1 Introduction

Invasive plants and dryland salinity both impose considerable costs on the Australian environment and agriculture. Estimates place their annual costs at $4 billion for weeds (Sinden *et al.*, 2004) and as high as $3.5 billion for salinity (CRC PBMDS, 2002). Expectations are for these costs to increase still further because both result from dynamic, and sometimes lengthy, processes.

Dryland salinity could be reduced through the introduction of a range of exotic pasture plant species. With this introduction comes the potential for the new plant to become invasive and thus add to the already high annual costs of introduced plants. How do we decide whether to introduce the new pasture plants or not? Current processes employ a precautionary approach that favours the exclusion of new plants to Australia through consideration of potential costs but not benefits. As the potential gains and the potential costs from the new pasture plants may be high, an alternative decision framework which provides a balanced consideration of costs and benefits is needed. This thesis presents an economic framework as an alternative, which is then applied to the decision to introduce birdsfoot trefoil (*Lotus corniculatus* L.) to the Kings Plain subcatchment within the Border Rivers Catchment of northern New South Wales.

The economics of plant management decisions has received wide attention (Adamson *et al.*, 2000; Cacho, 2004; Cacho *et al.*, 2006; de Wit *et al.*, 2001; Hester *et al.*, 2006; Higgins *et al.*, 1997a; Higgins *et al.*, 1997b; Jones, 2003; Jones *et al.*, 2000; Jones and Vere, 1998; Odom, 2002; van Wilgen *et al.*, 2002; Vere and Campbell, 1979; Vere and Dellow, 1984; Vere *et al.*, 2003). Similarly, the problem of dryland salinity has been considered in a wide range of contexts (Bathgate *et al.*, 2004; Cacho *et al.*, 2001; Cacho *et al.*, 2004; Greiner, 1994; 1996; 1997; 1998; Greiner and Cacho, 2001; Hajkowicz and Young, 2002; Hertzler and Barton, 1992; Kingwell *et al.*, 2003; Mueller *et al.*, 1999; O'Connell, 2003; Pannell, 2001; Pannell *et al.*, 2001; Salerian, 1991). The need to consider plant introductions to reduce salinity and potential weed invasions concurrently has been identified (Bennett and Virtue, 2005), but the problem has yet to be considered fully. The main aims of this study are to

demonstrate the economic framework as an alternative to current processes, and to identify the appropriate decision that will maximise welfare to the community for a range of conditions and plant characteristics.

This chapter introduces the study by providing the background to the problem, followed by an outline of the impacts of salinity and invasive plants in agricultural and natural environment areas. Following this is a brief description of the research approach and an overview of the thesis structure. Contributions of the research then conclude the chapter.

1.1 Background

Dryland salinity results when there is excess recharge water entering groundwater systems. As groundwater levels rise, salts are mobilised and brought near the surface. A range of approaches has been, and is being, investigated to reduce excess recharge of groundwater systems and as such mitigate dryland salinity (hereafter, salinity). These include agroforestry, new agricultural plants, engineering options and new pasture plants.

In general, the agricultural plants being considered for introduction are exotic and permission is required for their importation to Australia. They primarily originate in the Mediterranean basin and Eurasia (Bennett, 2003b). The Weed Risk Assessment (WRA) used by Biosecurity Australia (BA) assesses plants for their potential to be invasive. Plants are allowed for importation, rejected or required to undergo further evaluation on the basis of a score. The WRA process does not consider the benefits of a plant for which introduction is sought which is consistent with the precautionary approach taken to plant introductions internationally and under Australia's World Trade Organisation obligations. New varieties of birdsfoot trefoil (*Lotus corniculatus* L.), hereafter referred to as BFT, have been developed to provide alternative and more persistent pasture options which can reduce the recharge that leads to dryland salinity. These varieties, 'Phoenix', 'Venture' and 'Matador' (Ayres *et al.*, 2008) are among a number of plants being considered for widespread introduction owing to their abilities to provide pasture and reduce excess recharge in farming systems.

As improved adaptations of a previously introduced commercial cultivar, Grasslands Goldie, the BFT varieties studied here do not require permission for introduction. However the release of more persistent lines, together with encouragement of widespread adoption, warrants consideration of their potential to be invasive. Therefore, the new varieties of BFT have been used in the case study application presented here.

1.2 Impacts of salinity and weeds

Salt in soil has the effect of making water less available to plants. Even in wet soil, a plant will experience stress similar to that in a drought when salt levels rise. When salt gets into a non-salt tolerant plant, it causes reduced growth rates, progressive leaf fall, and eventually death (Stirzaker *et al.*, 2000). The main impacts of dryland salinity are manifest as crop and pasture yield losses, biodiversity and habitat damage and saline water run off (Hajkowicz and Young, 2002). It is generally accepted that these impacts will begin when water tables approach two metres from the surface (Heaney *et al.*, 2001; MDBC, 2002).

Salinisation does not only render soils unsuitable for conventional agriculture and affect biodiversity in native ecosystems, but also has the potential to directly erode infrastructure such as roads and buildings, and contribute to increased stream salinisation (Heaney *et al.*, 2001). The impact of salinisation on fauna can be direct through loss of plants that form the diet of a particular species, or indirect through the salinisation of the aquatic environments of animals. These losses flow on to the species that feed on these species and so on. A less obvious effect of salinity is the loss of soil biodiversity (NLWRA, 2001). The location and timing of these impacts, and the effectiveness of management strategies to mitigate them, are uncertain as a result of the highly dynamic and lengthy hydrological processes that lead to salinisation.

Weeds have the effect of damaging the quantity and quality of outputs from the system they invade (Cousens and Mortimer, 1995) as they compete with desired plants (either exotic or native). Introduced plants which have become invasive

impose considerable impacts on agricultural industries, in natural areas and amenities available to society. Examples of such impacts include: increases in the cost of food production, through the reduced productivity of crops and pastures; allergies and sicknesses; poisoning of, and injury to, pets and livestock; blocked and polluted waterways resulting in increased cost of management and decreased water quality; increased risk of herbicide resistance in agriculture; increased risk of bushfires; shelter for feral animals and spread of disease; reduced biodiversity and contribution to the extinction of native flora and fauna; spoilt landscapes with loss of tourist appeal; and reduced use of natural areas for recreational purposes (Cousens and Mortimer, 1995; Groves *et al.*, 2001; Odom, 2002)

The potential for a plant to become invasive is governed by the interaction of factors intrinsic and extrinsic to the plant population (Cousens and Mortimer, 1995). Intrinsic factors such as dispersal characteristics, seed production and rates of population increase are influenced by the occurrence of extrinsic factors such as suitable habitats, favourable weather patterns and presence and strength of dispersal vectors. These interactions mean the dynamics of plant populations will vary considerably between different plants and different locations. This is exacerbated by the potential for extrinsic factors to change, for seeds to remain viable in seed banks for extended periods and for the dispersal of seeds to be difficult to detect and contain.

1.3 Modelling to support an economic decision framework

The objectives of this research are a) to develop an alternative approach to defining whether a plant is a weed by considering competing dynamic resource problems, b) to demonstrate the approach by applying it to a case study and c) to identify the appropriate decisions, including supporting policy, which will maximise welfare to the community for a range of conditions and plant characteristics.

To attain these objectives, two sub-models have been developed and incorporated within a guiding benefit-cost framework using economic surplus as a measure of society's welfare. The sub-models quantify the value of positive and negative changes in economic surplus resulting from the introduction of the new varieties of

BFT, so that the net contribution of the introduction to society's welfare can be estimated.

The first sub-model estimates the benefits of the introduction of BFT to a subcatchment. A linear programming (LP) representative farm model is embedded within a dynamic programming (DP) model to estimate the optimal enterprise mix and resulting maximum profits available with and without the option of BFT over a planning period. The DP model is designed to choose the sequence of agricultural activities that maximises the present value of annual profits for a given initial state: the watertable depth. The model takes account of recharge associated with the various farm options, discharge from the catchment and the net change in the watertable depth in annual stages. The estimated benefits of the introduction of BFT are found as the difference between the optima with and without BFT. Sensitivity analysis is undertaken to identify plant characteristics which influence the value of benefits and the range of potential benefits with respect to subcatchment and plant characteristics.

The second sub-model estimates the costs of the plant's invasion of the surrounding catchment in which the subcatchment is located. A biophysical simulation model of plant spread is developed to estimate the size of the invasion at annual stages. Damage, as a function of the density of the invasion, is estimated annually to calculate the present value of costs over the planning period. The model takes account of BFT population dynamics, density and the value of land use options (agricultural production, biodiversity). Uncertainty is incorporated by applying beta distributions to key population parameters in Monte Carlo simulations. Sensitivity analysis is undertaken to identify plant characteristics which influence the costs of the invasion.

The estimates of the first (*salinity subcatchment*) and second (*plant population catchment*) sub-models are brought together within the benefit-cost framework. Breakeven analysis is undertaken to identify plant, subcatchment, catchment and dispersal characteristics that influence net benefits for a range of thresholds of certainty. From this analysis, approaches to identifying the appropriate introduction decision, and policies that might accompany the decision under uncertainty are

developed. These are applied within a range of contexts and include the evaluation of externalities.

1.4 Thesis Structure

Chapter 2 provides a discussion of the nature of plant populations and the processes to identify plants as 'weeds'. This includes an overview of the processes used in the literature as well as in practice in Australia and other jurisdictions. Evidence for the need to restrict plant introductions and issues pertinent to the modelling of plant populations is presented.

The problem of salinity in agriculture and natural environments is discussed in Chapter 3. The discussion includes details of salinisation processes, evidence to support the need for additional management options and a review of studies that have addressed salinity management problems. Issues pertinent to the modelling of salinity are identified.

The economic framework for the research problem is explained in Chapter 4. The impacts of changes in the salinity state and the plant population state are demonstrated in terms of supply and demand and with this, consideration of uncertainty and externalities to provide the basis for an economic decision criterion for this problem.

The salinity subcatchment and plant population catchment sub-models are then developed in Chapter 5 in the context of the benefit-cost framework derived in Chapter 4. This chapter includes both the mathematical and numerical models. Data and assumptions for the model and parameters specific to the new varieties of BFT as well as the Kings Plain subcatchment and the Border Rivers Catchment are detailed in Chapter 6.

The analysis begins in Chapter 7. Analysis of the benefits of the plant's introduction to the subcatchment is first undertaken, followed by analysis of the costs of the plant's invasion of the surrounding agricultural and natural areas of the catchment. The net benefits of the plant's introduction are then analysed.

The analysis continues in Chapter 8, where the uncertain nature of both the salinity and the invasive plant problem is considered. The range of potential benefits is presented followed by the results of the stochastic analysis of the plant's invasion.

Findings from Chapters 7 and 8 are brought together in Chapter 9 to develop policy approaches that incorporate uncertainty and externalities. This includes both the policy approach to the introduction decision itself and policies to minimise externalities that might accompany an introduction.

Finally, conclusions are offered in Chapter 10. The intention of the study undertaken is to provide a method to help address the conflict that exists where alternative scientific camps are striving to ensure the future of Australia's natural resources and the work of one brings with it the risk of exacerbating another's. The policy recommendations are designed to minimise welfare losses that might occur as a result of the external impacts of an introduction decision. It is hoped the identification of factors which influence costs and benefits will guide researchers with respect to the plant characteristics they seek and as such enhance the possibility that introductions will deliver net benefits.

1.5 Contributions of the Research

This study makes a contribution to the field of natural resource management from a range of perspectives. The study incorporates two concurrent supply shifts within an economic surplus framework and provides analysis of two uncertain and dynamic natural resource issues within a single analytical model. More specifically, the problem of the emergence of an invasive plant population and salinity is analysed together in detail.

The analysis from this study contributes:

- A decision criterion and associated definition of what constitutes a weed that can be used to help resolve plant introduction conflicts.
- An examination of a non-salinity external impact of salinity management and information to help researchers to select further plants for managing salinity.

- A policy framework to accommodate uncertainty within the decision making process and identify conditions when policies accompanying an introduction may minimise the uncertain external costs.
- The groundwork for using this framework to resolve a range of biological introduction decisions from an *ex ante* perspective.

These contributions are elaborated, together with key findings, in Chapter 10.

2 Plants: Transfers, Invasions and Weeds

The transfer of plants, and animals, outside of their native regions has been vital to the welfare of human societies throughout history (Ewel *et al.*, 1999). Opportunities for further transfers exist, including of plants with the potential to deliver enhanced management of salinity. Risks associated with plant transfers are increasingly being recognised, as the costs of plants which have become invasive are realised. Current border protection measures which restrict plant imports limit such risks but they also limit the potential to introduce plants with significant benefits such as those with the potential to enhance management of salinity.

This chapter provides an overview of the importance of plant introductions, the need to restrict them, and a review of methods used to identify which plants might be invasive. This is followed by an overview of the need for a new approach to plant introduction decisions where conflicts exist. To conclude, issues relevant to the management and analysis of plant populations are discussed.

2.1 The importance of plant introductions

Australia has a long history of deliberate plant introductions (Groves, 2004; Lonsdale, 1994) from which it has benefited enormously. Broadacre cropping industries alone, all of which are derived from introduced species, have an annual farm gate value in excess of $10 billion[1] (ABARE, 2004b), while wool and meat production industries, particularly in southern Australia, are reliant on introduced grasses and legumes within pasture systems (Bennett and Virtue, 2005; Groves, 2002). Australia's nursery and garden industries, which are dominated by introduced species, contribute an additional $0.5 billion per annum (NGIA, 2003) as well as the value generated by those enjoying the gardening, gardens, parks and public areas enhanced by this industry.

[1] Includes average value of winter and summer grains (inc. Rice), Cotton and Sugar in the five years to 2002-03.

The benefits of introduced plants, deliberate or otherwise, have been accompanied by adverse impacts on agriculture, the environment, society and the economy (Sinden *et al.*, 2005). The extent of these adverse impacts, and to whom they accrue, provides a sound case for the need to restrict plant introductions.

2.2 The need to restrict plant introductions

Plants that spread and are not controlled can impose external costs (de Wit *et al.*, 2001; Odom, 2002; Pannell, 1994). Individual importers of new plants act on the basis of self-interest and do not take into account costs that are external to them. The presence of external costs therefore supports the need for the government to restrict new plant introductions.

This section provides an overview of invasive plants in Australia and their costs. In doing so, it provides an appreciation of the extent of the external impacts that can result when introduced plants become invasive and the need for restrictions to be placed on plant introductions. The cost of invasive plants in a sample of other countries is also presented for comparison and to illustrate the justification in other jurisdictions for restrictions on plant introductions.

