimal damage (in total and for each life stage), along with probability distributions, were obtained for chinee apple in an upland zone. A threshold frontier was also obtained for given control costs and weed-free grazing gross margins. Due to the stochasticity of woody weed populations, weed managers must not only consider the expected return (median benefit) but also the associated risk, the probability that the chosen management strategy will not have the expected results.

A temporal stochastic simulation is presented using the IWM decision rule **U***. The simulation shows under the optimal control strategy. The weed population density, weed damage, and NPVs may change over time. The benefits of including biocontrol into IWM is also investigated, based on initial set up costs and levels of efficacy, to determine a threshold frontier of biocontrol. Overall it has been concluded that the inclusion of stochastic recruitment may not need to be included in the woody-weed management decision as stochasticity does not affect the optimal decision rule. This means that the problem could be solved deterministically.

There is conclusive evidence that controlling woody weeds is not viable when balanced against current grazing returns and weed control options. However, this research can provide an estimation of the compensation required from external stakeholders for a grazier to control an area of woody weeds. Alternatively, the SDP model can provide a set of control decisions based on a willingness to pay.

8 CONCLUSIONS AND FUTURE DIRECTIONS

This chapter summarises the research findings with respect to the eight aims established in Chapter 1. Then an overview is provided of the major research contributions from this body of work, followed by a discussion on research limitations and opportunities, and then some concluding comments are given.

8.1 RESEARCH FINDINGS

Aim 1 – To review both economic and ecological literature to identify current knowledge (and knowledge-gaps) to aid in establishing a bioeconomic framework for the optimisation of woody weed management.

No published research could be found that established an integrated weed management (IWM) strategy for rangeland woody weeds to maximise grazing profits. The strengths and weaknesses of economic and ecological paradigms were evaluated to establish the requirements for an economic framework to optimise woody-weed management within northern Australian rangeland systems. Weed management requires an understanding of the key ecological processes and the associated costs and efficacies of different control methods. The review provides a definition of woody weeds within the Australian rangelands, the methods for their control and management strategies available with current technologies.

The management of rangeland woody weeds differs from annual cropping-weeds in control frequency, the temporal benefits, and importance of plant size. For woody-weed management, it may be many years between controls; two control methods are rarely used in the same year. Additionally, weed management decisions are affected by seedling recruitment, driven by both background and stochastic processes of rainfall at a regional level. These key ecological drivers need to be captured within the modelling framework to optimise weed management strategies. From a modelling perspective, stage

projection matrix (SPM) models and dynamic programming models are in agreeance, as both models capture stochastic processes and the stated transitions of the infestation.

Aim 2 – To construct an ecological density-dependent population model to determine the most vulnerable life stages.

Plant populations often exhibit sigmoidal growth functions in terms of biomass or area occupied, where the population initially grows exponentially and then tends to asymptote due to density dependence (Gotelli, 2001). This is also the case for woody weeds. However, the number of plants will conform to a quadratic function with respect to time. Initially the number of plants increases exponentially, with the population having many smaller plants. As the plants mature and increase in size, the area can no longer sustain so many larger plants. Due to self-thinning, the number of plants will decrease. This is the consequence of natural succession and the plasticity of existing plants capturing area as it becomes available, restricting the recruitment of seedlings. This study determined that the summation of the area occupied by individual plants is the a reliable predictor of woody weed density dependence, based on the principles of Gap theory (Shugart & Smith, 1996).

One problem often encountered with modelling plant population density dependence is overcompensating for shocks and exhibits simulated population oscillation (Buckley *et al.*, 2001). Most woody plant populations do not exhibit oscillating population dynamics as a result of a shock, due to their slow recovery rates compared to faster growing weed species. The density-dependent SDP developed in this research has the ability to accommodate shocks to the system without overcompensating (i.e. no oscillation). Given sufficient time, the population density will tend to a steady state. Additionally, this modelling framework allows for simulated populations to start with any population structure, satisfying Markov chain theorem modelling. The population model showed that the targeting of adults stages results in the greatest increase in pasture production, with the benefits occurring sooner and for a longer time period than from targeting juvenile or seed stages. This type of analysis can aid in the ranking of life cycle stages based on ecological and economic perspectives.

Aim 3 – To derive a woody-weed damage function that measures damages to grazing production and to estimate the temporal benefits to pasture production from different weed control strategies.

To determine when, and to what degree an infestation will affect grazing pasture production requires knowledge of the weed's steady-state occupancy, colonisation rates, and its density-damage ratio. Cropping-weed literature proposes a relationship between weed density (number of weeds) and weed damage, which can be described with rectangular hyperbola functions (Jones, 2003; Cousens, 1985a). The damage from woody weeds was modelled as a rectangular-hyperbola function, with weed density (measured as the summed area occupied) as the independent variable. Damage cannot exceed 100% and tends to asymptote over time.

All three case study weeds (chinee apple, prickly acacia, and parkinsonia) have similar life stages; however, the growth rates and time required by each species to asymptote varies significantly due to biological factors. A single species infestation of parkinsonia is predicted to be the faster coloniser of an area, closely followed by prickly acacia and then chinee apple. However, once the population reaches steady-state, chinee apple infestations result in a slightly higher damage than the other species. Overall, parkinsonia damage occurred first, followed by prickly acacia, and then chinee apple. Whereas chinee apple was the slowest coloniser, it had the highest steady state damage.

Various simulation were also undertaken to simulate different management strategies a steady-state chinee apple infestation in a riparian[20]. The initial simulations removed all individuals in each life stage and allowed the population to recover. The targeting of the different life stages had similar impacts to the infestation as estimated with the elasticity analysis. The simulation also showed that the benefit of controlling different life stages was not only the immediate reduction in weed damage, but also the

[20] It was assumed in the modeling of benefits of control that the pasture grasses had instantly reoccupied the controlled area. In reality there will be a delay in this process; however some the benefit of control is also from allowing cattle to areas within the infestation that was inaccessible prior to control.

longevity of the benefit. Another simulation looked at removing all the adults with different frequencies of control. The frequency of control could be used to have a once-off benefit, continued benefits (maintaining the infestation at a lower lever of damage), or the eradication of the infestation. All three scenarios were provided.

Aim 4 – To investigate whether biocontrol of woody weeds has the potential to return positive economic returns on investment.

Biological control (biocontrol) is often overlooked for woody weed control due to the perception that a clearly noticeable reduction in the infestation is required. In the short term, this rarely occurs (Hulme, 2006; Norris *et al.*, 2003; Sindel, 2000a). From an economic perspective, if the benefits of biocontrol are greater than the costs of biocontrol, then biocontrol can be a successful management tool and should not be disregarded lightly.

The potential benefits of a seed-eating biocontrol agent were investigated using eigenvalue analysis of the seed life stages. A correlation exists between the longevity of the species and the decreased potential of seed-eating biocontrol agents. Similar to Parker (2000), it was found that fecundity needs to be reduced by 95 % in order to significantly reduce population growth rates. However, what is not indicated by this type of analysis is the potential for biocontrol to decrease the steady-state damage. This is in fact the main benefit of biocontrol, where it creates a new ecological balance between the weed and the control agent to a new, lower "general equilibrium position" (steady-state) and level of damage.

An alternative analysis was developed to determine the benefits from biocontrol agents with respect to the decreased steady-state (damage) of the infestation. In all the simulations, there was a linear relationship between biocontrol efficacy and decreased weed damage, until seed-eating biocontrol efficacy was greater than 80 %. After this point there was a sharp decline in weed damage as biocontrol efficacy increased.

The results from the SDP model developed to establish IWM policies indicate the potential benefits of biocontrol, even when the efficacy of seed removal is low. Once biocontrol is implemented, it will take decades for the woody weed infestation to reach a new (lower) steady state (Buckley *et al.*, 2004). This process can, however, be expedited through the use of other types of weed control within an IWM strategy[21]. Whilst biocontrol has less effect on non seed-limited populations, these populations can become seed limited through management strategies. These include targeting adults (with mechanical or chemical control) and the seedbank (with burning). With such control strategies, the rejuvenation of the weed population is dependent on smaller adults, which naturally produce fewer seeds than their larger counterparts. Additionally, this lower fecundity will be further reduced through biocontrol. Biocontrol also resulted in the benefits from other control measures lasting far longer than they otherwise would have. Using biocontrol as part of IWM also reduced the risk of weed control with respect to episodic recruitment due to climatic conditions (see Aim 6).

Aim 5 – To study the dynamics of multispecies infestations with a model and determine the effects of various weed management strategies.