2.2.1 Australia

The benefits of introduced plants, while large, have been accompanied by costs in the form of species which have become invasive. Plants such as blackberry (*Rubus fruticosus* agg.), introduced for its fruit, willow (*Salix* spp.) and bitou bush (*Chrysanthemoides monilifera),* introduced to assist with erosion and hymenachne (*Hymenachne amplexicaulis)* introduced for pasture purposes, were introduced with good intention but are now large contributors to the annual cost of invasive plants in Australia.

There have been in excess of 28,500 plant species introduced to Australia, of which more than 95 percent were introduced intentionally (Randall, 2006). Introductions have been dominated by plants for the horticultural and nursery industries, but also

include plants for pasture, fodder, cropping and forestry. Of total plant introductions to Australia, about 3,000 species are now documented as weeds. Plants introduced for park and garden use account for 65 percent of documented weeds (Table 2.1).

Table 2.1 Purposes for which plants have been introduced to Australia (%)

Category	**All Plant Introductions** (weedy and non-weedy) [a]	**All Weeds** [b]	**Top 20 WONS** [c]
for Parks and gardens	90	65	45
for Agriculture	5	7	30
as a Contaminant of imports	<1	2	10
other	3	6	10
unknown	<2	20 [d]	5
Totals	*100*	*100*	*100*

Source: [a] Adapted from Virtue *et al.* (2004), [b] Adapted from CRC for Australian Weed Management, (2002), [c] Weeds of National Significance: approximation derived from Appendix A, [d] The extent to which the origin of Australian weeds is unknown has been challenged by Cooke & Dias (2006).

While plants escaped from parks and gardens represent the bulk of documented weeds, this is primarily due to the large number of plants introduced in this category rather than their propensity as a group to be weeds. Some 25,000 species, or 90 percent of all introductions, have been introduced for gardening but only 10 percent of these have become weedy (Virtue *et al.*, 2004). By comparison 38 percent of all introduced pasture plants have become weedy (Virtue *et al.*, 2004).

Table 2.1 provides a comparison between the source of all weeds, the sources of Australia's top 20 Weeds of National Significance (Thorp and Lynch, 2000)[2] and the sources of all plant introductions. The relative contribution of each source of plants to each category indicates that plants introduced for agricultural purposes have had a higher propensity to become weeds, because they represent a disproportionately higher number of Australia's significant invasive plants. Selection of plants as new pastures that are fast growing, aggressive, drought-tolerant and prolific seeders is

[2] A systematic ranking process was conducted to assist prioritisation of weed control efforts and led to the establishment of an Australian list of Weeds of National Significance (WONS). Of 71 tropical, sub-tropical and temperate plants nominated for prioritisation, the top 20 ranked weeds are listed in Appendix A.

attributed to this representation because these features are shared by many invasive plants (Paynter *et al.*, 2003).

Initial recognition of the extent of the cost of invasive plants in Australia began with Combellack (1987), who estimated the financial costs of such species in Australia to be $2,096 million annually. This estimate was for gross losses and included both production and control costs for cropping, pasture, horticulture and other land uses (including forests, aquatic weeds, railways, national parks).

A more recent estimate by Sinden *et al.* (2004) assessed the net losses from invasive plants in Australia as changes in welfare. Sinden *et al.* found the annual cost of weeds in Australia to be $4,039 million. The estimate was undertaken in a stochastic framework such that the mean loss was $4,039 million, and the 95 percent confidence interval was $3,554 - $4,532 million. The estimate includes losses to the cropping, livestock and horticulture industries. The estimate does not include lost values associated with invasive plants in natural and public areas, social losses associated with health and amenity costs and a number of additional non-yield costs to agricultural production (e.g. seed contamination). As such the estimate is a conservative value of the actual annual costs to the Australian economy.

Sinden *et al.* (2004) drew on the work of a number of previous authors in their estimate of total annual costs of weeds to Australia. As they pointed out, the majority of previous studies have assessed the costs to agriculture due to the relative ease with which such estimates can be undertaken. Estimates of losses to agricultural industries have been characterised by aggregation of estimates across invasive plant species to obtain industry-wide estimates (see for example Jones *et al.*, 2000 and; Sloane Cook & King Pty Ltd, 1988). Table 2.2 presents a selection of studies where the impacts of a particular weed species have been the focus of cost estimation. These studies illustrate the extent of the costs that have been estimated to result from individual species alone.

Table 2.2 Selected estimates of weed costs in Australia - by species

Species	Estimate	Includes	Author (Year)
Blackberry	$4.7 million per annum in central western NSW	Costs of control and value of lost agricultural production.	Vere and Dellow (1984)
	$41.5 million per annum	Costs of control and lost value of agricultural production across Australia.	James and Lockwood (1998)
Rubber vine	Reduce production by 25 percent, increase management costs by $10/ha and increase mustering costs by 36c/ha.	Medium density infestation costs to beef industry.	Adamson and Lynch (2000)
Wild Oats	$42 million in 1987-88	Losses in wheat production excluding contamination, host pathogen impacts and increased resistance impacts.	Medd and Pandey (1990)
Siam Weed	$291 million per annum.	Expected annual losses to horticultural plantations and beef production in Queensland, if unmanaged	Adamson *et al.* (2000)
Serrated Tussock	$40 million per annum	Cost of reduced livestock carrying capacity in NSW	Jones and Vere (1998)
	$5.1 million per annum	Cost of control in Victoria	Nicholson *et al.* (1997)
	$11.8 million per annum	Net loss in wool income.	Vere and Campbell (1979)

The costs of invasive plants in natural areas in terms of lost biodiversity and the social costs resultant from invasive plants (e.g. human health costs) are reported to be significant (CRC for Australian Weed Management, 2002; Natural Resource Management Ministerial Council, 2006; Panetta *et al.*, 2001), but estimates of these costs are limited. This is largely because data on such impacts are few and causal links difficult to establish (Groves, 2002). However, some studies have incorporated the estimation of the cost of invasive plants in natural areas (for example, Odom *et al.*, 2003a).

2.2.2 International context

The impact of weeds has also been found to be significant in countries such as the USA, New Zealand and South Africa, where studies have been undertaken to assess their cost (Pimentel *et al.*, 2002; van Wilgen *et al.*, 2002; Williams and Timmins, 2002).

Table 2.3 provides a comparison of the reported damage and control costs for Australia, New Zealand and the United States. This comparison shows the actual cost to the United States to far exceed those of Australia and the costs to Australia to far exceed those to New Zealand. But, when considered in terms of each country's gross domestic product, the costs to Australia are around 60 percent higher than those of the United States. Comparison on the basis of costs per hectare of agricultural land reveals the costs to be considerably higher in the United States. However, regardless of the relative size of the problems, these data show that from a range of perspectives, and even on the basis of conservative estimates, invasive plants impose very real costs both in Australia and internationally.

Table 2.3 Weed costs in Australia, New Zealand and the United States

	Australia	**New Zealand**	**United States**
Estimate of total cost per annum (billion)	$A 4.04[a]	$NZ 0.10[b]	$US 34.5[c]
Method of Estimation	Direct costs and net welfare losses from plant invasions of agriculture	Direct costs and gross production losses from plant invasions of agriculture	Direct costs and gross production losses from plant invasions of agriculture and limited incorporation of environmental losses.
Total (billion 2003)	$A 4.042[d]	$A 0.097[d]	$A 52.931[d]
GDP (billion 2003)	$A 707.22[e]	$A 107.50[e]	$A 15,257.64[e]
% GDP	**0.57 %**	**0.09%**	**0.35%**
Agricultural land (million ha)	367[f]	12[g]	396[h]
Per ha per annum	**$A 11.01**	**$A 8.10**	**$A 133.66**

[a] Sinden *et al.* (2004), [b] Williams and Timmins (2002), [c] Pimentel (2002), [d] inflated to 2003 using country CPI and to Australian dollars ($A) on the basis of $A=$US0.72 and $A=$NZ1.12, [e] www.dfat.gov.au/geo/fs and converted to $A on the basis of $A=$US0.72, [f] ABS (2006), [g] www.stats.govt.nz/Hectares Used and Farms by Land Use, [h] USDA (2002a) Farm numbers and average farm size.

Given the varying methods used to determine the costs, different authors and differing relative value of agriculture in each country, care should be taken in interpretation of these comparisons. In particular, it should be noted that the estimate of costs to Australia are net welfare losses as opposed to the gross losses estimated for the USA. Common to each available estimate, however, is the omission or limited inclusion of costs associated with lost biodiversity/environmental values. This results from the general difficulty of measuring the value of the costs of invasive plants.

Attempts to value the impact of invasive plants, both in Australia and internationally, have found costs to be significant. The extent of the costs warrants caution with respect to the potential new introductions but disproportionate caution has the potential to limit access to possible benefits. Whether the right balance is established will depend on the seemingly rudimentary question of '*how do you define a weed?*'

2.3 When is a plant a weed?

"A weed is a plant whose virtues have not yet been discovered"

(American Diary of Organic Gardening, from Riley, 2004)

"A weed is no more than a flower in disguise"

(James Lowell, from Riley, 2004)

"Weeds are plants that interfere with the management objectives of a given area of land" (Sheley *et al.*, 2001).

Processes used to determine if plants can be introduced to new areas are based on implicit definitions of what constitutes a weed. At one extreme, a weed may be defined as *any* plant outside of its area of origin. The definition of weediness or 'what is a weed' varies throughout the literature, but most define weeds in terms of the range of impacts a plant may have.

Richardson *et al*. (2004) for instance define weeds as "plants (not necessarily alien) that are undesirable from a human point of view….usually taxa with detectable economic or environmental effects". Thorp and Lynch identify weeds as those

"invasive plants that impose financial, environmental and social costs" (2000) and Bennett and Virtue (2005) consider a weed to be a plant species that causes negative economic, ecological and social impacts. Bennett and Virtue also highlight the importance of a plant's circumstances and environment with the example of radiata pine (*Pinus radiata*) as a plant species that can be a forestry crop of high value, but also a significant threat to some native ecosystems.

The definitions of a weed, and the processes used to define a weed, employed by international agencies and countries accommodate a range of impacts. The United Nations' Food and Agriculture Organisation (UN FAO)[3] define 'a quarantine pest' as a pest, including weeds, "of potential economic importance to the area endangered thereby and not yet present there, or present but not widely distributed and being officially controlled" (FAO, 1996). The American Animal and Plant Health Inspection Service (APHIS) define a weed as "any plant that poses a major threat to agriculture and/or natural ecosystems within the United States" (USDA, 2001).

On behalf of the Department of Agriculture Fisheries and Forestry (DAFF), Biosecurity Australia (BA) is responsible for undertaking import risk analyses of plants and animals, and their products, proposed for import and introduction to Australia. BA uses a formalised Weed Risk Assessment (WRA) to define which plants have weed potential. The WRA (Pheloung, 1995; Pheloung *et al.*, 1999) employs a series of scored and weighted questions to identify a level of certainty with respect to the weed potential of plants when introduced to Australia. Plants are assessed as being *allowed* for import, *rejected* for import or that *more information* is required for an assessment[4].

Both the BA/WRA and APHIS definitions of weeds, and also their definitions of 'pests' and 'pest risk analysis' approaches, are harmonized with the FAO approach to weeds and pests in general. This is primarily as a result of World Trade Organisation Sanitary and Phytosanitary considerations and obligations.

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[3] At the FAO, responsibility for weeds lies with the International Plant Protection Convention Secretariat.
[4] Further detail relating to the WRA is provided in Appendix I.

Table 2.4 summarises the considerations of Biosecurity Australia, the APHIS and the FAO in terms of a triple bottom line approach and as such includes financial, environmental and social elements.

Table 2.4 Considerations for the assessment of plant introductions

		WRA[a]	APHIS[b]	FAO[c]
Financial	*Costs*	√	√	√
	Benefits	-	-	-
Environmental	*Costs*	√	√	√
	Benefits	-	-	-
Social	*Costs*	√	-	√
	Benefits	-	-	-

Source: [a] Pheloung (1995), [b] USDA (2002b), [c] FAO (1996).

All three processes consider the costs of an introduction in terms of both financial and environmental impacts, and both the WRA and FAO also consider social impacts, though not comprehensively. Social considerations are limited to whether 'perceived social costs such as unemployment' will result (FAO, 1996) and whether a plant will cause allergies or be otherwise toxic to humans (Pheloung, 1995). The three processes consider the uncertainty of costs in limited ways:

- scoring in relation to the likelihood of spread (USDA, 2002b);
- investigation of the species' history of repeated introductions outside of its natural range (Pheloung, 1995); or
- identification of factors which will influence the spread potential after establishment (FAO, 1996).

None of these processes considers benefits and none explicitly considers the uncertainty with which impacts will be realised in the area to which introduction is intended. In general, the decision criterion of these processes can be interpreted as "*if the costs could be high, consider it a weed and exclude, while if costs will not be high,*

do not consider it a weed and allow it". These processes therefore provide for a precautionary approach to plant introduction decisions and reflect the definitions in the literature. This is because they, including the WRA, have an emphasis on the potential costs of introduced plants. They provide for the exclusion of new plant introductions which may exacerbate the already high costs of weeds. However, disproportionate caution also provides for the potential exclusion of plants which may offer benefits further to the already high benefits from previously introduced plants.

2.4 The need for a new approach

The increasing realisation of the extent of the costs of introduced plants has prompted the development of more rigorous approaches to determining if new plants should be introduced. In Australia this included the development and adoption of the aforementioned WRA (Pheloung, 1995; Pheloung *et al.*, 1999) which has an emphasis on the potential costs of a plant's introduction. Predecessor approaches to weed risk assessment in Australia also focused on the costs of a plant introduction (Hazard, 1988; Panetta, 1993).

There is, however, wide recognition that introduced organisms, including plants, have contributed both costs and benefits (Ewel *et al.*, 1999; Groves, 2002; Pimentel, 2002) and it has been noted that future introductions could provide for the continued productivity of agricultural systems, environmental management and economic development (Ewel *et al.*, 1999). As such there is the need for an assessment process which includes balanced consideration of the potential for benefits. This approach might augment the existing WRA process or be used to supplement it in the context of post border plant management decisions.

The ability of the WRA to accurately identify which plants should not be introduced to Australia has been examined (Caley *et al.*, 2006; Hughes and Madden, 2003; NWRAS Review Group, 2005; Smith *et al.*, 1999). Weed risk assessment, such as that provided by the WRA, has been described as one of trade-offs between two wrong decisions (Hughes and Madden, 2003); the costs resulting from a false negative and the costs resulting from a false positive. A false negative result inaccurately predicts a plant will not be a weed and as such allows a plant that will add to the weed

burden, while a false positive inaccurately predicts a plant will be a weed and as such incorrectly excludes the plant and any of the benefits it offers. Therefore any efforts to improve the ability of the WRA to accurately predict which plants will be weeds will reduce the scope for plants with potential benefits to be excluded. However, because this involves a trade off between two wrong decisions, where the prevailing view is that costs are more important than benefits (Hughes and Madden, 2003), a more accurate WRA will not accommodate all plant introduction decisions.

That new plants may not be accommodated by existing decision processes has been identified with respect to new plants with the potential to reduce salinity in Australian landscapes (Bennett and Virtue, 2005; Virtue *et al.*, 2004). Kalisch Gordon (2004) proposed that an economic framework, which identifies the decision that makes the greatest contribution to economic welfare, be developed to resolve the conflict that can arise between the potential costs and potential benefits of a new plant introduction.

The literature includes a range of examples of the application of an approach that includes both costs and benefits[5]. For example, an economic framework was used to identify the best strategy for control of *Acacia mearnsii* (black wattle) in South Africa, where there exists a conflict between the use of the plant commercially and its invasion of natural areas (de Wit *et al.*, 2001). More recently an economic approach was used to identify optimal strategies for the management of *Cytisus scoparius*, L (scotch broom) in the Barrington Tops National Park, NSW (Odom, 2002) and to evaluate the feasibility of eradicating invasions (Cacho *et al.*, 2006). However, the purpose of these economic applications has been on the management of existing plant invasions, and not *ex ante* evaluation for new introductions.

2.5 Issues to consider with respect to introduced plants

Invasive plant populations have a number of characteristics that influence the way introduced plants should be considered. They may be the source of both benefits and

[5] These are a small selection of relevant previous studies and a greater review is provided in Chapter 4.

costs, can impose external impacts, can result in the loss of unpriced goods and can be highly dynamic and uncertain over time. These features are now discussed.

2.5.1 Benefits and costs

Akin to the saying *"One man's meat is another man's poison"* introduced plants may also be described as "*one landholder's productive plant while another landholder's weed"*. Introduced plants may be 'both a boon and a bane to society' (Ewel *et al.*, 1999).

Introduced plants may provide benefits when introduced for one application and offer only costs to another where the land-use and natural resources are different. In the case of individual landholders, it is simple to decide whether to introduce a plant because the landholder can assess its compatibility with their current land-use and its potential benefits: benefits and costs are weighed up and experienced by a single agent.

Where a decision is made on behalf of a number of landholders, as is done by the government when a decision is made at the border, the decision is more complex. The government must take account of the potential positive and negative impacts of the introduction and establish an approach which takes a balanced view, presumably for the increased welfare of the country. The need for the government to be involved in this decision results from the fact that external impacts may result from a plant introduction.

2.5.2 External impacts

If introduced plants are invasive and are not controlled they may invade areas outside of the intended area of introduction. They may escape from gardens into natural environments and agricultural areas, from agricultural areas to other agricultural areas and from agricultural areas to gardens and natural environments.

The impacts of an introduced plant if it does invade external areas may be positive or negative. The concern with invasive plant populations is their negative impacts in

external areas. In agricultural areas, negative externalities might include loss of pastures and reduced yields, contamination of grain crops, death of stock where plants are poisonous and an increased risk of herbicide resistance. In natural and public areas, negative externalities might include reduced numbers, or extinction, of native plants and animals and blocked and polluted streams and rivers all resulting in loss of biodiversity. In gardens, negative externalities might include reduced amenity value and additional management (herbicides and physical weeding). In all areas, other negative externalities might include increased risk of bushfire, allergies and sickness, shelter for feral animals and disease spread and spoilt landscapes with reduced tourist and recreational appeal (CRC for Australian Weed Management, 2002).

All of these impacts may be external to an individual wishing to introduce a new plant. As such, introduction decisions made without the consideration of externalities may result in the loss of a public good: freedom from additional weeds. If external costs are not considered in the analysis of plant populations, decisions may not be made in the interest of increasing the welfare of society and the potential need for policy to minimise the external impacts will not be recognised.

2.5.3 Unpriced values

When negative externalities flow from the invasion of natural areas there is an additional challenge to considering plant introductions. Incorporating the value of these external impacts presents a challenge because the services and outputs of natural areas, broadly defined as biodiversity or environmental services, are not traded within a market and as such are unpriced. Odom (2002) lists the following range of potential impacts of plant invasions of natural areas that are difficult to value:

- Discomfort or injury due to thorns, burns, prickles and pollen;
- Reduced walking and recreational use access;
- Replacement of desired native species, lost plant diversity, reduced native animal populations;
- Increased fire risk;
- Reduced swimming and other water-based recreational opportunities; and
- Reduced fish catches.

Wainger and King (2001) highlight the difficulties and issues that arise with respect to attempts to link these biophysical processes to valuation of benefits, or lost benefits. However an attempt must be made because when values for these impacts are not incorporated in an assessment of a proposed introduction, the decision may not improve the welfare of society.

2.5.4 Uncertain dynamic processes

Environmental conditions and species characteristics will determine if non-indigenous species, including plants, will establish in a new area and result in adverse impacts (Leung *et al.*, 2002). The impacts of invasive plant populations are inherently uncertain because plant populations are dynamic and affected by uncertain factors such as rainfall, predator presence and environmental disturbance. There is also uncertainty relating to the initial transfer or spread of plant material which is affected by uncertainty of transfer vectors.

There is a plethora of factors that affect plant population dynamics. Broadly these are the properties of the species, the environment it is released into and the way it is introduced (Smith *et al.*, 1999). Species properties include seed production, seed dispersal, germination rates, plant survival and seed mortality rates, and these are all prone to uncertain environmental influences (Cousens and Mortimer, 1995). This means that population rates of increase and spread will vary considerably from year to year. Furthermore, plant populations may potentially be sleeper weeds where there is seemingly no population growth for many years, followed by a rapid explosion in the population following a climatic episode, change in land-use or other environmental factor (Cunningham *et al.*, 2003; Woldendorp *et al.*, 2004).

In order to consider whether a plant will deliver costs or benefits, or both, in an area where it is introduced, the potential growth of the population must first be understood. The number of individuals per unit area is the central consideration in the analysis of plant populations (Cousens and Mortimer, 1995) and thus their impact. The population may be expressed in terms of number of individuals, seeds or plants at a

particular time. These variables are known as state variables (Cousens and Mortimer, 1995).

Figure 2.1, adapted from Campbell and Grice (2000), illustrates a simplified version of the lifecycle of a perennial plant population which reproduces via seeds. The state of the population in this example at any given time is represented by four state variables which represent the four relevant cohorts, or group of individuals in a given life stage. The four cohorts represented are the seed bank; the dormant seed bank; juvenile plants and mature plants. Intrinsic and extrinsic factors are embodied within the parameters that drive the lifecycle: germination, survival, mortality and seed production (Cousens and Mortimer, 1995).

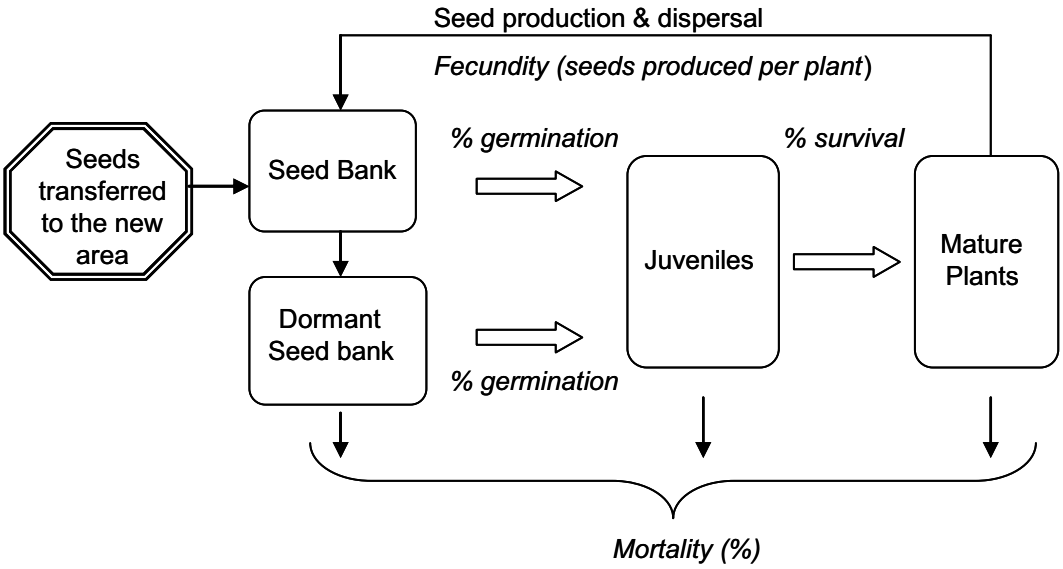

Figure 2.1 Plant life cycle – Adapted from Campbell and Grice (2000)

A plant population requires the initial introduction of plant reproductive material or seeds to initiate a lifecycle in a new area. The dispersal of plant materials into a new area, whether the new area is a paddock, a catchment, state or country, is in most cases dependent on the characteristics of the seeds and the plant as well as the existence of vectors to aid dispersal. Seeds, for example, may be barbed or may float, such that the opportunity to catch on to a passing animal or fall into a river is required for the plant's spread to a new location. Cousens and Mortimer (1995) identify a

number of dispersal vectors: wind; animals; water; and human activities such as tillage and harvesting. Uncertainty of plant population dynamics is enhanced considerably by each of these vectors as well as weather events and other factors that contribute to the effectiveness of each vector.

Invasive plant populations are characterised by their longevity and ability to spread. More specifically, this means plants which become invasive are likely to have (Cousens and Mortimer, 1995; Pysek and Richardson, 2007; Williamson, 1996):

- efficient reproductive capacity and the presence of factors which aid reproduction (e.g. high heat to stimulate seed germination, rhizobia for nodulation);
- efficient dispersal mechanisms and the presence of vectors to aid dispersal (e.g. wind, water, animals to transport seeds or vegetative material);
- tolerance to the environment (e.g. ability to withstand drought, water logging etc); and
- an absence of predators for all cohort stages of a plant's lifecycle (fungal diseases, insects, forage animals).

The interplay of these intrinsic and extrinsic factors underpins the dynamic and uncertain processes that must be considered when a plant is assessed as invasive or not, costs are estimated and management options analysed. Failure to incorporate the dynamics and uncertainty of plant populations in the assessment of a proposed introduction will likely over or under estimate the costs and benefits that may be realised.

2.6 Conclusions

In general, weeds are considered to be plants in places 'where they are not wanted' (Thorp and Lynch, 2000). Definitions used to describe plants as weeds (including those associated with import assessment processes) consider plants on the basis of potential costs. Invasions of introduced plant species can result in extensive financial, social and environmental impacts. Estimates of the cost of weeds have been found to

be high in countries such as Australia, the United States, South Africa and New Zealand. There is a substantiated need to restrict new plants because the potential costs can be large and external to those who introduced the plant in the first place, but the benefits of plant introductions must also be considered.

Introduction of new plants to Australia has been crucial to the country's development and future introductions may include plants with the potential to contribute agricultural and environmental benefits, such as reduced salinity. Such plants present the need to reconsider the definition of weeds on the basis of costs alone for the purpose of plant introduction decisions. Any such redefinition would need to include benefits, costs, externalities, unpriced values and incorporate the uncertain and dynamic processes of plant populations to ensure that introduction decisions enhance society's welfare.

The following chapter presents the background for considering the benefits of a plant's introduction when a plant offers the potential to reduce salinity.

3 Plants to reduce dryland salinity in Australia

Dryland salinisation of the Australian landscape is accepted as one of Australia's most significant resource degradation problems (Ewing and Dolling, 2003; Kingwell *et al.*, 2003; Pannell, 2001; Smitt *et al.*, 2003). The costs of salinity are significant and shared by a range of stakeholders, though to date, costs to agricultural land holders have been the most widely studied. New perennial pastures have been identified which may reduce the incidence of dryland salinity (Dear and Ewing, 2008; Ewing and Dolling, 2003).

In order to consider the potential benefits of a new plant introduction, it is necessary to understand the nature of the problem the plant is intended to mitigate and the way in which the new plant can assist. This chapter reviews the problem of salinity and studies that have addressed salinity management. Issues pertinent to the analysis of salinity are then presented. Like invasive plants, salinisation is a dynamic and uncertain process which can result in externalities and thus a role for government.

3.1 The biophysical nature of dryland salinity

Salinisation is the accumulation of salts via the actions of water in the sub-soil to a level that causes degradation of the soil (MDBC, 2002). Primary salinity results from the natural process of salt being concentrated in the soil through evaporation and transpiration by plants (Colmer *et al.*, 2004). Secondary salinity results from irrigation or in response to changes in ground water balance, primarily from human action (NLWRA, 2001). Dryland salinity is the specific term used to describe secondary salinity that results from changes in groundwater balance, and is the subject of this review[6].

[6] Secondary salinity can also result from irrigation. The nature and management of irrigation salinity varies considerably from that of dryland salinity.

3.1.1 The process of dryland salinisation

Underlying the Australian landscape are series of local, intermediate and regional groundwater tables (MDBC, 2002). At equilibrium, the water entering a groundwater table, *recharge*, is equivalent to water leaving it, *discharge*. When there is excess recharge, groundwater tables rise in order to restore the water balance throughout the system. In the process of restoring balance, salts stored beneath the surface may be mobilised to the surface or to root zones of plants (SCSI, 2004) resulting in dryland salinity. Soil salinisation is highly correlated with the depth of groundwater because rising ground water is the vehicle for mobilising salt.

The primary[7,8] cause of excess recharge entering groundwater tables is replacement of perennial native vegetation with farming and grazing systems that enable a larger proportion of rain to enter the groundwater system (Greiner and Cacho, 2001; Hajkowicz and Young, 2002; MDBC, 2002; Pannell, 2001; Stirzaker *et al.*, 2000). When compared to annual crops and pastures which have replaced it in such a large part of the Australian landscape, perennial vegetation has a greater water use capacity and as such avoids the tendency for water to drain beyond the root zone when soils approach saturation (MDBC, 2002). This capacity stems from the native perennial vegetation being deep rooted, sourcing water throughout the year and having growth periods which coincide with the region's rainfall.