A density-dependent multispecies SPM model was developed to explore weed control. This component of the study sought to rank the (1) species and (2) life stages that had the greatest impact on infestations and to determine if the extra complexity affects management decisions. Initially, an eigenvalue elasticity analysis was undertaken but this failed to identify any other species (and their life stages) than the dominant species at time *t*. Using these results for multispecies weed management decisions is inconsistent with the underlying premise of integrated weed management (IWM) which endeavours to manage different species simultaneously. Various simulations were run to investigate

[21] It is assumed that other control strategies, such as mechanical removal of the adult plants will result in a decreased food source for seed-eating biocontrol agent, resulting in a decreased population of biocontrol agents. However, as there will still be a larger than sustainable population of biocontrol agents the efficacy of biocontrol on the remaining plants will not decrease.

the dynamics of these multispecies infestations and the effect of targeting different species and life stages. The results of these simulations indicated that the elasticity analysis offered little useful information for multispecies IWM policies.

A proportional benefit analysis was developed for multispecies infestations to rank the control of species and life stages with respect to the effect on the infestation. Woody-weed management often consists of using a single method of control targeting similar life stages of different species simultaneously. Consequently, the results from the proportional benefit analysis were summed into three main life stages: seeds, juveniles, and adults. The ranking of the species for control changed with respect to the dominance of species at time *t*. Over time, the benefit of weed control generally moves from the fastest growing species to the slowest (parkinsonia to prickly acacia and then chinee apple). The ranking of life stages for control generally moved from adults, to juveniles and lastly seeds. Only in the very early years of the infestation did seeds have a high ranking.

Aim 6 – To simulate the stochastic processes of woody-weed recruitment based on historical rainfalls.

The episodic woody-weed recruitment model presented in this research considered the effect of climate during both the wet and the dry seasons. A wet season with an above average rainfall results in above average germination rates (Kriticos *et al.*, 1999a; Watson *et al.*, 1997a). However, many of the seedlings will perish during the dry season. If a dry season has an above average rainfall, an above average number of seedlings will survive. Years with above average rainfalls for both the wet and dry-season can result in a larger number of woody-weed cohorts both germinating and surviving the dry-season. During the following wet-season, they have the opportunity to become well established.

Episodic weed recruitment adds risk to management decisions, as it adds uncertainty to how the infestation will respond in future years to current controls. Although episodic events can result in spectacular population explosions, they should not be targeted blindly for management purposes at the expense of ignoring continuous or background recruitment processes, which also impact pasture

production. The population dynamics of long-lived woody plants are slow and will not change greatly between years (Radford *et al.*, 2002; Watson *et al.*, 1997a). Moreover, it takes years for an episodic event to affect pasture production, and it will also take many years for the impact to diminish. Additionally, changes in the infestation tend to be greater for increasing years than decreasing years (Higgins *et al.*, 2000). Although the damage caused by weed infestations is mostly a function of adult plants, the inclusion of episodic processes into weed management decisions is vital, due to the low margins offered from grazing and relatively high weed management costs.

The results of the SDP model resulted in a skewed distribution of net returns due to the initial high investment cost of weed control and episodic climatic events. Even when the infestation is ignored, there is still variance (risk) in grazing gross margin due to climatic conditions; which tend to be normally distributed. The IWM policy without biocontrol resulted in negative net values approximately six per cent of the time. One of the main benefits of using biocontrol was the reduction in risk. When biocontrol was used as part of the IWM policy, the distribution of net benefits became more normally distributed, similar to the distribution (risk) of 'ignoring' the infestation. As a result, even if the IWM without biocontrol results in the same expected (median) return as IWM including biocontrol, the latter may offer lower risks of adverse weed control events.

Aim 7 – To establish an optimising bioeconomic framework for the IWM of woody weeds within the northern Australian rangelands.

The optimising model developed in Chapter 6 included two interlinking models: the density-dependent SPM and the stochastic ecological driver for woody weed recruitment based on historical rainfalls. The framework is designed to capture the stochasticity of the system and complex population dynamics, as well as significantly reducing the 'curse of dimensionality', commonly encountered in DP models (Bellman, 1957). This was achieved through the parsimonious grouping of possible populations and a reduction in the number of states within the SDP model. The development process of the SDP model required weighing the cost of losing valuable population dynamics information against the curse of

dimensionality. Another consideration in establishing possible states is, for control reasons, the practicality of measuring the state of the infestation out in the field.

The modelling framework developed here considers not only the number of plants in the infestation but also the population structure, as this impacts pasture production and the variation in weed-control efficacy. This framework provides a state-contingent based management tool, i.e. given all the known information, if the weed population state is x, then management decisions $u^*\{x\}$ should be undertaken. This control decision does not consider past control events. It is, in the true sense, Markovian (Pandey & Medd, 1991), as management decisions are made independently at each stage in time. The population state variables need to portray the nature of the system (with an implied memory of past events), containing all relevant information to solve the decision problem at each stage. In this way, the model maintains a condition of Markovian independence (Nemhauser, 1966).

The results of the SDP model were consistent with observations within the northern Australian rangelands, where woody weed infestations are intensely controlled for the first couple of years and then controlled periodically. For chinee apple, fire has little effect on all but the seed and seedling stages and is therefore often disregarded as a control option. However, the IWM policy occasionally recommended fire for seedling regrowth and the seedbank in the years following mechanical control.

Overall it has been concluded that the inclusion of stochastic recruitment, does not warrant sufficient risk for it to be included in woody weed management decision or for future economic research work on the optimisation of woody weeds. These problems can be solved deterministically.

Aim 8 – To calculate the level of compensation required by graziers from external stakeholders to implement IWM policies, when the cost of woody weed control exceeds private benefits.

Current weed-free grazing gross-margins (π_{wf}) are estimated to be around \$4.50, \$6.50, and \$13.21 ha^{-1}, for poor, average and good land conditions respectively (N. D. Macleod 2008, pers. comm. (2008)). As all of these values are to the left of the management threshold-frontier (using normal

control costs), the control of chinee apple infestations is economically suboptimal. Using base parameters and normal control cost, the threshold frontier from the SDP model indicates that graziers require a weed-free grazing gross-margin of about \$65 ha^{-1} to control upland chinee apple infestations. Therefore, if an external stakeholder wants a grazier to control a 100 hectare infestation, with a weed-free gross margin of ~\$13 ha^{-1}, the external party would need to pay approximately \$4680 annually[22] to control this infestation. However, the grazier may also need to be compensated for the additional risk of IWM due to episodic recruitment, using the lower 95 percentile of expected returns (Figure 7-3).

8.2 CONTRIBUTION OF THIS RESEARCH

This research has developed a number of different ecological and economic models to derive a framework for optimising woody weed management decisions. The model has two main ecological components: density-dependent SPM and a stochastic recruitment generator to simulate the effects of episodic rainfall. These ecological models were established as an integral component of the bioeconomic optimisation framework for woody weed management, which included a woody weed damage function and a stochastic dynamic programming (SDP) model. A major contribution of this research is the development of a framework to enhance the links between ecological and economic paradigms of rangeland woody weed control.

Ecological and economic modelling

Most density-dependent population models are based on the number of plants, or total biomass per unit area, to indicate the demand for resources (Jones *et al.*, 2006; Buckley *et al.*, 2005; Grice, 1996).

[22] As the π_{wf} threshold is \$65 ha^{-1} and the current π_{wf} is \$13 ha^{-1}, the grazier needs an additional \$52 ha^{-1} to undertake control. However, the grazier will also benefit from increased pasture production as damage will drop from ~45% down to ~5%; this equates to a benefit of ~ \$5.20 (\$13 x 0.4). Therefore the total annual compensation needs to be ~\$46.80 ha^{-1}; or \$4680 annually for a 100 ha infestation.

Although biomass can be used to quantify woody weeds for scientific purposes, it is often impractical for weed managers, as it requires the removing, drying, and weighing of individual plants. Moreover, it is difficult to use in multispecies infestations, where different species have different weights and densities. Using only the quantity of plants, which is often used for weeds in cropping systems, is also problematic for woody weed infestations. Plant size affects competition between plants and damage to pasture production, as well as the cost and efficacies of different controls. A density-dependent SPM model was created to capture the state of the infestation with respect to the number and size of plants per unit area, representing the competition for horizontal space within the landscape (Shugart & Smith, 1996). As Gap theory can be based on the competition for horizontal space, it can be assumed that as the quantity of plants and/or the size of plants in an area increases, the average distance between plants will decrease and the level of completion will increase (Shugart & Smith, 1996; Menaut *et al.*, 1990). An important distinction must be made about the term "gap". It does not refer to the space between the tree canopies (i.e. 1-cover), but rather the gap in the landscape competition. The SPM is also used in a way that considers both the age of juveniles and the size of adults. This model was chosen because there is relatively little long-term ecological information of the population dynamics of most woody-weed species. The input ecological parameters used for this case study were relatively easy to estimate through information obtained from previous studies and expert opinion. As additional ecological information becomes available, it can be incorporated into this framework. This population model is used to simulate the temporal effects of different management strategies on the population structure.

An economic SDP framework was developed to capture the complex population dynamics of woody-weed infestations within the Australian rangelands, as well as reducing the 'curse of dimensionality' to establish a set of optimal control decisions (Shugart & Smith, 1996). One option to overcome the curse of dimensionality within the SDP model is to simplify the ecological population model. This will, however, result in losing vital biological information. Oversimplifying the population model can result in annual errors compounding, thereby invalidating long-term prescriptions. Although the ecological population model (and predicted consequences of a control event) requires additional detailed population structure information, weed control decisions can be based on more general population

information. The use of mechanical removal if the population is mostly adult, or chemical control if dominated by juveniles, is an example. The method derived in this study retains the biological information within the ecological model whilst grouping similar life stages for the SDP model. The parsimonious grouping of possible population states within the SDP model must weigh up the cost of losing valuable population-dynamic information within the SDP model against the curse of dimensionally. With this modelling approach, it is possible to capture the finer scale interactions between individual weeds in each life stage and their relevant life-cycle stage, whilst allowing the control decision to be made at a more general (coarser) ecological scale.

Identifying woody weed thresholds and control decisions

Until now, it has been unknown if woody-weed control in Australian grazing systems is cost effective and what the optimal integrated weed management (IWM) strategy should be. In cases where weed control is not cost effective, control may still be undertaken for land stewardship purposes, legislative requirements or through external parties. In such cases, knowledge of the most effective and efficient management strategies is desirable.

This study has provided a framework to identify both the threshold frontier and a set of control decisions based on the state of the infestation, weed-free grazing gross-margins, along with the cost and efficacy of the different types of weed control. The threshold frontier is provided for various combinations of weed-free grazing gross-margins and control costs. This research has quantified that currently it is economically suboptimal to control chinee apple infestations. Based on the results presented in Chapter 7, it is now possible to estimate the level of compensation required by graziers from external stakeholders to undertake control (Aim 8). The threshold frontier can also be used to derive the optimal IWM policy based on the willingness to pay by a grazier for land stewardship purposes. This economic framework may assist in evaluating strategies to reduce control cost whilst achieving desired management goal through IWM strategies.

Potential benefits of biocontrol on woody weeds

It is commonly assumed that biocontrol has little to offer in controlling woody weeds. This research challenges that assumption. It has identified opportunities for its use as part of IWM. Human perception often restricts the adoption of biocontrol, as it does not result in spectacular decreases in populations (Vitelli & Pitt, 2006; van Klinken, 2006). Primary benefits of using biocontrol for woody weeds are the long-term reduction in damage to grazing, little or no ongoing costs and its application over vast areas (Julien, 2006; Kasasian & Walker, 1971). This study has shown there is an opportunity to use biocontrol agents to decrease the steady state of the infestation. Biocontrol may not result in spectacular decreases in infestation densities, especially for the longer lived woody weed species, but it may result in positive economic returns, even with relatively low biocontrol efficacies. A theoretical seed eating biocontrol agent was used as a case study but other types of biocontrol agents can be used within this framework, such as the use of dieback or defoliating agents on parkinsonia.

This framework can also be used to evaluate the potential net benefits of biocontrol agents before an agent is even sought. Economic analysis is often undertaken after the biocontrol agent is found and released to determine if it was cost effective (Julien *et al.*, 2007; Fowler *et al.*, 2000). This economic analysis can be used in the early stages of biocontrol research to determine if the potential biocontrol agent is likely to be cost effective, or even to assist in estimating the research budget of possible biocontrol agents. That is, the budget must be less than or equal to the aggregate net benefit over a given or potential area. Another key finding is the reduction in climatic risk of IWM due to the interaction between the biocontrol agent and the infestation. Moreover, IWM can result in the benefits of biocontrol being brought forward and biocontrol extends the temporal benefit of other controls.

Dynamics and control of multispecies infestations

A multispecies density-dependent SPM model was constructed to investigate the dynamics of managing these infestations. Although weed control will decrease the overall damage of a multispecies infestation, it is possible that species within the infestation benefit from the control event, due to

reduced competition. With this model such events can be anticipated and incorporated into the IWM strategy. The multispecies model indicated that initially, the faster growing (shorter lived) species will tend to be the pioneer species infesting the area. However, the longer lived (slower growing) species will tend to out-compete the other species over the longer-time horizon and will be the successional species. This concurs with empirical studies that found chinee apple to be the most competitive species at plot and transect scales (Lawes & Grice, 2010). In general, it is best to target the adult stages of a multispecies infestation as this will have the greatest effects on the infestation (and reduce fecundity), then target juvenile stages and finally the seedbank. It should be expected that after the initial control of adult plants (especially with mechanical control), there can be a regrowth pulse (especially in the faster growing species), due to the reduction in competition from the previously dominating adult species.

Due to the rapid regrowth and lower control efficacy of multispecies infestations relative to single-species infestations, it is even less likely that control of multispecies infestations will be cost effective. As a general rule if there is a species within a multispecies infestation that is deemed economically unmanageable, it can be assumed that the multispecies infestation is unmanageable.

Evaluating land prices

Grazing land with woody weed infestations sell for much the same price as weed-free land, despite the decrease in productive output, as future speculative land prices tend to drive current land prices (Reichard & Hamilton, 1997; Miller, 1996). The damage function developed in the research can be used to estimate the real value of infested land. Similar to other capital investments, grazing land should be valued based on the net present value of all future profits. This model can help to estimate current and future income streams, based on the current state of the infestation, assisting in estimating the real productive value of the land.

8.3 RESEARCH LIMITATIONS AND OPPORTUNITIES

Spatial spread of infestations

Prior to this research there was no existing bioeconomic optimisation framework for woody weed management in rangeland systems. The models developed in the research brought together the many of the relevant economic and ecological processes; however, spatial spread was not included due to time, financial and space limitations of the project.

The spatial structure of an invasion may have important implications for control strategies and management weed management decisions. In some cases, controlling and/or containing a small area may produce widespread benefits by preventing a larger weed-free area from being infested. This means that, even if the cost of control is higher than the benefit for the controlled hectare, it is possible for the total benefit to exceed total cost because benefits per hectare are multiplied by a larger area. A spatial model would be able to capture these factors. Another reason for considering spatial heterogeneity is to determine whether it is worth concentrating efforts in certain areas, or applying different management strategies in different areas. Producing a spatial model requires the introduction of a dispersal kernel in addition to the local growth (matrix) model, which is now available. The drawback of the spatial approach is in its application to the SDP model, which will have a large number of infestation states (life-stages and number of discrete spatial areas) to solve the problem. Simpler spread models have been developed (Moody & Mack, 1988) but these have not looked at the productive capacity of the land.

The model presented in this research is designed for generic application at either the farm or regional level. However, a spatial optimisation model will generally be designed specifically for the farm or region of interest, as the dispersal kernels and habitat suitability are typically used for the different types of modelling. At the farm level, the spread is likely to be modelled within and between fields. At the regional level, spreads through the landscape would be based on soil types, land conditions, and natural and man-made features such as waterways and roads.

Although the model developed in this research is not spatially explicit, it can still be used for spatial decisions. Parameter values were estimated for riparian and upland zones and these could be divided into sub-zones with different land characteristics (landscape heterogeneity) and invasion intensity (states). In this way, a larger infestation can be separated into sub-infestations based on their state and treated accordingly. The centre of an untreated infestation is likely to be older and thicker and the optimal decision may be to use mechanical control. In contrast, the perimeter infestation may be in a state that requires chemical control and the outer perimeter may consist of seedlings that may require burning.

Environmental costs of woody weed infestations

Another aspect that was not considered in this research is the environmental cost of these weeds within the landscape (Brown *et al*., 1999). Woody weeds may decrease biodiversity, result in soil erosion, and change fire regimes. Including these environmental damages in the model may increase the benefits of controlling the infestation. Whilst rangeland ecosystems are very important, most of them are unlikely to have a high non-market value to external stakeholders. Therefore, the unit value of these benefits is likely to be low. The environmental value of these systems, however, can easily be incorporated into the model by adding it to the weed-free grazing gross-margins per hectare.

Land condition and prices

Within the Australian rangelands, the reduced pasture production from weed infestations is normally not reflected in land prices, as land price do not change even when they are weed infested and offer a lower pasture production. This is essentially a market failure. As land prices are not affected by the presence of weeds, the end effect on land prices has not been included in the model. If this situation changes, land values can easily be incorporated into the model as a terminal value of the SDP. This may result in lower weed control thresholds and higher levels of weed control.