The process of salinisation is illustrated in Figure 3.1 which is adapted from Hertzler and Barton (1992). In panel 2, following the reduction in the extent of perennial plants as shown in panel 1, there is increased recharge of the groundwater system and it rises to maintain balance throughout the system. The result of this is shown in panel 3, where the entire groundwater table rises and in some areas this puts it sufficiently close to the surface that there will be effects of the salts mobilised in the process. Dryland salinisation is generally accepted as the worst impact of the

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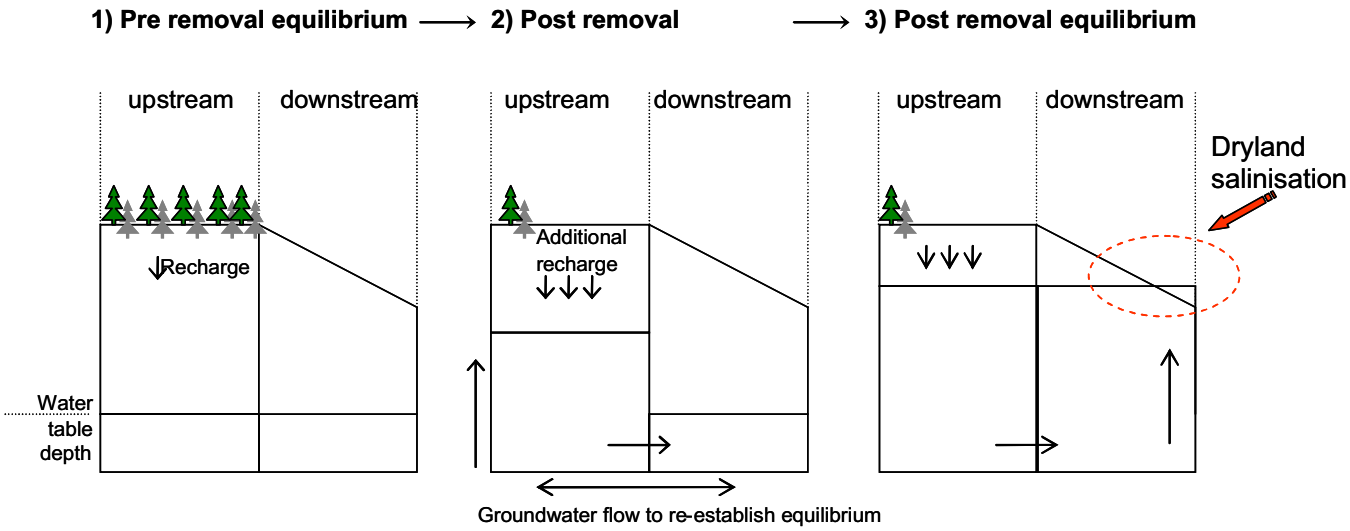
[7] The historical and natural occurrence of salinity, including dryland salinity, is widely accepted. CSIRO estimates there to be just under 30 million hectares of naturally saline land in Australia (SCSI 2004).

[8] Natural Resource Intelligence (NRI) have disputed the conventional rising groundwater explanation of the salinisation process, and as an alternative propose that 'salinity is generally associated with a decline in soil structure that is largely caused by a decline in soil organic matter' (in SCSI (2004)). In this model, tree clearing exacerbates, rather than causes, salinity: salinity is caused by soil degradation 'primarily due to unsustainable land use practices rather than land clearing per se'.

removal of deep-rooted native perennials. Land degradation, soil erosion and soil acidification, also result from this change to natural landscapes (Hodgson and Hatton, 2003). A further combined consequence of these impacts may be the facilitation of weed invasions.



Source: Adapted from Hertzler and Barton (1992)

Figure 3.1 Illustration of the process of dryland salinisation

The groundwater systems underlying the Australian landscape extend laterally from less than around five kilometres in the case of local groundwater systems, to in excess of 50 kilometres in the case of regional systems. The expanse of groundwater systems means discharge can occur in the vicinity of the original recharge or many kilometres from the recharge area depending on the type of underlying groundwater system. As such dryland salinisation can be an example of an externality whereby activities in upstream parts of catchments have hydrological effects on the well-being of farmers and communities downstream (Greiner and Cacho, 2001; MDBC, 2002). The NLWRA (2001) describe the local, intermediate and regional systems as being associated with recharge response times of 30-50, 50-100 and over 100 years, respectively. Similarly, because the movement of ground waters relies on the shape and nature of the aquifers and underlying rock structure over what may be up to hundreds of kilometres, there is likely to be a significant delay between the time that recharge and corresponding discharge occur. The complex nature of groundwater systems means that the location, timing and the extent of discharge will be uncertain, even if the size of the groundwater system is known.

3.1.2 The impacts of salinisation

Salt in soil has the effect of making water less available to plants. Even in wet soil, a plant will experience stress similar to that in a drought when salt levels rise. When salt gets into a plant that is not salt tolerant, it causes reduced growth rates, progressive leaf fall, and eventually, death (Stirzaker *et al.*, 2000). The main impacts of dryland salinity are manifest as crop and pasture yield loss, biodiversity and habitat damage and saline water run off (Hajkowicz and Young, 2002; Hodgson and Hatton, 2003). These impacts will begin when water tables approach, generally, two metres from the surface (Heaney *et al.*, 2001; MDBC, 2002).

The most obvious effect of salinity in landscape terms is loss of vegetation, a condition which compounds the problem of dryland salinity because as more vegetation is lost, more recharge occurs (NLWRA, 2001). However, the impacts of dryland salinisation are in a continuum rather than discrete 'step' impacts. For example, in soils either mildly affected by salinity or in the early stages of salinisation, there may be few outward signs of plant stress. Decline in pasture or crop productivity, stunted growth and absence of sensitive species such as white clover are key indicators of mildly saline areas (MDBC, 2002). The Murray Darling Basin Commission (2002) identify progressive indicators of dryland salinity and groundwater discharge as:

- loss of plant vigour;
- invasion of pastures by salt tolerant species;
- salt fluorescence;
- development of bare soil; and
- groundwater seepage from the soil surface, or in erosion gullies.

Salinisation does not only render soils unsuitable for conventional agriculture and native ecosystems and changes the plant species composition, but also has the potential to directly erode infrastructure such as roads and buildings, and contribute to increased stream salinisation. The impact of salinisation on fauna can be directly through loss of plants that form the diet of a particular species, or indirectly through the salinisation of the aquatic environments of animals. These losses flow on to the

species that feed on these species and so on. A less obvious effect of salinity is the loss of soil biodiversity[9] (Heaney *et al.*, 2001; NLWRA, 2001).

3.2 The scale of the problem of dryland salinity

3.2.1 The extent of dryland salinisation

Three key studies provide the most comprehensive estimates of the areas of Australia impacted by salinity. These have been undertaken by the Australian Bureau of Statistics, the Prime Minister's Science, Engineering and Innovation Council and for the Australian Dryland Salinity Assessment.

The Australian Bureau of Statistics (ABS), using a survey approach, estimated that 2.0 million hectares of agricultural land on 20,000 farms had signs of salinity (irrigation or dryland) (ABS, 2002). An estimated 800,000 hectares of the 2 million were found to be no longer able to sustain agricultural production. Dryland salinity was found to be the dominant form of salinity with less than 10 percent of the reported areas of salinity associated with irrigation. Western Australia had the highest incidence of salinity: over 1.2 million hectares, on 7,000 farms, showed signs of dryland salinity. An earlier study by the Prime Minister's Science, Engineering and Innovation Council (PMSEIC, 1999) also found Western Australia had the highest incidence of salinity and both studies report low incidence of dryland salinity in Queensland, Tasmania and the Northern Territory. The PMSEIC study found a total of 2.5 million hectares to be affected by salinity as compared to 2.0 million estimated by ABS. This difference might largely be explained by the PMSEIC inclusion of non-agricultural land.

The Australian Dryland Salinity Assessment (NLWRA, 2001) did not assess areas currently impacted by, or showing signs of, salinity, but rather assessed areas at risk of dryland salinity. Using information on water tables, a total of 4.7 million hectares of agricultural land and 0.6 million hectares of remnant and planted perennial

[9] Soil biodiversity refers to the range of micro-organisms which inhabit soils and recycle organic debris such as dead plant and animal matter (in the process releasing nutrients into the soil so plants can reuse them) and break down chemicals and pollutants that enter the soil.

vegetation was assessed to be at *risk* of dryland salinity. The assessment also estimated areas likely to be at risk by 2020 and 2050 on the same basis. Expectations are for 6.4 million hectares of agricultural land and 0.8 million hectares of remnant and planted perennial vegetation to be impacted by 2020 and likely to rise to 13.7 million and 2.0 million hectares respectively by 2050. Kingwell (2003) used Assessment findings to estimate that 2.6 million hectares of the total agricultural land affected falls within cropping zones of Australia and is likely to increase to 3.7 million hectares by 2020. The Assessment also identified streams and lakes (11,800km), rail (1,600km) and road (19,900km) assets, towns (68) and wetlands (80) at risk of dryland salinity. The key difference between estimates provided by the Assessment and previous national estimates, as well as most state estimates, is that it quantifies areas at risk of salinisation, not salinity affected areas. This difference is often not recognised in the literature.

More recently there has been a re-evaluation of the extent to which salinity poses a threat in Australia. The complete re-evaluation is yet to be completed but the prevailing view is that the threat of dryland salinity is less extensive in some areas than previous assessments had found. The view is now that dryland salinity is an issue for some 2 million hectares across Australia. Unchanged however is that the greatest incidence is in Western Australia with an estimated 1 million hectares affected (George, 2008).

Salinisation is emerging as a major resource degradation problem around the world (Bathgate and Pannell, 2000). Iran, South Africa, India, Egypt, China and Pakistan are among the countries most affected by salinisation (Ghassemi *et al.*, 1995). In these countries, salinisation is however primarily associated with irrigation. Dryland salinity affects the northern Great Plains of the United States, including the states of Montana, North Dakota and South Dakota, and the plains of Canada, including the provinces of Alberta, Saskatchewan and Manitoba. Dryland salinity has also been identified in Thailand and is expected to emerge in central Asian countries such as Kazakhstan (Ghassemi *et al.*, 1995). It appears however that while Australia is not alone when facing the problem of salinity, few countries share its high incidence of dryland salinity as compared to salinity from irrigation.

3.2.2 Costs of salinisation

> *The current and predicted impact of salinity on infrastructure, water quality, productive land, bio-diversity, remnant vegetation and conservation reserves is significant. The costs imposed on landholders, governments and residents of rural towns are considerable (SCSI, 2004).*

Pannell (2001) provides a summary of the costs of dryland salinity, by type: a) direct losses and b) replacement, repairs and maintenance. Examples of a) direct losses provided include:

- *Agricultural* - reduced agricultural production; reduced flexibility of farm management; and
- *Non- agricultural* - extinctions, loss of biodiversity, loss of amenity, loss of aesthetic values, loss of water resources, eutrophication of water ways, loss of development opportunities on flood plains.

Examples of b) replacement, repairs and maintenance provided include:

- *Agricultural* - repairs to buildings, replacement of dams, establishment of deep drains to lower saline groundwater; and
- *Non-agricultural* - repairs to houses and other buildings, desalinisation of water resources, repairs to infrastructure, restoration of natural environments.

Pannell (2001) also identifies the costs of dryland salinity to the community. Dryland salinity is identified as one more factor contributing to the decline in farm numbers and farm incomes, which flows on to reduced rural populations, reduced services and reduced rural wellbeing. Impacts on mental health are also identified by Pannell as a social impact associated with dryland salinity.

An alternative way to consider costs is from a management perspective. Pannell and Lefroy (2000) divide costs of salinity into four types on the basis of control prospects:

1) Costs of damage to towns, roads, rivers, and native vegetation that **cannot** be controlled by farm treatments.
2) Costs of damage to towns, roads, rivers, and native vegetation that **can** be controlled by farm treatments.
3) Damage to farms where there is little chance of recovery.
4) Damage to farms that is preventable.

The Murray Darling Basin Commission (MDBC) as part of the National Dryland Salinity Program (MDBC, 2002) identify costs resulting from saline water supplies and shallow saline water tables which are borne by households, businesses, local and state governments, public utilities and farming operations. The MDBC group these costs into the following categories:

1) Repair and maintenance costs.
2) Costs from the reduced lifespan of infrastructure.
3) Costs from preventative action.
4) Increased operating costs.
5) Increased construction costs.
6) Foregone income.
7) Environmental and heritage costs.

While the categorisations of Pannell (2001), Lefroy & Pannell (2000) and the MDBC (MDBC, 2002) vary in their approach, common to each is recognition of costs to a range of stakeholders, both on and off site, and consideration of opportunity and direct costs.

A range of authors have endeavoured to quantify these costs. Hajkowicz and Young (2002) cite estimates from the National Dryland Salinity Program that place the annual cost of lost agricultural production in the order of $130 million, infrastructure damage in the order of $40 million and the value of lost environmental assets at $40 million. Warnick (2003) estimates the value of lost agricultural production to be $200 million per annum and while not quantifying them, recognises that further costs result from reduced water quality, ecological health of streams and terrestrial biodiversity, soil erosion, increased flood risk, damage to infrastructure and fixtures and social

impacts. Inclusion of both the cost of lost agricultural production and environmental damage attributable to salinity increases estimates of the annual cost of salinity to at least $1.0 billion according to Healy (2001).

The CRC for Plant-Based Management of Dryland Salinity estimate that the annual national costs of dryland salinity could be as high as $3.5 billion with inclusion of the cost to rural communities of declining population, loss of business (both existing and potential), the cost of rural restructure when farms become unprofitable, and increased health problems due to stress on families affected by change, as well as lost production and costs of control (CRC PBMDS, 2002). Common to most estimates of national loss has been the inclusion of the costs of lost production on farm and the conclusion that they contribute the largest part of total losses.

3.3 Salinity management options

As the financial, environmental and social costs of dryland salinity have become increasingly apparent, substantial, primarily public, investment has been made to better understand the causes of salinity and develop options to minimise the costs. A number of options have proven effective for particular niches, but in general there is consensus that a suite of complementary options is required (Hodgson and Hatton, 2003; Stirzaker *et al.*, 2000) especially when competing financial, environmental (SCC, 2001) and social outcomes must be considered.

Salinity management can be approached in terms of prevention, remediation or adaptation (Pannell, 2001; SCSI, 2004). Each approach aims to reduce the cost of salinity, the first through the avoidance of salinisation of soil in the first place, the second through reversing the salinised state of an area, and the third by altering land use to make the best use of it in its salinised state. In an agricultural context, the assurance of financial returns relative to existing land uses is crucial to the adoption and ultimate success of any management approach. Recommendations of the House of Representatives Standing Committee on Science and Innovation Salinity Inquiry

included greater emphasis on Commonwealth investment in the development of new land and water use systems, because a sufficient number of economically viable management options to address salinity are not yet available (SCSI, 2004). The inquiry concluded that there is a need for salinity management options that are sufficiently attractive financially so that they will be adopted irrespective of the benefits to be gained in terms of salinity management.

Prevention of excess recharge entering the water table requires strategies focused on returning deep-rooted perennials to land management systems or reducing recharge within existing systems. These strategies include the use of existing annual crops and perennial plants. Engineering options, though often too costly at the required scale (Cacho *et al.*, 2001), may also be relevant.

3.3.1 Perennial plants

The aim of introducing perennial plant options for salinity management is to profitably mimic the recharge associated with native perennial vegetation. Typically 5-15 percent of the long-term average rainfall passes the roots of annual plants whereas less than one percent passes beyond the roots of native perennial vegetation (Stirzaker *et al.*, 2000). In terms of rainfall, this translates to between 15 and 300 millimetres per annum of leakage past the root zone of annual crops while the capacity of the landscape to drain groundwater to rivers is in the order of only 0.5-10 millimetres per annum, an amount comparable to that which drains beyond native perennials (Stirzaker *et al.*, 2000). Stirzaker *et al*. (2000) identify a range of perennial plant options for the prevention of dryland salinisation. Table 3.1 is based on these options and includes comments from Stirzaker *et al.* and Kingwell (2003).

Table 3.1 Perennial plant options

Perennial plant options	Description	Comment
New agricultural plants	Breeding or selection of long season, perennial and/or deep rooted perennial cultivars of current crop and fodder plants to fit new farming systems.	Likely to be long delays in development and use of biotechnology [a].
High rainfall tree species	Adoption of tree crops in high rainfall areas.	Limited areas for adoption [c].
		Likely to significantly diminish surface water yield [a].
Low rainfall tree species	Adoption of tree crops such as oils, pharmaceuticals, bush foods etc, in low rainfall areas.	Potentially most effective, but least viable due to slow growth of trees in low rainfall areas and new market development is required [a].
Agroforestry	Planting of commercial woodlots.	Effectiveness depends on the proportion of the area planted to trees and choice of where trees are planted [c].
		Uneconomic in low rainfall areas due to slow growth rates [c].
		A variety of technical and economic issues affect widespread incorporation in grain production systems as a salinity management option [b].
Perennial pastures	Grasses or legumes which leak significantly less beneath the root zone than annuals, and provide a viable fodder alternative (e.g. Lucerne). Alternatives likely to be developed from new native or exotic species, rather than currently used native or exotic cultivars.	Limited effectiveness in higher winter dominant rainfall, acid and shallow soils and under grazing pressures [a].

[a] Stirzaker *et al.* (2000), [b] Kingwell (2003), [c] review and consultation from this study.

Strategic and large scale planting of trees, the ultimate deep-rooted perennial, has been long advocated as an effective means of lowering water tables. In excess of 770,000 hectares of trees have been planted for salinity management across Australia (ABS, 2002), yet a number of difficulties have been identified with this as a means of reducing recharge:

- Survey evidence suggests that the benefits in some areas are localised to an area not more than 10-30 metres outside of a planted area (van Buren and Pannell, 1999). Similarly, biophysical modelling studies indicate that extensive areas can be necessary in a catchment to significantly affect the rate of spread of salinised land, reduce productivity losses and reduce salt loads in rivers.
- The returns to tree planting generally do not rank favourably with annual crops (Cacho *et al.*, 2001; Heaney *et al.*, 2001). The lag between the planting costs and harvest benefits, and the large scale required are major impediments (MDBC, 2002).
- Re-forestation of headwaters can result in downstream water quality issues because the catchment's surface water yield may decline, leading to concentration of stream salinity (Heaney *et al.*, 2001; MDBC, 2002).
- It is likely that despite tree planting water tables will still continue to rise, though to an equilibrium lower than they would have been in the absence of tree planting (van Buren and Pannell, 1999).

Given these difficulties, alternative perennial pasture plant options appear superior and so have been investigated. The introduction of perennial pasture plants with the potential to stem recharge in salinity areas is an alternative diversification option being assessed from both an agricultural and an environmental perspective. The search for plant options suitable to Australian farming systems and landscapes has extended throughout Australia *and* internationally because suitable native Australian perennial grass pastures are primarily suited to high rainfall areas and often fail to significantly reduce recharge (Bennett *et al.*, 2002).

Birdsfoot trefoil (*Lotus corniculatus)* is just one of a wide range of exotic grasses and legumes with the potential to stem recharge in Australian farming systems which have been identified for evaluation and potential introduction. Table 3.2 provides some examples of the perennial plants that are currently being investigated.

Table 3.2 Plants identified with the potential to reduce recharge in Australian farming systems

Species	Environmental Niche	Notable Characteristics	Region of Origin
Dorycnium hirsutum (hairy canary clover)	Dryland conditions on soils with low fertility	Good persistence and summer production, drought and insect tolerant, good tap root.	Mediterranean Basin
Trifolium hybridum (alsike clover)	High rainfall areas of western Victoria	Waterlogging tolerance, some drought tolerance. Aerial seeding. Tolerance to alkalinity and acidity.	Eurasia
Lotus corniculatus (birds foot trefoil)	650-1000 mm high rainfall zones[a].	High genetic diversity, water logging tolerant.	Eurasia
Hedysarum coronarium (sulla)	Medium to fine textured calcareous soils	Genetically diverse, good drought tolerance with branching tap root, vigorous autumn production. Contains condensed tannins. Non-bloating, insect resistant, aerial seeding, non-shattering.	Mediterranean Basin
Onobrychis viccifolia (sainfoin)	Similar niche to lucerne, well drained medium to fine textured soils in low rainfall areas.	Good insect control, non-bloating (condensed tannins). Erect habit, green manuring option, non shattering, aerial seeding.	Mediterranean Basin
Lotus glaber (narrow leaf trefoil)	Winter-waterlogged, black and grey clay soils of Murray Darling Basin.	Reasonable flooding tolerance	Eurasia

Source: Adapted from Bennett *et al.* (2002). [a] Ayres *et al.* (2006b)

In many cases, the attraction of the identified exotic species is that they offer reductions in recharge similar to lucerne. In many areas of Australia lucerne (*Medicago sativa L.*) provides an ideal pasture option which is viable financially and assists in the reduction of groundwater recharge. However, in many other areas lucerne alone is not a suitable perennial plant option for reducing salinity. Lucerne is generally prone to water-logging and is intolerant of acidic and high aluminium soils. The widespread use of lucerne would, irrespective of soil types, present the risk of diseases and pests arising from such a monoculture. Alternative plants require development so benefits similar to those provided by lucerne, in terms of on-farm profitability and reduced recharge, can be achieved in areas where lucerne cannot, or has not, been grown.

The potential benefits of the new pasture species with regard to reducing recharge are largely accepted (Cocks, 2003; Ewing and Dolling, 2003; Gintzburger and Houerou, 2003). However some concerns remain as listed below.

- Perennial pastures must potentially be established on a large scale to achieve recharge reductions. Some estimate this might need to be 70-80 percent of an area to make a difference to landscape salinisation (George *et al.*, 1999).
- The importance of any new plant's profitability is recognised as key to adoption on the required scale (Ewing and Dolling, 2003).
- Nitrogen-fixing leguminous perennials may add to already high levels of nitrogen in water tables (Gintzburger and Houerou, 2003). This is referred to as 'nitrogen pollution'.
- The potential application of new exotics may be limited in many cases to rainfall areas with no water-logging, low acid and low aluminium soils (Cocks, 2003), such that a number of species would be required to be developed from exotic accessions to identify plants for the range of soil-climate-farming system niches.
- In many salinity provinces the adoption of perennial vegetation will not sufficiently control dryland salinity within reasonable timeframes: there is too much additional recharge (MDBC, 2002). The time it takes for groundwater to pass out of the landscape and for the system to recover hydraulically to previous levels is substantial and beyond the feasible planning timeframes of most landholders.

One of the greatest concerns, however, is the potential for the new species such as these to become weedy either in the landscape to which they are introduced or other areas, including natural environments (Bennett and Virtue, 2005; Emms *et al.*, 2004; Semple *et al.*, 2004).

Management of salinity using perennial vegetation presents the opportunity for benefits additional to those from reduced salinity. Pannell (2001) for example, identified additional, or complementary benefits, from increased perennial vegetation; including carbon sequestration, biodiversity maintenance, reductions in wind and water erosion, farm income diversification and the potential to manage herbicide

resistance. Pannell also identified social benefits related to regional development, aesthetics and amenity values.

The eventual success of deep-rooted perennials in reducing the extent of dryland salinity will depend primarily on the farm-level economic performance of available production systems. This will be influenced by short-term production issues, dynamic factors, sustainability factors, risk factors and whole-farm factors (Bathgate and Pannell, 2000).

3.3.2 Alternative management strategies

Alternative salinity management strategies include options other than perennial plants to reduce the emergence of salinisation, remediation of saline lands to their non-saline state and adaptation to the saline state. The distinction between prevention and remediation of the impacts of salinity and remediation and adaptation of saline sites is not clear cut, probably a result of the widely accepted difficulty facing remediation of saline sites. Reversal of salinisation is even more difficult than prevention (Pannell, 2001) and actions with this aim have had limited impact (Goss, 2008).

Due to difficulties with prevention and remediation, adaptation may in some cases be the preferable option (Pannell, 2001). Depending on the degree of salinisation, adaptation using alternative plant species might include salt-tolerant species such as saltbush, balansa clover or tall wheat grass (Pannell, 2001). The CRC PBMDS is developing more salt-tolerant plant species, as well as the perennial plants they are developing to manage recharge. These include salt-tolerant barley, using transgenic methods to introduce the salt tolerance of a species such as sea barley grass (*Hordeum marinum*) to traditional wheat varieties (Colmer *et al.*, 2004), and breeding to develop the salt tolerance of pastures such as *Trifolium michelanium* and *Trigonella balansae* (CRC PBMDS, undated). Further alternative adaptation options might include salt water aquaculture, electricity generation, irrigation with brackish water, algae production and salt/mineral mining (MDBC, 2002; Pannell, 2001). A further adaptation option is to completely stop use of a saline land area and intensify, or optimise, production on non-saline sites (NLWRA, 2001; Pannell, 2001).

3.4 Issues to consider with respect to dryland salinity management

The problem of dryland salinity has a number of characteristics that influence the way its management and management options should be considered. Dryland salinity results from processes which are highly dynamic, can impose external impacts and may be uncertain. These characteristics are discussed separately below.

3.4.1 Dynamic processes

Groundwater systems are often characterised by processes with considerable time lags (Smitt *et al.*, 2003; Stirzaker *et al.*, 2000) and causalities (Cacho *et al.*, 2001; Greiner, 1996). These features mean that actions to manage dryland salinity in one period will lead to changes in the state of the watertable in subsequent periods. Choices in land-use will result in changes to the state of the watertable and may lead to feedback effects on the performance of those land-use choices. These causal relationships may occur over many years.

Failure to incorporate these features in an evaluation of management options would likely undervalue the watertable as a resource and as such undervalue the benefits of management. Any approaches to management need to consider the lengthy time periods for the realisation of salinity and do so within a framework which incorporates its dynamic processes.

3.4.2 External impacts

The groundwater systems that underlie Australian landscapes are often not confined within the boundaries of a single landholder and as such can impose external impacts. Similarly, the costs of salinity may be borne by another landholder many years into the future due to the considerable time lags associated with groundwater systems. As such there are inter-temporal external impacts as well as spatial external impacts.

Consequently, management of salinity often requires action by one landholder for the betterment of another. Evaluation of options for the management of dryland salinity

very often then requires the incorporation of external impacts. Failure to do so may: a) under or over estimate the costs of implementation; and b) result in poor policy design because the interests of all relevant parties have not been accounted for.

The importance of external impacts of the management of salinity will depend on the level (e.g. farm, catchment, regional) at which it is considered and the approach to management that is being considered (e.g. biological, engineering). The failure to account for external impacts of management that are not manifest as changes in off-site salinity, for example where a plant option might become invasive, is likely to over estimate the benefits of the management option.

3.4.3 Uncertainty

Where, when and to what degree dryland salinity will become evident will generally be uncertain. Variance in the reaction of groundwater systems results from the wide variance in features, such as soil and rock type and composition, rainfall in different locations, size and elevation of landscapes and rock layers beneath the surface, between groundwater systems.

Given the uncertainty of the systems, the effectiveness of management options will also be uncertain. This might be further exacerbated when management includes a plant and there is uncertainty in relation to how the plant will respond in a new environment. For example, to what extent will the plant actually reduce recharge in a new area given the soil and climate of that particular area?

When the review of management options is not considered in the context of uncertainty, their benefits may be overstated. Any evaluation must necessarily therefore include consideration of the range of expected values of the benefits.

3.5 Summary

As is the case with any natural resource decision, it is necessary to consider all of the potential options for addressing a problem prior to selecting a particular approach.

With respect to salinity, this requires that a decision to introduce a new perennial pasture species be considered in the context of other options to achieve the same reductions in recharge and therefore the same reduction in salinisation. This would include whether engineering technologies, for example, might be the best option to achieve the reductions, or whether the development of new varieties of native species might be a better approach (Wallace, 2006).

This chapter has provided an overview of the options for salinity management and in doing so has summarised the findings from extensive consideration, largely managed by the CRC for Plant-Based Management of Dryland Salinity, of the range of options for managing salinity. From this it has been found that, in the case of managing recharge in agricultural systems, there is a role for the introduction of new pasture plants because engineering options are often inappropriate and largely too expensive, tree options are limited in their viability, adaptation options are constrained by technology or markets or are inappropriate in many areas and the adaptation of native species is limited by their characteristics.

However support for the role of new perennial plant species with respect to management of salinity is prefaced by concern that the plants may become invasive. The potential introduction of these plants presents a conflict for conventional approaches to assessing new plant introductions so that an alternative approach, where the introduction is assessed on the basis of its contribution to society's welfare, is considered appropriate.

The processes that lead to dryland salinity are dynamic and uncertain and likely to result in external impacts. It is evident that dryland salinisation is a problem for which Australia requires solutions. However assessment of solutions must be mindful that ill-defined management options, analysis and policy approaches may render the natural resource base in a position worse than 'business as usual' (Pannell, 2001). Any analysis to assess the benefits of a management option for salinity must therefore accommodate the dynamic processes, externalities and consider the range of expected outcomes.

4 A conceptual approach to the problem

In this chapter an economic framework is developed to help resolve whether a plant should be introduced when there are potential costs and potential benefits. In doing so, what constitutes a weed is defined on the basis of whether a plant makes positive contributions to society's welfare.

In particular, the approach is developed to accommodate the problem of whether to introduce a new plant which has the potential to reduce recharge that leads to dryland salinity. This requires that the two problems be considered in a single framework, because the potential solution to one brings with it the potential for the other.

This chapter presents the conceptual basis for estimation of the costs and benefits of an introduced plant solution to dryland salinity. Based on the reviews of Chapters 2 and 3, this necessarily includes incorporation of externalities and uncertainty and an approach to economic modelling which accommodates the dynamic process of each.

4.1 The economic framework

The considerations required for an economic approach to the problem of new plants with the potential for both costs and benefits are now presented in turn. These are then brought together in the formal presentation of the decision criteria to be applied to the introduction decision.