Optimising multispecies infestations

The SDP model presented in this research was based on chinee apple as a case study species and the optimising model was not used for multispecies infestations. It is theoretically possible to derive the optimal solution for a multispecies infestation but the required higher number of infestation states, due to the different possible combinations of species states, will cause the SDP to again run into the curse of dimensionality. Results from this research also indicate that there is less benefit from managing multispecies infestations than single species infestation. Moreover, if a single species infestation is deemed uneconomic to control and it exists within a multispecies infestation, that infestation is also deemed uneconomic to control. This model can provide information about the feasibility of controlling the individual species within a multispecies infestation, to aid in management decisions.

Socio-economic aspects

There are many socio-economic aspects that may affect farm decisions. These include debt levels, cash-flows, family costs and expectations, personal goals, education, social influences, stage in life, time horizons, and other on-farm priorities (Harrington *et al.*, 1984b). Many of these aspects will also affect graziers' decisions in weed management. These aspects were clearly out of the scope of this study, which focused on applying population dynamics and dynamic optimisation. The inclusion of these social aspects can add another dimension to this ecological-economic modelling framework.

Climate change and other changes

Climate change and other temporal events may result in a shift in the population dynamics of some woody weeds. Carter and Cowan (1993) suggested that increases in CO_2 levels within the atmosphere will enhance the growth of prickly acacia (a C3 species). If such data becomes available, these sorts of changes can be easily accommodated within this modelling framework by changing the population parameters of the different species.

Poisonous woody weeds

The three case study species are non-poisonous to cattle. However, other woody weed species can be fatal to livestock (Parsons & Cuthbertson, 2001; Grice, 2000). The modelling of these species may require a different damage function, where it steps up from non-present (poisoning costs equals zero) to present (poisoning cost is greater than zero), as even a few plants can result in high cost. Similar to the damage function used here, damage increases as weed density increases. This will essentially result in two types of damage, each with its own response function, which can be added together. This may increase the benefit of weed control and the likelihood of seeking greater weed control.

8.3.1.1 Summary of future research directions

The bioeconomic model developed here can be further developed in its accuracy and complexity, by incorporating additional information as it becomes available. Three interesting extensions are: spatial analysis, socio-economic aspects and applications at the farm level.

There are a number of different ways to consider the spatial spread of an infestation within a weed management strategy. Some options include treating the infestation as:

- An invasion front with a diffusion function or single infestation with a radial diffusion function (Skellam, 1951).
- A single infestation with a diffusion and a probable satellite establishment function (Moody & Mack, 1988).
- A farm model, where each field has a spatial spread function and fences or other landscape features operate as barriers.
- Using actual infestation data from remote sensing or geographic information systems (GIS) to derive statistical dispersal kernels and local growth function.

These options have been listed from the most likely to the least likely, to be included within the bioeconomic optimisation framework presented in this research. This is mostly due to the curse of dimensionality of SDP models and current computing capacity.

Social-economics can also affect weed management decisions. For example, a farmer who has higher levels of debt or poor cash flow may not be able to financially undertake weed control, whereas another farmer without these restraints may control an infestation. Even social standing may influence the control of land, for reasons as land stewardship or socially acceptable norms.

Weed management is just one many farm decisions that must be made. But weed management can be incorporated into other farm decisions; for example, the spelling of a paddock for production purposes can be incorporated into effective weed management strategies. Similarly, the size and shape of fields or watering points for cattle affect both cattle production and the spread or control of weeds.

8.4 CONCLUDING COMMENTS

This thesis melds ecological and economic paradigms to derive integrated weed management (IWM) strategies for woody weeds in Australian rangeland grazing systems. The encroachment of woody weeds within Australian rangeland systems poses a significant threat to the 12 billion dollar p.a. Australian grazing industry. To understand the costs and benefits of controlling woody weeds requires knowledge of how these infestations change over time, their impacts on pasture production and their response to management. A density-dependent stage projection model (SPM) was developed for three case study weeds: chinee apple, prickly acacia, and parkinsonia. The model was used to investigate their effect on pasture production and response to weed control. The SPM was then expanded into a multispecies population to investigate the complex relationships between specie, highlighting the need for IWM policies. Woody-weed management policies must also consider the impact climate has on recruitment processes, for which an episodic recruitment model was constructed. A bioeconomic optimisation framework was developed based on stochastic dynamic programming (SDP), to derive the optimal weed control decisions and a management threshold frontier for various combinations of weed control costs and grazing gross margins. The credibility of these economic control decisions has been achieved by maintaining the full stochastic woody weed population model, whilst dealing with the curse of dimensionality often encountered with SDP models. In this way, the models captured key

ecological processes, and maintain economic robustness. The aim of this research was not to derive the optimal IWM strategy for the selected case study weeds. Rather, the aim was to develop a framework for the numerous woody weeds currently and potentially invading rangelands within and outside Australia.

When the optimal decision falls below the threshold frontier, this research also provides information on the degree to which gross margins must increase and/or weed control costs decrease for weed control to be undertaken. If external stakeholders wish graziers to undertake weed control, compensation may be required. An estimation of an appropriate value can now be identified. Alternatively, graziers may undertake weed control for land stewardship purposes and again, this framework can be adapted to maximise weed kill rates (minimise damage) when subjected to a financial budget (cost constraint).

Every limitation of the model identified in the previous section represents an opportunity for future research. There is also the possibility of extending the usefulness of the model without extending the model itself. Using this model currently requires expert computer programming and modelling skills. The next step can be to use this model as a ‘decision support system’ by extension officers, where localised input variables can be included through a graphical user interface (GUI) on a standard personal computer (PC), to provide control decisions onsite and directly to the manager.

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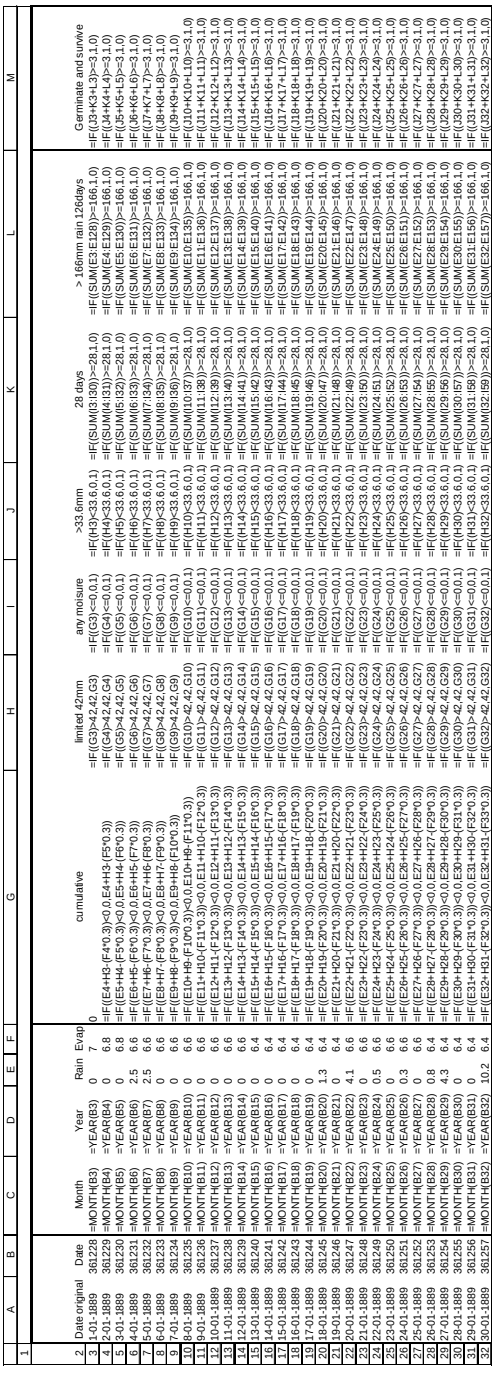
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APPENDICES