4.2.1 Potential benefits

The range of *potential* benefits that may be associated with a plant's introduction to reduce recharge associated with salinisation could include:

- reduced existing area of salinised agricultural and natural land;
- avoided future salinisation of agricultural and natural land;
- additional pasture options;

- avoided monoculture disease and/or pest concerns that might result from limited salinity management pasture options; and
- species specific features (e.g. some flowering perennials may offer apiarists benefits, or a species may offer native fauna additional habitat).

Figure 4.1 illustrates the reduced incidence of salinity resulting from the introduction of a perennial pasture plant using a further adaptation of Hertzler and Barton's (1992) diagram of dryland salinisation which was presented in Chapter 3.

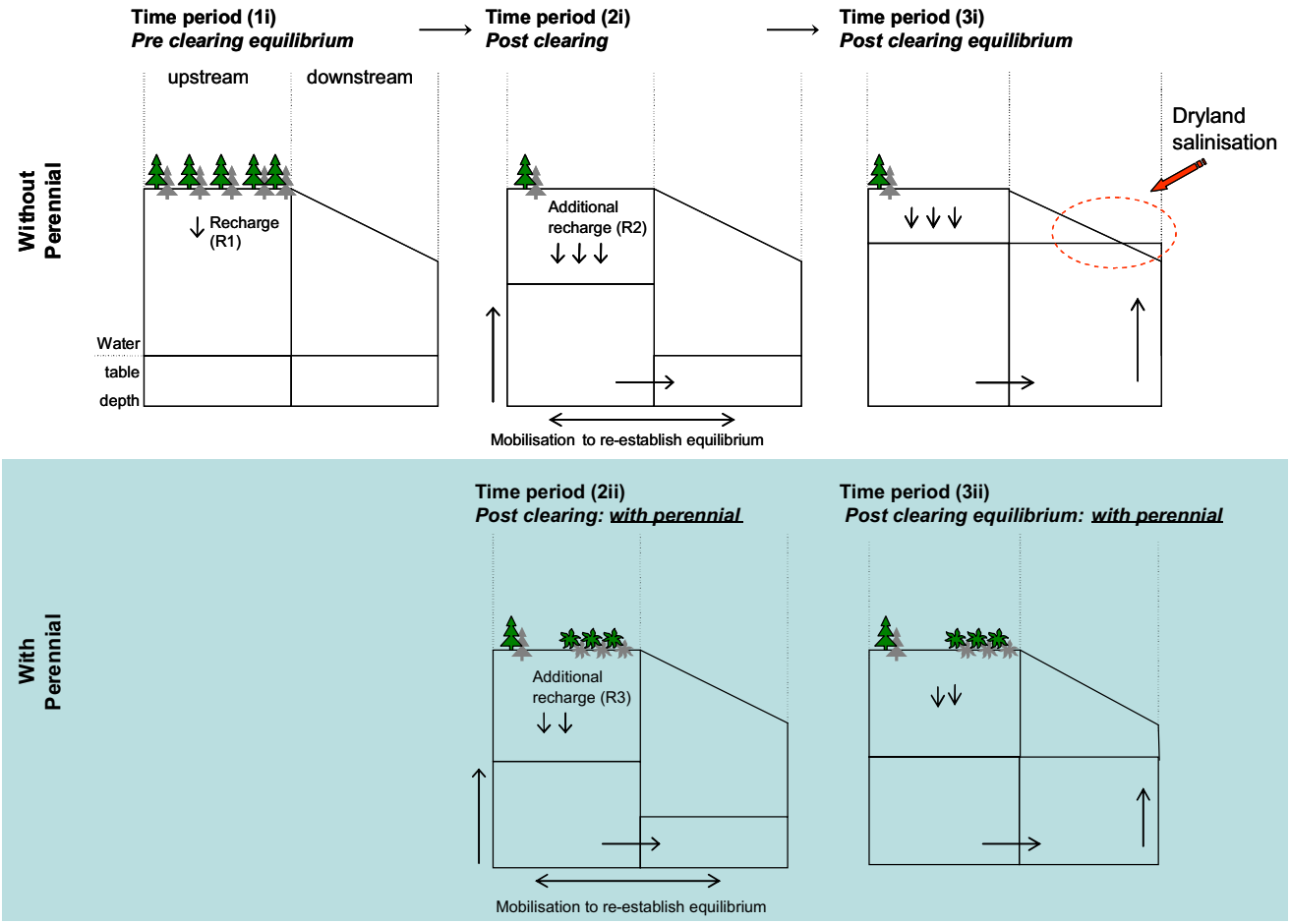


Figure 4.1 Impact of perennial pasture species by reducing recharge
(in three time periods 1, 2 & 3, without (i) and with (ii) the introduced pasture plant)

Subfigures 1i - 3i of Figure 4.1 show salinisation as a result of the removal of deep rooted perennials. Recharge following the clearing of deep rooted perennials is R2. Recharge R2 exceeds R1, the recharge prior to clearing. Subfigures 2ii - 3ii illustrate

the impact of the planting of deep rooted perennial pastures in previously cleared areas. Recharge following the planting of the perennial pastures is R3. The aim of the planting of new perennial pastures is for R3 to be less than R2, so that at worst, future salinisation is slowed or, at best, is halted and existing salinisation reversed. Achieving reductions in salinity is the fundamental reason for the introduction of these plants so the focus in this study is on these benefits.

The impact of reduced salinity can be illustrated within an economic surplus framework as shown in Figure 4.2. The figure illustrates the benefits to just one sector, agriculture.

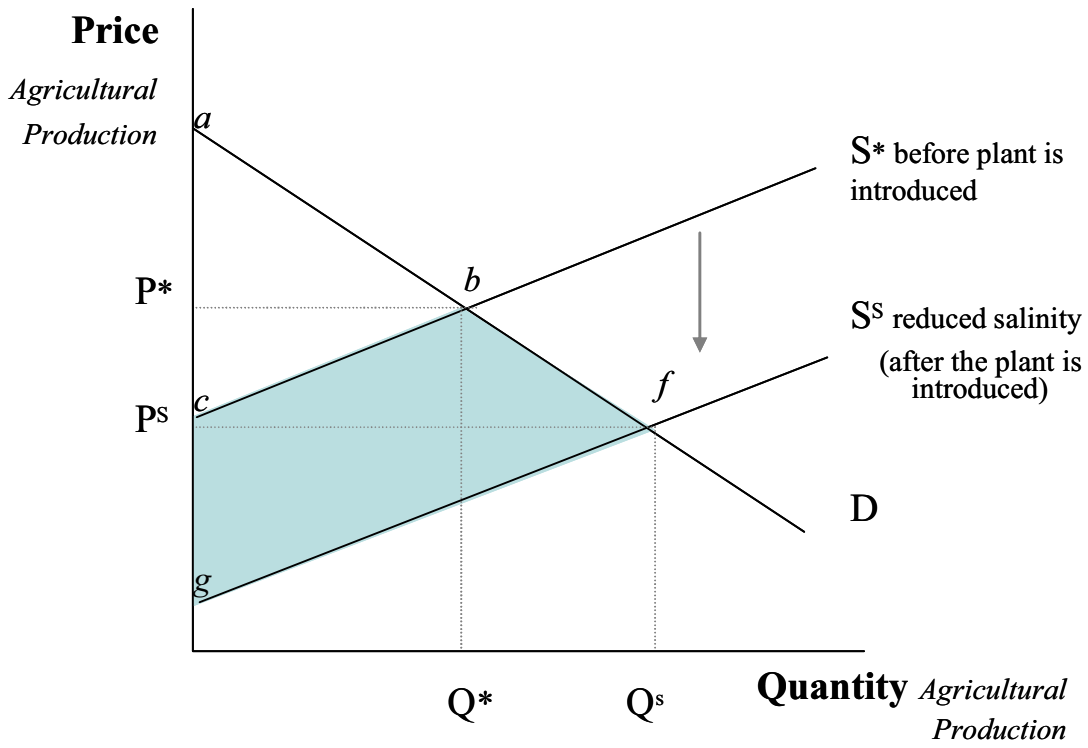


Figure 4.2 Potential benefits of an introduced perennial pasture species, through reduced salinity

In the production of Q* quantity of agricultural production at price P*, the total amount of economic surplus captured is represented by the area of the triangle ***abc***. This economic surplus is shared between consumers (the sum of additional value they derive from a unit of agricultural production over and above the price paid for the unit) and producers (the sum of the value earned from the sale of the unit, in excess of

its cost of supply = profit). It is desirable for an economy to maximise the economic surplus available to its producers and consumers (i.e., maximise the area ***abc***).

Prior to a plant's introduction, economic surplus is represented by the area ***abc***. Where the plant is introduced and salinity is avoided or reduced, the supply of agricultural production, S*, shifts to S^s. The economic surplus captured by society is now represented by the area ***afg***, an increase in economic surplus of ***cbfg*** (the shaded area). This increase results from the corresponding shift in quantity to Q^s and fall in price to P^s.

4.2.3 Potential costs

The range of *potential* costs that may be associated with a plant's introduction in the context of its establishment and invasiveness could include:

- costs of establishment / propagation;
- loss of agricultural production;
- loss and/or degradation of biodiversity and natural landscapes;
- loss of recreational amenity;
- costs associated with adverse health impacts (e.g. hayfever);
- costs of control; and
- loss of genetic diversity (including of rhizobia where leguminous species require importation of rhizobia).

Figure 4.3 illustrates within the economic surplus framework (as used in Figure 4.2), the impact of the costs of a plant's introduction when a plant becomes invasive. The figure illustrates the benefits to just one sector, agriculture.

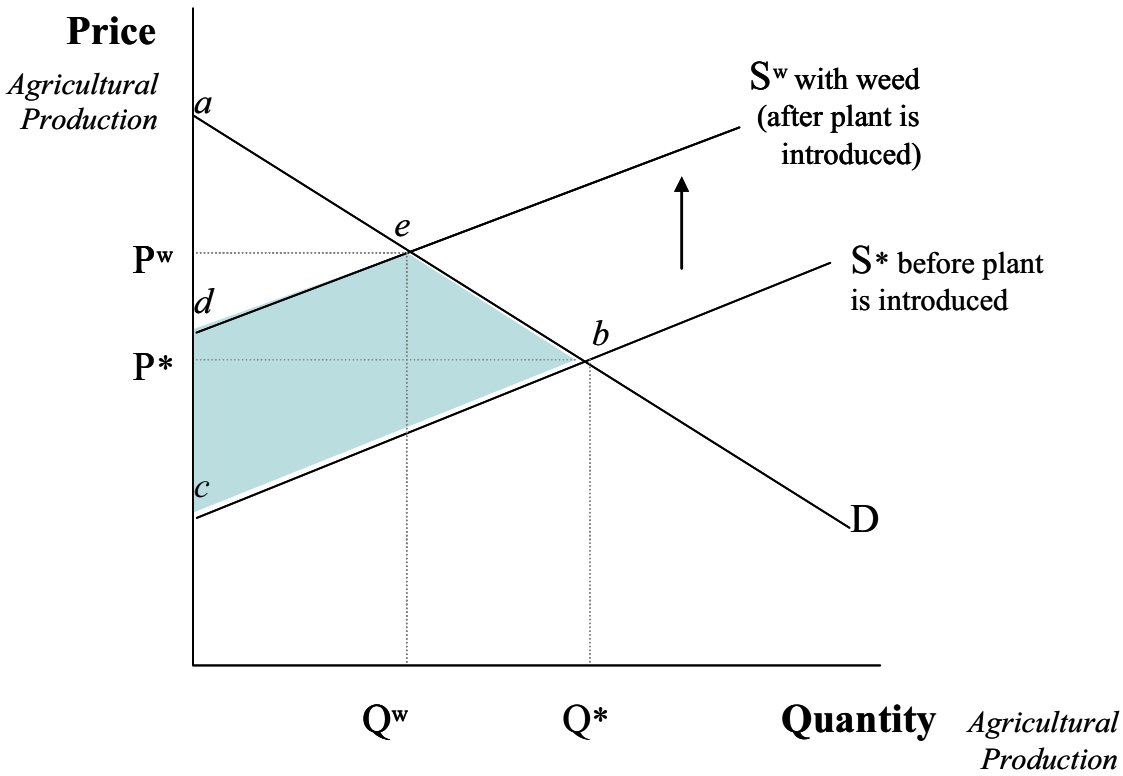


Figure 4.3 Potential costs of an introduced perennial pasture species, through invasiveness

Prior to a plant's introduction, economic surplus is represented by the area ***abc***, which corresponds to the price P* and the Quantity Q*. Where the plant is introduced and becomes invasive in agricultural areas, S* shifts to S^w. The economic surplus captured by the community is now represented by the area ***aed***, a decline in economic surplus represented by the area ***cdeb*** (the shaded area). This decrease results from the corresponding shift in quantity to Q^w and increase in price to P^w.

4.2.3 Costs and benefits

From an economic perspective, actions and policies are considered favourable where there is a resultant increase in the total well-being of society (Sinden and Thampapillai, 1995). That is, where there is an increase in the net benefits from production and consumption of goods and services to give an increase in economic surplus.

The impacts described in 4.3.1 and 4.3.2 must be considered together to determine if there is a net increase or net decrease in economic surplus following a plant's introduction. Figure 4.4 illustrates concurrent consideration of the costs and benefits in an economic surplus framework.