APPENDIX 1: CALCULATIONS OF LAWES' RULES



	A	B	C	D	E	F	G	H	I	J	K	L	M
1													
2	Date original	Date	Month	Year	Rain	Evap	cumulative	limited 42mm	any moisure	>33.6mm	28 days	> 166mm rain 126days	Germinate and survive
3	1-01-1889	361228	=MONTH(B3)	=YEAR(B3)	0	7	0	=IF((G3)>42,42,G3)	=IF(G3)<=0,0,1)	=IF(H3>33.6,0,1)	=IF(SUM(I3:I30)>=28,1,0)	=IF(SUM(E3:E128))>=166,1,0)	=IF((J3+K3+L3)>=3,1,0)
4	2-01-1889	361229	=MONTH(B4)	=YEAR(B4)	0	6.8	=IF(E4+H3-(F4*0.3)<0,0,E4+H3-(F5*0.3))	=IF((G4)>42,42,G4)	=IF(G4)<=0,0,1)	=IF(H4>33.6,0,1)	=IF(SUM(I4:I31)>=28,1,0)	=IF(SUM(E4:E129))>=166,1,0)	=IF((J4+K4+L4)>=3,1,0)
5	3-01-1889	361230	=MONTH(B5)	=YEAR(B5)	0	6.8	=IF(E5+H4-(F5*0.3)<0,0,E5+H4-(F6*0.3))	=IF((G5)>42,42,G5)	=IF(G5)<=0,0,1)	=IF(H5>33.6,0,1)	=IF(SUM(I5:I32)>=28,1,0)	=IF(SUM(E5:E130))>=166,1,0)	=IF((J5+K5+L5)>=3,1,0)
6	4-01-1889	361231	=MONTH(B6)	=YEAR(B6)	2.5	6.6	=IF(E6+H5-(F6*0.3)<0,0,E6+H5-(F7*0.3))	=IF((G6)>42,42,G6)	=IF(G6)<=0,0,1)	=IF(H6>33.6,0,1)	=IF(SUM(I6:I33)>=28,1,0)	=IF(SUM(E6:E131))>=166,1,0)	=IF((J6+K6+L6)>=3,1,0)
7	5-01-1889	361232	=MONTH(B7)	=YEAR(B7)	2.5	6.6	=IF(E7+H6-(F7*0.3)<0,0,E7+H6-(F8*0.3))	=IF((G7)>42,42,G7)	=IF(G7)<=0,0,1)	=IF(H7>33.6,0,1)	=IF(SUM(I7:I34)>=28,1,0)	=IF(SUM(E7:E132))>=166,1,0)	=IF((J7+K7+L7)>=3,1,0)
8	6-01-1889	361233	=MONTH(B8)	=YEAR(B8)	0	6.6	=IF(E8+H7-(F8*0.3)<0,0,E8+H7-(F9*0.3))	=IF((G8)>42,42,G8)	=IF(G8)<=0,0,1)	=IF(H8>33.6,0,1)	=IF(SUM(I8:I35)>=28,1,0)	=IF(SUM(E8:E133))>=166,1,0)	=IF((J8+K8+L8)>=3,1,0)
9	7-01-1889	361234	=MONTH(B9)	=YEAR(B9)	0	6.6	=IF(E9+H8-(F9*0.3)<0,0,E9+H8-(F10*0.3))	=IF((G9)>42,42,G9)	=IF(G9)<=0,0,1)	=IF(H9>33.6,0,1)	=IF(SUM(I9:I36)>=28,1,0)	=IF(SUM(E9:E134))>=166,1,0)	=IF((J9+K9+L9)>=3,1,0)
10	8-01-1889	361235	=MONTH(B10)	=YEAR(B10)	0	6.6	=IF(E10+H9-(F10*0.3)<0,0,E10+H9-(F11*0.3))	=IF((G10)>42,42,G10)	=IF(G10)<=0,0,1)	=IF(H10>33.6,0,1)	=IF(SUM(I10:I37)>=28,1,0)	=IF(SUM(E10:E135))>=166,1,0)	=IF((J10+K10+L10)>=3,1,0)
11	9-01-1889	361236	=MONTH(B11)	=YEAR(B11)	0	6.6	=IF(E11+H10-(F11*0.3)<0,0,E11+H10-(F12*0.3))	=IF((G11)>42,42,G11)	=IF(G11)<=0,0,1)	=IF(H11>33.6,0,1)	=IF(SUM(I11:I38)>=28,1,0)	=IF(SUM(E11:E136))>=166,1,0)	=IF((J11+K11+L11)>=3,1,0)
12	10-01-1889	361237	=MONTH(B12)	=YEAR(B12)	0	6.6	=IF(E12+H11-(F12*0.3)<0,0,E12+H11-(F13*0.3))	=IF((G12)>42,42,G12)	=IF(G12)<=0,0,1)	=IF(H12>33.6,0,1)	=IF(SUM(I12:I39)>=28,1,0)	=IF(SUM(E12:E137))>=166,1,0)	=IF((J12+K12+L12)>=3,1,0)
13	11-01-1889	361238	=MONTH(B13)	=YEAR(B13)	0	6.6	=IF(E13+H12-(F13*0.3)<0,0,E13+H12-(F14*0.3))	=IF((G13)>42,42,G13)	=IF(G13)<=0,0,1)	=IF(H13>33.6,0,1)	=IF(SUM(I13:I40)>=28,1,0)	=IF(SUM(E13:E138))>=166,1,0)	=IF((J13+K13+L13)>=3,1,0)
14	12-01-1889	361239	=MONTH(B14)	=YEAR(B14)	0	6.6	=IF(E14+H13-(F14*0.3)<0,0,E14+H13-(F15*0.3))	=IF((G14)>42,42,G14)	=IF(G14)<=0,0,1)	=IF(H14>33.6,0,1)	=IF(SUM(I14:I41)>=28,1,0)	=IF(SUM(E14:E139))>=166,1,0)	=IF((J14+K14+L14)>=3,1,0)
15	13-01-1889	361240	=MONTH(B15)	=YEAR(B15)	0	6.4	=IF(E15+H14-(F15*0.3)<0,0,E15+H14-(F16*0.3))	=IF((G15)>42,42,G15)	=IF(G15)<=0,0,1)	=IF(H15>33.6,0,1)	=IF(SUM(I15:I42)>=28,1,0)	=IF(SUM(E15:E140))>=166,1,0)	=IF((J15+K15+L15)>=3,1,0)
16	14-01-1889	361241	=MONTH(B16)	=YEAR(B16)	0	6.4	=IF(E16+H15-(F16*0.3)<0,0,E16+H15-(F17*0.3))	=IF((G16)>42,42,G16)	=IF(G16)<=0,0,1)	=IF(H16>33.6,0,1)	=IF(SUM(I16:I43)>=28,1,0)	=IF(SUM(E16:E141))>=166,1,0)	=IF((J16+K16+L16)>=3,1,0)
17	15-01-1889	361242	=MONTH(B17)	=YEAR(B17)	0	6.4	=IF(E17+H16-(F17*0.3)<0,0,E17+H16-(F18*0.3))	=IF((G17)>42,42,G17)	=IF(G17)<=0,0,1)	=IF(H17>33.6,0,1)	=IF(SUM(I17:I44)>=28,1,0)	=IF(SUM(E17:E142))>=166,1,0)	=IF((J17+K17+L17)>=3,1,0)
18	16-01-1889	361243	=MONTH(B18)	=YEAR(B18)	0	6.4	=IF(E18+H17-(F18*0.3)<0,0,E18+H17-(F19*0.3))	=IF((G18)>42,42,G18)	=IF(G18)<=0,0,1)	=IF(H18>33.6,0,1)	=IF(SUM(I18:I45)>=28,1,0)	=IF(SUM(E18:E143))>=166,1,0)	=IF((J18+K18+L18)>=3,1,0)
19	17-01-1889	361244	=MONTH(B19)	=YEAR(B19)	0	6.4	=IF(E19+H18-(F19*0.3)<0,0,E19+H18-(F20*0.3))	=IF((G19)>42,42,G19)	=IF(G19)<=0,0,1)	=IF(H19>33.6,0,1)	=IF(SUM(I19:I46)>=28,1,0)	=IF(SUM(E19:E144))>=166,1,0)	=IF((J19+K19+L19)>=3,1,0)
20	18-01-1889	361245	=MONTH(B20)	=YEAR(B20)	1.3	6.4	=IF(E20+H19-(F20*0.3)<0,0,E20+H19-(F21*0.3))	=IF((G20)>42,42,G20)	=IF(G20)<=0,0,1)	=IF(H20>33.6,0,1)	=IF(SUM(I20:I47)>=28,1,0)	=IF(SUM(E20:E145))>=166,1,0)	=IF((J20+K20+L20)>=3,1,0)
21	19-01-1889	361246	=MONTH(B21)	=YEAR(B21)	0	6.4	=IF(E21+H20-(F21*0.3)<0,0,E21+H20-(F22*0.3))	=IF((G21)>42,42,G21)	=IF(G21)<=0,0,1)	=IF(H21>33.6,0,1)	=IF(SUM(I21:I48)>=28,1,0)	=IF(SUM(E21:E146))>=166,1,0)	=IF((J21+K21+L21)>=3,1,0)
22	20-01-1889	361247	=MONTH(B22)	=YEAR(B22)	4.1	6.6	=IF(E22+H21-(F22*0.3)<0,0,E22+H21-(F23*0.3))	=IF((G22)>42,42,G22)	=IF(G22)<=0,0,1)	=IF(H22>33.6,0,1)	=IF(SUM(I22:I49)>=28,1,0)	=IF(SUM(E22:E147))>=166,1,0)	=IF((J22+K22+L22)>=3,1,0)
23	21-01-1889	361248	=MONTH(B23)	=YEAR(B23)	0	6.6	=IF(E23+H22-(F23*0.3)<0,0,E23+H22-(F24*0.3))	=IF((G23)>42,42,G23)	=IF(G23)<=0,0,1)	=IF(H23>33.6,0,1)	=IF(SUM(I23:I50)>=28,1,0)	=IF(SUM(E23:E148))>=166,1,0)	=IF((J23+K23+L23)>=3,1,0)
24	22-01-1889	361249	=MONTH(B24)	=YEAR(B24)	0.5	6.6	=IF(E24+H23-(F24*0.3)<0,0,E24+H23-(F25*0.3))	=IF((G24)>42,42,G24)	=IF(G24)<=0,0,1)	=IF(H24>33.6,0,1)	=IF(SUM(I24:I51)>=28,1,0)	=IF(SUM(E24:E149))>=166,1,0)	=IF((J24+K24+L24)>=3,1,0)
25	23-01-1889	361250	=MONTH(B25)	=YEAR(B25)	0	6.6	=IF(E25+H24-(F25*0.3)<0,0,E25+H24-(F26*0.3))	=IF((G25)>42,42,G25)	=IF(G25)<=0,0,1)	=IF(H25>33.6,0,1)	=IF(SUM(I25:I52)>=28,1,0)	=IF(SUM(E25:E150))>=166,1,0)	=IF((J25+K25+L25)>=3,1,0)
26	24-01-1889	361251	=MONTH(B26)	=YEAR(B26)	0.3	6.6	=IF(E26+H25-(F26*0.3)<0,0,E26+H25-(F27*0.3))	=IF((G26)>42,42,G26)	=IF(G26)<=0,0,1)	=IF(H26>33.6,0,1)	=IF(SUM(I26:I53)>=28,1,0)	=IF(SUM(E26:E151))>=166,1,0)	=IF((J26+K26+L26)>=3,1,0)
27	25-01-1889	361252	=MONTH(B27)	=YEAR(B27)	0	6.6	=IF(E27+H26-(F27*0.3)<0,0,E27+H26-(F28*0.3))	=IF((G27)>42,42,G27)	=IF(G27)<=0,0,1)	=IF(H27)-33.6<0,0,1)	=IF(SUM(I27:I54)>=28,1,0)	=IF(SUM(E27:E152))>=166,1,0)	=IF((J27+K27+L27)>=3,1,0)
28	26-01-1889	361253	=MONTH(B28)	=YEAR(B28)	0.8	6.4	=IF(E28+H27-(F28*0.3)<0,0,E28+H27-(F29*0.3))	=IF((G28)>42,42,G28)	=IF(G28)<=0,0,1)	=IF(H28)-33.6<0,0,1)	=IF(SUM(I28:I55)>=28,1,0)	=IF(SUM(E28:E153))>=166,1,0)	=IF((J28+K28+L28)>=3,1,0)
29	27-01-1889	361254	=MONTH(B29)	=YEAR(B29)	4.3	6.4	=IF(E29+H28-(F29*0.3)<0,0,E29+H28-(F30*0.3))	=IF((G29)>42,42,G29)	=IF(G29)<=0,0,1)	=IF(H29)-33.6<0,0,1)	=IF(SUM(I29:I56)>=28,1,0)	=IF(SUM(E29:E154))>=166,1,0)	=IF((J29+K29+L29)>=3,1,0)
30	28-01-1889	361255	=MONTH(B30)	=YEAR(B30)	0	6.4	=IF(E30+H29-(F30*0.3)<0,0,E30+H29-(F31*0.3))	=IF((G30)>42,42,G30)	=IF(G30)<=0,0,1)	=IF(H30)-33.6<0,0,1)	=IF(SUM(I30:I57)>=28,1,0)	=IF(SUM(E30:E155))>=166,1,0)	=IF((J30+K30+L30)>=3,1,0)
31	29-01-1889	361256	=MONTH(B31)	=YEAR(B31)	0	6.4	=IF(E31+H30-(F31*0.3)<0,0,E31+H30-(F32*0.3))	=IF((G31)>42,42,G31)	=IF(G31)<=0,0,1)	=IF(H31)-33.6<0,0,1)	=IF(SUM(I31:I58)>=28,1,0)	=IF(SUM(E31:E156))>=166,1,0)	=IF((J31+K31+L31)>=3,1,0)
32	30-01-1889	361257	=MONTH(B32)	=YEAR(B32)	10.2	6.4	=IF(E32+H31-(F32*0.3)<0,0,E32+H31-(F33*0.3))	=IF((G32)>42,42,G32)	=IF(G32)<=0,0,1)	=IF(H32)-33.6<0,0,1)	=IF(SUM(I32:I59)>=28,1,0)	=IF(SUM(E32:E157))>=166,1,0)	=IF((J32+K32+L32)>=3,1,0)