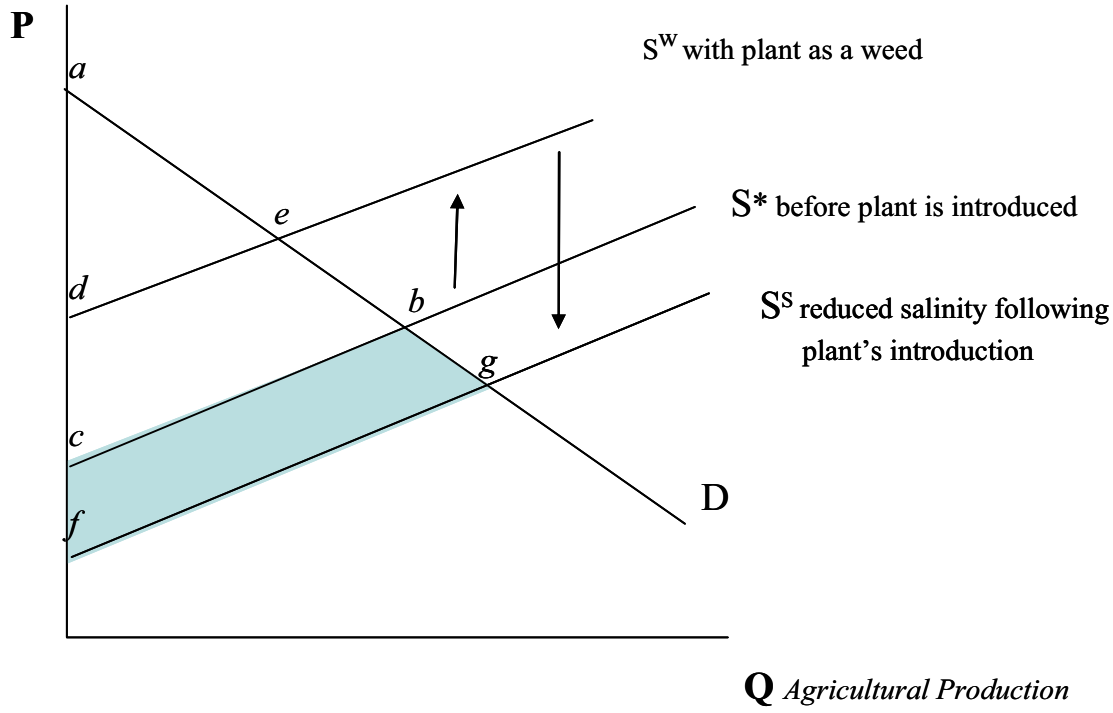


Figure 4.4 Costs and benefits of a new plant introduction - *the case of net benefits*

In this figure, the case of net benefits, or an increase in economic surplus, is illustrated. Prior to the plant's introduction, economic surplus is represented by the area ***abc***. The impact of the plant's invasiveness is to shift S* to S^W, with a corresponding reduction in economic surplus by the shaded area ***debc***, to ***aed***. The impact of reduced salinity is to shift S^W to S^S, an increase in economic surplus from ***aed*** to ***agf***. The net impact of the two shifts is an increase in economic surplus from the original ***abc*** to ***agf*** - an net increase of ***cbgf***.

The surplus framework has been used to examine a wide range of impacts in agriculture. Pioneering applications include that of Griliches (1958) and Akino and Hayami (1975) who respectively examined the impact of research and development on the maize and rice industries. Lindner and Jarratt (1978) used the economic

surplus approach to examine research benefits in agriculture and specifically the impact when alternative supply-shift assumptions are adopted within an analysis.

Alston (1991), also in an examination of the research benefits, provided the following equations used to assess changes in economic surplus (ΔES), where changes in consumer surplus (ΔCS) and producer surplus (ΔPS) are calculated using absolute values for the elasticities of demand (η) and supply (ε) and *K* is an expression of the vertical shift in the supply function as a percentage of the initial equilibrium price (P* in Figures 4.2 and 4.3). P_1 and Q_1 are price and quantity of production respectively. *Z* is an expression for the percentage reduction in price arising from the supply shift.

$$\Delta CS = P_1 Q_1 Z(1 + \frac{1}{2} Z\eta) \quad \textbf{(4.1)}$$

$$\Delta PS = P_1 Q_1 (K - Z)(1 + \frac{1}{2} Z\eta) \quad \textbf{(4.2)}$$

$$Z = \frac{K\varepsilon}{\varepsilon + \eta} \quad \textbf{(4.3)}$$

$$\Delta ES = \Delta CS + \Delta PS = P_1 Q_1 (1 + 1/2 Z\eta) \quad \textbf{(4.4)}$$

The approach shown formally by Alston (1991) has been used widely with respect to the impact of invasive plants on Australian agriculture. Vere *et al.* (1997), for instance, used this framework to consider the impact of weeds on agricultural production systems together with the benefit of research outcomes to reduce the on-farm and industry costs of weeds. The costs and benefits of specific invasive plants has also been addressed using this framework, including for serrated tussock (Jones and Vere, 1998) and vulpia (Vere *et al.*, 2003). Similarly this framework has been used to assess the impact of weeds to an industry, winter cropping (Jones *et al.*, 2000), as well as to all agricultural industries (Sinden *et al.*, 2004).

The examples cited indicate the wide use of the surplus approach both in agriculture generally and specifically with respect to invasive plants. Common to all of the examples is the assessment of a single impact on agriculture. The decision to introduce a plant to reduce recharge requires concurrent consideration of two distinct impacts to determine the net impact on economic surplus. As such the potential introduction of plants to reduce the recharge associated with salinisation presents the opportunity to apply the surplus framework to the problem of two distinct but related natural resource problems.

4.2.4 Externalities

Externalities are costs or benefits borne by parties other than those who incurred the corresponding costs or benefits of an action, or as described by Mullen (2001), the result when off-site effects are not confined to those who cause them. The framework so far (Figures 4.2 through 4.4) illustrates the potential marginal costs and benefits in the context of one agent: the agricultural industry. In the case of a plant introduced to reduce recharge associated with salinity, costs and benefits are likely to be disparate over time and space such that a framework needs to recognise external impacts.

The inclusion of externalities in consideration of natural resource management is necessitated by the divergence in the interests of individual parties and the community. Without the inclusion of external costs in the analysis, resource use by an individual is generally of a rate considered exploitative by the community (Mullen, 2001). Externalities need to be incorporated within the framework otherwise a decision may not be not be made on the basis of contributions to society's welfare.

Externalities with respect to salinity have been widely considered. To date this has been limited to costs and benefits manifest as off-site changes in salinisation. This represents an external cost of the action of clearing when the two zones are not owned by the same stakeholder. Similarly, the cost of introducing a deep-rooted perennial pasture may be borne in one location, while the benefit of reduced salinisation may be captured in another location. This is an external benefit if the two zones are not owned by the same stakeholder. This type of externality has been considered before

(for example, Greiner, 1996; Pannell *et al.*, 2001) and its relevance is likely to vary. Therefore and for the purpose of the analysis for this study, this type of externality is not included: welfare changes are measured at the subcatchment level as a whole rather than considering individual landholders within the subcatchment. The distribution of benefits and costs, however, is a factor which may influence relevant potential policy responses to this problem.

Instead the focus of this study is on another external impact because the plants used to manage salinity may become invasive. That is, there is the potential that costs associated with a plant's invasiveness will not be borne by those capturing the benefits of reduced salinisation. The external cost of the plant's invasiveness may be borne:

- by other farmers (with other crops or livestock, in other subcatchments, states, or zones);
- in natural areas (other catchments, in other states, other zones); and/or
- other industries and the community (recreational, health and social amenity as well as by non-agricultural industries).

Figure 4.5 provides a simplified representation of the way the external impact of plant invasiveness might be considered when a plant is introduced to a subcatchment. While assessment of both the benefits and costs within the initial area of plant introduction, the subcatchment, is necessary, the costs and benefits in external areas must also be assessed in order to consider the impact of the plant introduction on society's welfare. In Figure 4.5, for example, one external area of a catchment which surrounds a subcatchment might be other agricultural production areas ($n=1$) and another might be natural environments such as national parks ($n=2$). In reality n may be as many as there are discrete units such as landholders, specific land use types or specific ecological communities etc.

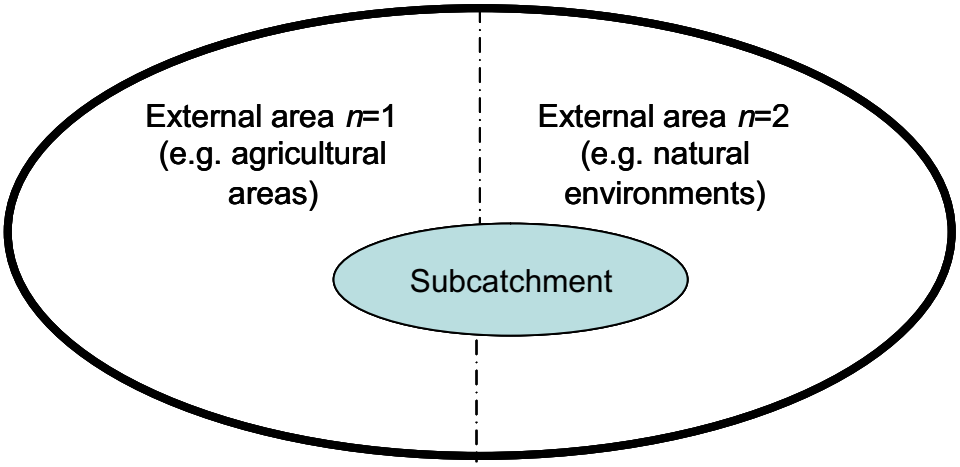


Figure 4.5 External costs - *consideration of 'n' external areas*

Due to the lengthy dynamic process associated with salinisation and its management, as well as the potentially lengthy processes associated with plant invasions, costs and benefits may be separated over time as well as spatially. This might especially be the case where so-called sleeper weeds are involved. Such temporal externalities must be considered in the determination of a plant's impact on economic welfare. Costs and benefits associated with salinisation and plant invasiveness may be borne now, but corresponding benefits and costs only realised by future generations. Similarly, the costs and benefits of actions in the past may only be realised by current generations. To accommodate such inter-temporal externalities, economic surplus estimation must not only incorporate n external areas, but also be undertaken over a relevant time period T.

The physical existence or absence of externalities is not a prescription for whether or not a plant should be introduced because it is necessary to consider the impact on social welfare. Rather, externalities should be included within the framework so that an optimal level of economic welfare is identified and, where relevant, welfare losses compensated or policies introduced which limit welfare losses. There may be cases where the optimal economic level of an externality is zero, just as there may be cases in which the optimal level is high. An optimal economic level of an externality is where the marginal net private benefit (MNPB) associated with the externality equals the marginal external cost (MEC) of the externality (Randall, 1987). As such, the optimal level will depend on the nature of the relevant MNPB and MEC curves. This

is illustrated in Figure 4.6, for the case of a plant introduced to reduce the recharge associated with salinity where the optimal level, for illustrative purposes, is shown as the number of hectares of pasture planted.

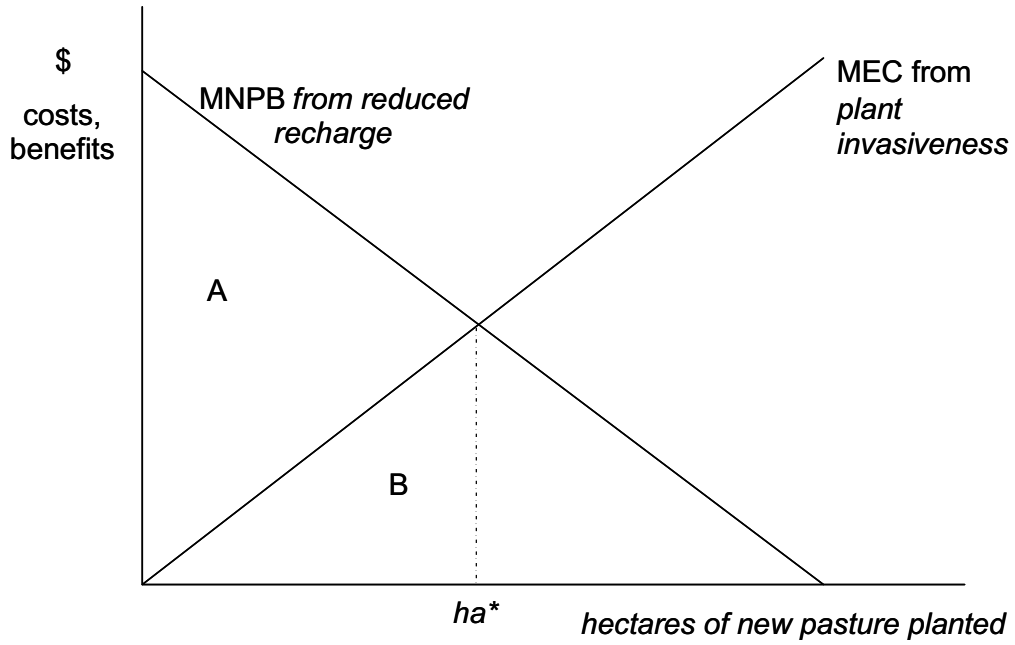


Figure 4.6 Optimal level of externality

The intersection of the MNPB and MEC curves identifies the number of hectares of the plant which are optimal from an economic perspective. The number, *ha**, is associated with an optimal amount of economic damage from the externality, the area B. As such, B can be estimated as the corresponding loss of surplus as illustrated in Figure 4.3 (***cdeb***) when Q^w is the quantity of output associated with *ha**.

The gain from introducing the plant is the area A. As such, A can be estimated as the corresponding gain in surplus as illustrated in Figure 4.2 (***cbfg***) when Q^s is the quantity of output associated with *ha**. Considering the problem in this way sets the basis for examining policies that might accompany a plant introduction.

4.2.5 Uncertainty

Neither the costs nor the benefits associated with the introduction of a plant to reduce recharge will be known with certainty. The necessity of including uncertainty and estimation of the expected values of the benefits and costs of deliberate plant introductions has been recognised (Caley *et al.*, 2006; Kalisch Gordon, 2004). The uncertainty of potential costs results from the many factors which influence the dynamic nature of plant populations and is exacerbated when plants are exotic so there is limited local history on which to discern their likely adaptation to the Australian environment. The uncertainty of benefits results from the many and varied factors which influence the hydrology of groundwater systems and the ability of plants to reduce recharge, as well as limited knowledge of the nature and form of most groundwater systems.

The uncertainty inherent in a decision to introduce a new plant to manage recharge and thus reduce salinity can be illustrated in a decision framework from Hardaker *et al.* (2004), as in Figure 4.7. In the absence of a plant being introduced, the costs of salinity (C_0) are incurred. If the plant is introduced, benefits (B_1 and B_2) and/or costs (C_1 and C_2) result. The benefits (B_1 and B_2 depending on the pathway) illustrated in this decision framework correspond to the magnitude of the surplus area ***cbfg*** in Figure 4.2. The costs (C_1 and C_2 depending on the pathway) correspond to the magnitude of the surplus area ***cdeb*** in Figure 4.3.

As illustrated in the decision framework, once a decision to introduce is made, the net impact on economic welfare is dependent not only on the costs or benefits but also on the probability that the plant spreads outside of managed pasture areas (p_{w1}, p_{w2}) and the probability that the plant reduces recharge (p_{r1}, p_{r2}). In the diagram, the payoffs, V_1-V_5 represent the net present values associated with a limited number of pathways. In practice there will be a payoff V (equal to $B_i - C_j$) associated with each possible pathway (i,j).