Figure 1: Formulae used in a spreadsheet for Lawes' Rules of woody weed seedling recruitment and survival using historical rainfall and evaporative data from the Charters Towers region between 1889 and 2007 (BoM, 2008)

APPENDIX 2: THE ALGORITHM FOR THE SDP MODEL

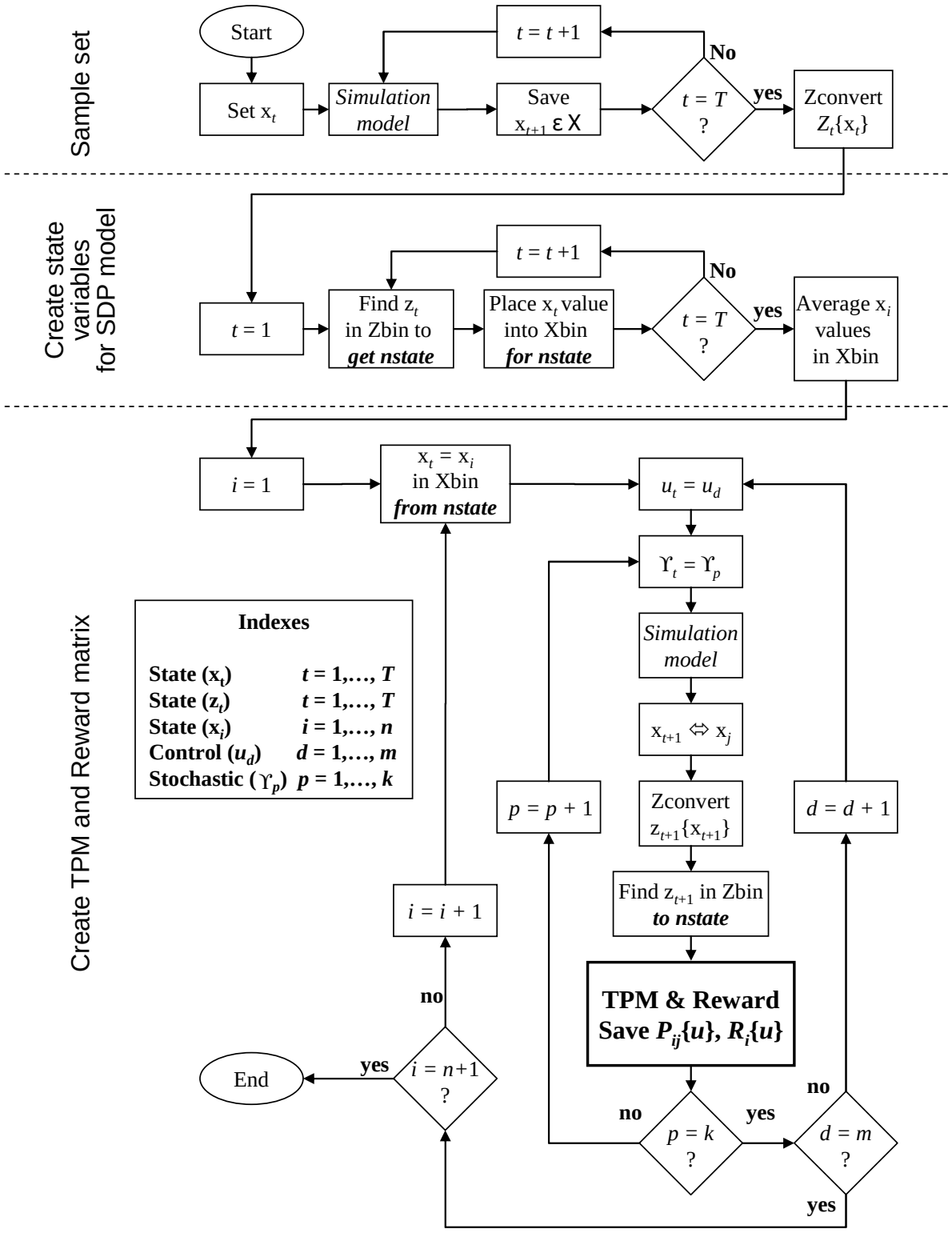


Figure 2: Programming algorithm for a reduced state transition probability matrix (TPM) for long-lived woody-weed management.

To derive the transition probability and reward matrices for long-lived woody weeds:

Step 1. Sample set

1. Get sample population using stage projection matrix $\mathrm{x}_t \in \mathrm{X}$.
2. Condense sample population into 3 main life cycles $z_t \in Z$.

Step 2. Create state variables for SDP

3. Create Zbin matrix (3,*n*) for possible population states
4. Find $z_t;\ t = 1,...,T$ in Zbin to get from state z_i.
5. Get population structure from x_t and place in x_t, average out population structures for all x_i.

Step 3. Create TPM and Reward matrix

6. Set dimensions (*n*, *m*) and initialise state set *Z* and control set **U**
7. Create an array of transition probability matrices $p\{n,n,m\}$ and a reward matrix $R\{n,m\}$ with zero values
8. Solve simulation model: $z_j \Leftrightarrow \mathrm{x}_j = \mathrm{x} + f\{\mathrm{x}_i, u, \Upsilon\},\ \text{for}\ \Upsilon = \Upsilon_1,...,\Upsilon_k$ where *k* is the number of Monte Carlo simulations, with each Υ_i drawn randomly from a lognormal distribution.
9. Use results from 8 to estimate $P_{ij}\{u\}$ given that $\sum_j P_{ij}\{u\} = 1$.
10. Calculate the reward $R_i = \pi_{wf}\left(1 - D_i\left\{x_i, u_d\right\}\right) - C_d$.
11. Repeat steps 8-10 for $\mathrm{x}_i,\ldots\mathrm{x}_n$ to fill up the rows of *P* and *R*.
12. Repeat step 11 with $u = u_1,\ldots,u_m$ to fill up the 3rd dimension of *P* and the columns of *R*.

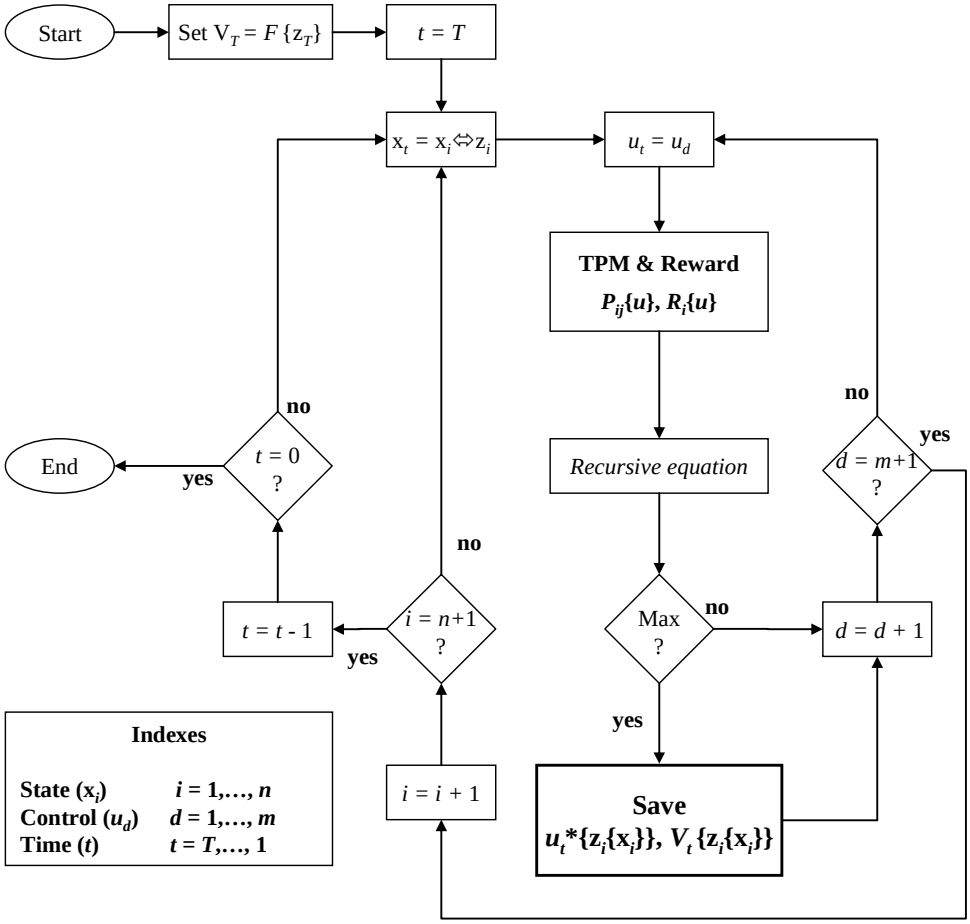


Figure 3: Programming algorithm of a reduced stochastic dynamic programming (SDP) for long-lived woody weeds

Step 4. Solve stochastic dynamic programming problem given state z_i:

13. To perform backward recursion set terminal value $V_T = F\{z_T\}$ and set $t = T$.

14. Apply the recursive equation for all $z_i \in Z$ and $u_t \in \mathrm{U}$, select optimal rule $u_t^*\{z_t\{\mathrm{x}_t\}\}$.

15. Decrease time counter and return to 2 until $t = 0$ or convergence is achieved.

16. Apply the optimal (state-contingent) decision rule $u_t^*\{z_i\{x_i\}\}$ to derive the optimal transition probability matrix $P_{ij}^* = P_{ij}\{u^*\{z_i\{\mathrm{x}_i\}\}\}$.

17. Derive the optimal path x_t^* by performing Monte Carlo simulations for any initial state x_0 and apply the optimal decision rule.

Note, P_{ij}* is the probability of going from state i to state j in the following period given the optimal control decision u^* is applied for x_i.

APPENDIX 3: RISK AND WEED MANAGEMENT DECISIONS

There have been a number of investigations into the effects of uncertainty (risk) on weed control decisions. These uncertainties include the distribution of weeds, initial weed densities, weed competitiveness, weed-management efficacies, production gross margins, weed damage, and future weed populations (Deen *et al.*, 1993; Pannell, 1990b; Auld & Tisdell, 1987). These studies found that the introduction of risk resulted in risk-neutral decision makers decreasing herbicide usage, by either reducing optimal application rates or increasing the weed density thresholds. Pannell (1990b) found that even when some types of risks had low impacts on decision makers, when coupled with other types of risks the compounded risk had a significant impact on risk-neutral decision makers. Results for optimistic (risk seekers) and pessimistic (risk averse) decision makers are ambiguous. Auld and Tisdell (1987) found that optimistic decision makers tended to increase thresholds and decrease control as a result of risk; whilst, pessimistic decision makers tended to lower thresholds and increase control. However, Pannell (1995) found that some risk resulted in decreased control even for pessimistic decision makers. This study will assume risk-neutral management decisions.

As there are different challenges for the management of annual weeds in cropping systems compared to managing woody weeds in grazing systems, so too are there different types risks with different implications. There are four major risks affecting the management of woody weeds, (1) the distribution of weeds, (2) costs of management, (3) weed-free grazing gross margins, and (4) future weed populations due to stochastic events. However, some of these risks can be addressed without including them as stochastic variables into the modelling framework. Each of these risks will be discussed separately.

Pannell (1990b) showed that an uneven distribution of weeds will result in lower level of damage to pasture production than for the same number of weed evenly distributed (Figure 4). Using a rectangular hyperbola damage function of weed densities and weed-damage (Cousens, 1985a), an even distribution of weeds with a density of W_2 results in a weed-damage of D_2. Whereas an uneven distribution of

weeds, where half the area is at density W_1 and the other half at W_3, also has a mean weed density of W_2, (where $W_2 = 0.5W_1 + 0.5W_3$), yet the mean damage is now at D_M, being lower than D_2. Therefore, on average, heterogeneous weed distribution will result in lower pasture production losses and weed management application rates. The uncertainty caused by uneven weed distribution can be reduced by dividing the area up into small cells, with each cell having a given density of weeds and receiving a given level of management. Therefore, instead of looking a whole field as a unit of area to be managed, we can use a lower spatial scale for modelling, such as a hectare. In Figure 4 if half the area is managed for a weed density of W_1 and the other half for weed density of W_3. The average damage to pasture production will now be D_M, even though the average weed density is at W_2. The spatial division of this area reduces spatial distribution risk.

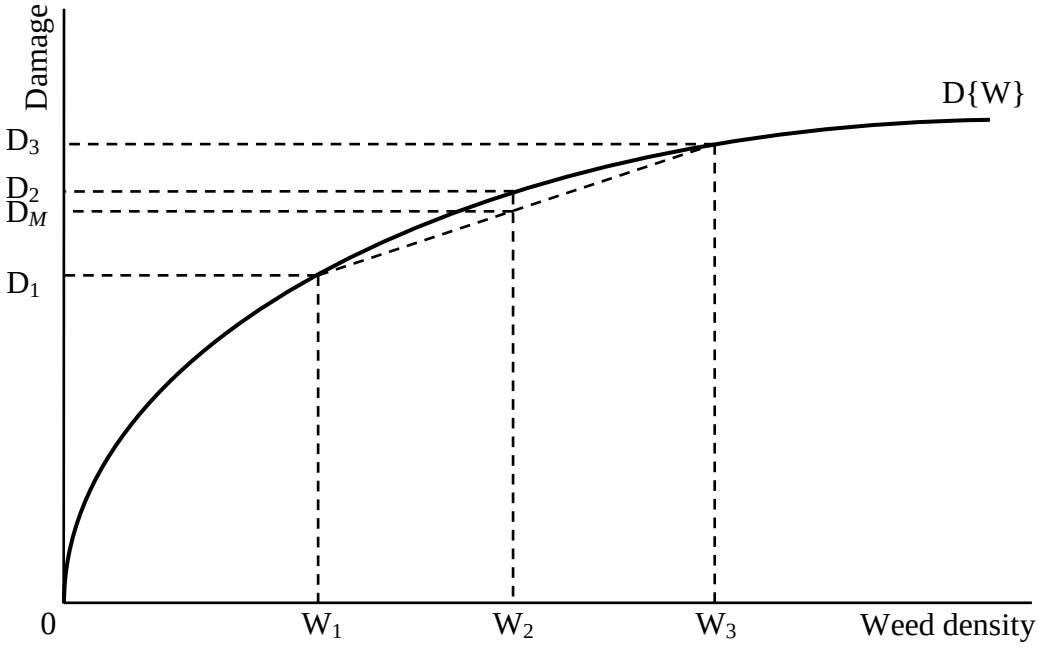


Figure 4: Effects of stochastic weed distribution on pasture production

Another major risk in managing woody weeds is stochastic population dynamics due to climatic conditions. Auld and Tisdell (1987) provide insight into how this may affect management decisions of risk-neutral decision makers. Now consider the net gain from weed management given the relationship between damage to pasture production and weed density $J\{W\}$ (Figure 5), including both current and future benefits, from managing different weed densities (W). If W_2 is the threshold weed density and

the expected gain from management is zero, $E(J) = 0$, and if future densities are known for given management practice, then managing any weed density greater than W_2 will result in a net gain. However, if future weed populations are uncertain, and assuming the weed density is between W_1 and W_3, with an even chance of future infestations being either W_1 or W_3, then the expected population is W_2 (where $W_2 = 0.5W_1 + 0.5W_3$). Now the expected return from managing a weed density of W_2 is negative, $J_M < 0$. Therefore, the introduction of uncertainty of future weed populations increases the threshold of weed density (i.e. a higher density of weeds is required to justify management). Assuming that episodic recruitment of woody weeds is rare, resulting in a skewed distribution. In this case the expected future density of woody weeds will tend towards W_1, with rare episodic recruitment events resulting in W_3, this results in a lower expected return from management and an even higher threshold, with graziers being less inclined to manage infestations.

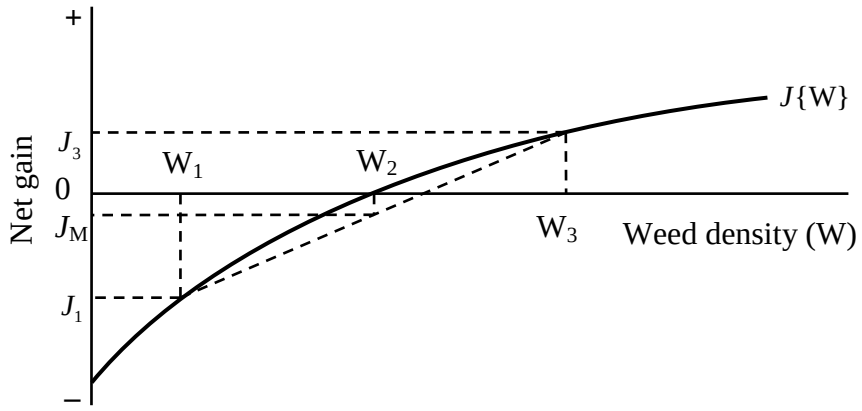


Figure 5: Net gains (*J*) for treating different level of weed density (W), where uncertainty of future weed densities lie between W_1 and W_3, with an equal probability of being either density, taken from (Auld & Tisdell, 1987).

If there is an episodic event next year then discounted net damages will be higher than if the event occurred in 20 years. Future management costs and weed-free grazing gross margins are unknown. However, sensitivity analysis can be conducted on both of these parameters to estimate how changes in these variables may influence the optimal control decision of infestations. If insignificant, they can be disregarded, if not they may required further investigation.

APPENDIX 4: IWM WITH LOW BIOCONTROL EFFICACY

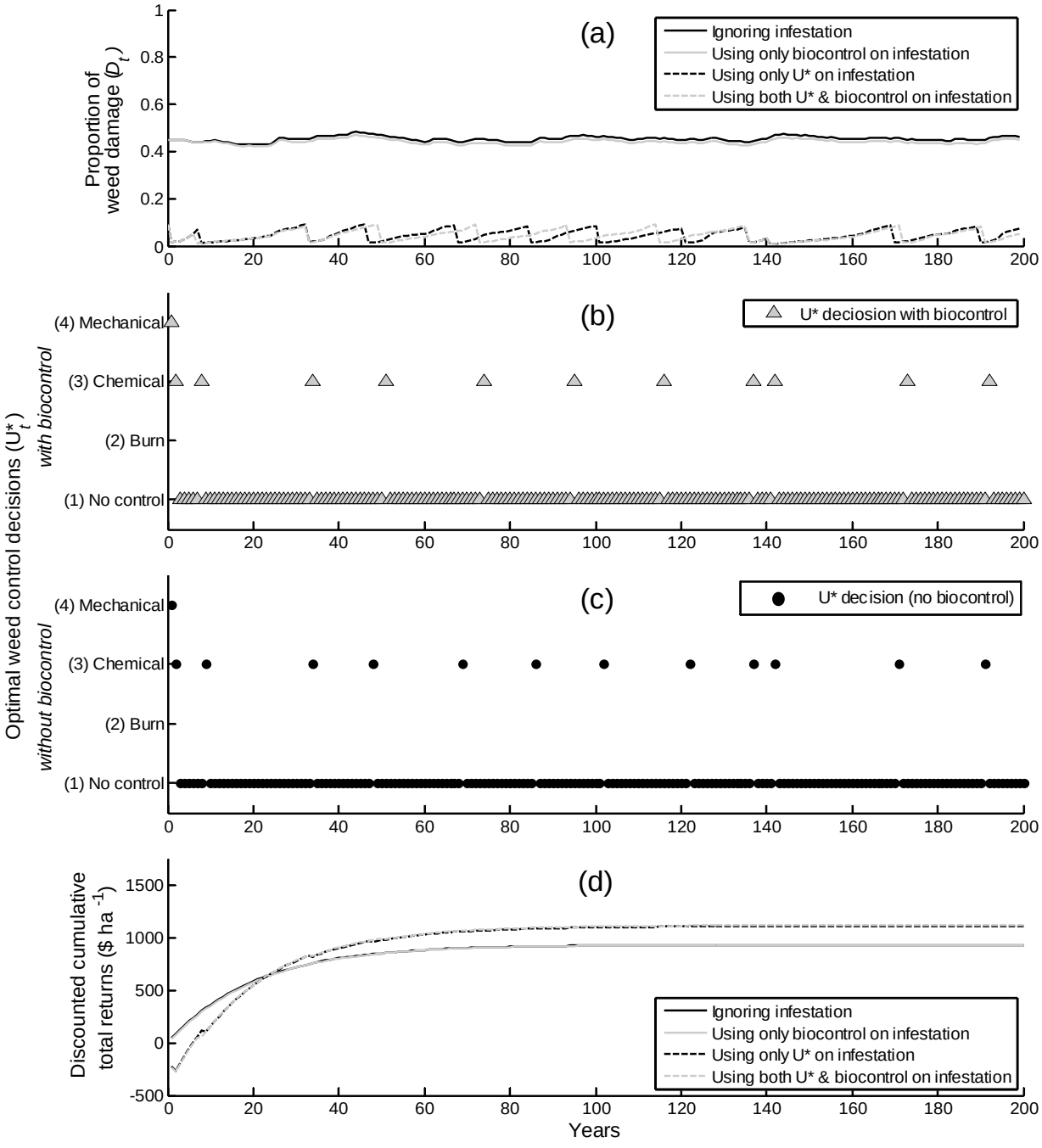


Figure 6: A simulation of IWM strategy that includes a seed-eating biocontrol agent with and efficacy of 10%.