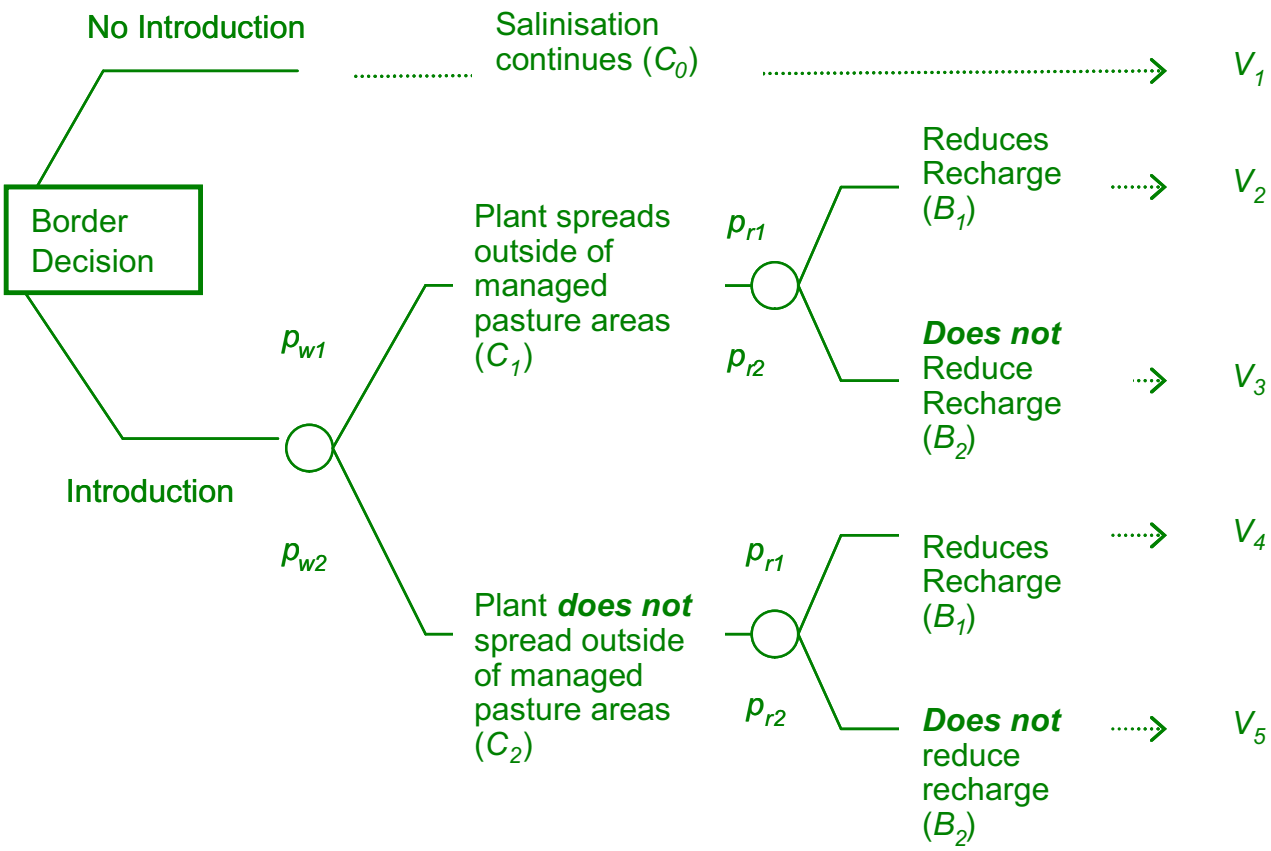


Figure 4.7 A plant introduction decision under uncertainty

4.2.6 Decision criteria

Given the discussion in sections 4.2.1 - 4.2.5, the definition of what constitutes a weed should be a plant for which…

"…the expected costs associated with the plant's introduction, including expected negative externalities EXCEED the expected benefits from the plant including expected positive externalities."

In this context, entry should be denied.

The following equation summarises the relationships expressed in this decision criterion. A plant is not a weed if:

$$\sum_{t=1}^{T}\sum_{d=1}^{D}\sum_{e=1}^{E}[(B_{d,t}\cdot p_{r_d} - C_{e,t}\cdot p_{w_e})(1+r)^{-t}] > 0 \qquad \textbf{(4.5)}$$

where:

d = designated area, where the plant could be introduced to D total areas

e = external area, where there are E total external areas

t = year, where there are T total years

$B_{d,t}$ = Benefit of the plant's introduction in time t for area d

$C_{e,t}$ = Cost of the plant's introduction in time t for area e

p_{r_d} = probability plant achieves intended aim in area d

p_{w_e} = probability that costs are realised in area e

r = discount rate

This decision criterion, consistent with benefit-cost analysis, sanctions an introduction where the net benefit (the net change in economic surplus, including positive and negative externalities) is positive.

A Pareto-efficient move is one where some persons are made better off but no-one is made worse off (Randall, 1987). The criterion detailed above will deliver an economically efficient outcome, but one which is unlikely to be a Pareto-efficient move. A decision to introduce a new plant will not only be based on the decision criterion described in Equation 4.5, but also on the relevant policies which accommodate the positive and negative externalities which may make others worse off.

Identification of externalities and, where possible, their optimal levels provides the basis for their amelioration. Mullen (2001) however notes that while consideration, and where possible, removal, of externalities is necessary for efficient resource use, equity must also be considered to avoid allocations being viewed by future generations as unfair. Similarly, for a government to even be involved in implementing policy which circumvents externalities, the efficiency gains from intervention need to be comparable to those gains possible from other uses and an appropriate means for implementing the policy must be available (Mullen, 2001).

4.2 Application of the framework

The conceptual framework detailed so far has been designed to accommodate the features of the salinity and invasive plant problems. In particular it incorporates costs, benefits, uncertainty and externalities. Also identified is the necessity for analysis to include the dynamics of each problem. Bioeconomic approaches have been applied to the dynamic analysis of both salinity and invasive plant management.

Bioeconomic modelling can provide a balance between consistent integration of biophysical and socioeconomic dimensions (Holden, 2004). Most simply defined, a bioeconomic model is one which consists of two parts: one which describes the production system; and one which relates the production system to market prices and constraints (Cacho, 1997).

In this context, applications of bioeconomic models have been many in areas such as the management of aquaculture (Cacho, 1997), natural resources (Holden, 2004) and agricultural production (Bywater and Cacho, 1994; Oriade and Dillion, 1997). The inter-temporal nature of many agricultural, natural resource and fishery problems necessitates the development of dynamic bioeconomic models. Hardaker (2004) defines "a dynamic decision problem as one in which a sequence of decisions must be made through time, these decisions typically being interleaved with the outcomes of uncertain events that impinge on later choices, as well as the consequences of

decisions taken earlier." This description is consistent with the nature of plant invasions and salinity processes.

This section reviews the application of bioeconomic approaches to salinity and invasive plant management with particular reference to those features identified in Chapters 2 and 3 as relevant to the topic of this study: uncertainty, externalities and unpriced values.

4.2.1 Bioeconomic applications to the problem of plant invasions

Economic analysis has been used, or at least recognised as useful, when choosing appropriate strategies to manage invasive plants (Cacho *et al.*, 2006; de Wit *et al.*, 2001; Higgins *et al.*, 1997a; Higgins *et al.*, 1997b; Odom, 2002; Paynter *et al.*, 2003; van Wilgen *et al.*, 2004; van Wilgen *et al.*, 2002; Wainger and King, 2001). These approaches can also be used to consider potential plant introductions. This section provides an overview of bioeconomic applications to the problem of invasive plants, followed by specific examination of the treatment of uncertainty and the valuation of unpriced values with respect to invasive plants in the literature.

Odom (2002) integrated an economic model with a biophysical simulation model of Scotch broom (*Cytisus scoparius*) spread in the Barrington Tops National Park in New South Wales. The economic component of the model described the costs and benefits associated with Scotch broom control in the National Park, while the biophysical simulation model described the population dynamics of Scotch broom using a simplified representation of the plant population's age structure. The resultant bioeconomic model was a deterministic dynamic programming model designed to determine the level and type of control of Scotch broom which maximises welfare to society. Odom (2002) identified characteristics for locations in the Park where control activity should be concentrated (i.e. where it will be most effective), that eradication is not an optimal strategy for the National Park and also that a combination of control measures is optimal due to the seed bank and budget constraint being key model drivers.Odom’s work was innovative with respect to invasive plant management in natural ecosystems because it considered the impact of a budget

constraint on optimal strategies. Uncertainty associated with a number of key factors was also introduced to the model in the form of probabilities. For instance, seedling survival, site disturbance and suitability for colonisation were represented as probabilities.

Cacho (2004) adopted a bioeconomic approach to assist in the formation of invasive plant management decision rules. In particular, Cacho developed the concept of 'switching points'. The switching points are: a) the point at which eradication is no longer optimal and containment becomes the optimal strategy; and b) the point where the optimal strategy changes from containment to 'do nothing'. Using the Scotch broom example for base parameters, Cacho identified some key decision rules pertaining to the switching points.

Bioeconomic approaches have been applied to consider plant invasions in natural ecosystems around the world. The impact of plant invasions on mountain fynbos ecosystems in South Africa has been examined using a simulation model approach (Higgins *et al.*, 1997b) and more generally, the approach has been considered as a means of informing decisions related to invasive plants, biological control options and native plants (Higgins *et al.*, 1997a).

While a number of bioeconomic studies have focused on invasive plant management within natural ecosystems (Higgins *et al.*, 1997a; Higgins *et al.*, 1997b; Odom, 2002) the majority has focused on management within agricultural systems. The RIM (Resistance and Integrated Management) model, for example is a bioeconomic model that simulates the dynamics of weed populations (Pannell, 2000) for the purpose of developing on-farm weed management strategies. A range of variations on the basic model has been developed including a multi-species RIM model which incorporates the coexistence of a number of species on farm (Monjardino *et al.*, 2003).

Bioeconomic modelling has also been used to evaluate the impact of weed research and development investment on farm and across agricultural industries. Vere *et al.* (2003) evaluated the impact of the CRC for Weed Management Systems investment into improved management of *Vulpia* subspecies. The Grazing Systems Simulation Model (GSM) was adopted for the biophysical component of the model and provided

estimated supply shifts. The GSM can include up to six ecological-functional species groups including introduced and native perennial grasses, legumes, annual grasses and broadleaf plants. The estimated supply shifts from GSM were incorporated within the economic component of their model, an economic surplus model. To accommodate the regional specificity of agricultural technologies in the surplus model, a temperate pasture zone and a rest of Australia zone were incorporated. To assess the relative merit of the CRC investment, surplus estimates were considered within a benefit-cost framework using Monte Carlo simulation to incorporate stochasticity associated with seasonal variation, losses per production unit and technology adoption. The Vere *et al.* analysis found substantial benefits attributable to the CRC's investment, captured primarily by producers in the temperate pasture zone.

4.2.1.1 Uncertainty in invasive species management assessment

Pest and weed management decisions are characterised by uncertainty. In this context the uncertainty is most commonly defined as exposure to *unfavourable* consequences. In an economic framework, however, it is necessary to consider the uncertainty of both favourable and unfavourable events. It is therefore desirable to consider the probabilities of an occurrence within an economic framework. These terms imply no preconceived value judgement regarding an impact's favourable or unfavourable nature. They form just one part of the standard definition of *risk = likelihood (or probability)* x *consequence* (Caley *et al.*, 2006), where consequences of interest just happen to generally be negative.

Odom *et al.* (2003b) provide a review of studies that measure the impact of weeds and other pests on the natural environment. In this review of 20 studies[10], only eleven had explicitly included uncertainty with respect to the plant's lifecycle, including five which did so only descriptively. The extent to which costs and benefits associated with plant populations are realised will however also depend on the way the plant spreads. Only 10 of the 20 studies reviewed by Odom, *et al.* include spread functions. These ranged from explicit population models to assumptions of constant rates of

[10] A total of 62 studies were reviewed. Twenty studies were selected as useful contributions to the economic literature.

spread, arbitrary functions and spread calculated on the basis of historical data. Two of the studies that included both quantification of uncertainty and spread functions, provide interesting approaches to the inclusion of uncertainty.

The first avoids the need to estimate probabilities. Jetter *et al.* (2000) assess the net benefits of proposed eradication of a 1997 invasion of imported red fire ants (*Solenopsis invicta* Buren) in California. The geographical spread of fire ants is uncertain and the success of eradication efforts is also uncertain. To quantify the rate of spread, the authors considered previous infestations of the pest, such as in Texas, and consulted scientists. From this, alternative spread functions for three scenarios were adopted for a 10-year period. The three scenarios were low (10% of total), medium (25%), and high (40%) infestation of total susceptible areas. The authors, however, recognised the difficulties of estimating the likelihood of success of eradication efforts and approached the problem in terms of thresholds of potential eradication success. The study provides estimates of the probability of eradication success required for the expected net benefits of eradication to be positive. These breakeven probabilities were found to be surprisingly low, ranging from a 1.72 percent chance of eradication success required in the low impact scenario to only 0.67 percent in the high impact scenario to warrant an eradication effort.

The second study of interest is an assessment of the introduction of a biological control agent, a rust introduced from Australia, to control Orange Wattle (*Acacia saligna* (Labill) H.) in an area of the Western Cape of South Africa. Orange Wattle, also known as Port Jackson Willow, is an Australian native whose invasion is stimulated by fire. The assessment by Higgins *et al.* (1997a) includes multiple probability parameters. Probabilities associated with fire frequency, seed decay and germination and plant mortality following fire, plant mortality following infection by the rust and mortality of an alternative plant following a fire incident are all incorporated within a population model as mean values, with analysis undertaken over a range of values around these means. This research indicated that the benefits of biological control can outweigh the costs of its introduction.

A study of the management of Karnal bunt (a disease caused by the fungus *Tilletia indica* Mitra affecting wheat, rye and triticale) in the United States, provides a further

example of the incorporation of uncertainty within an analysis of plant population management. It illustrated that management decisions can vary considerably when probabilities are incorporated directly within benefit-cost analyses rather than independently. Glauber and Narrod (2003) did not focus on the probabilities of an individual event, but rather assessed the overall probability of a pathway to an incursion occurring. The probability of an outbreak of Karnal bunt, was expressed as:

$$p^* = 1 - (1 - p_1)(1 - p_2)(1 - p_2)(1 - p_3).....(1 - p_n) \qquad \textbf{(4.6)}$$

where p_i (i= 1…n) represents the probability of each of a range of n events contributing to the overall pathway probability. For example, p_1 may represent the probability that Karnal bunt infected seed is transferred into an unaffected area, p_2 may represent the probability of infected harvesting machinery being used, and so on. Monte Carlo analysis with 10,000 iterations for each management option was undertaken across 30 identified uncertain events. The probabilities of the parameters included a range of triangular, beta and lognormal distributions and incorporated data from previous USDA risk assessments. This multiplicative approach allows joint consideration of all the factors that individually drive the probability of an event discretely, and recognises how management options may alter the probabilities of individual influences differently to the overall event.

4.2.1.2 Valuation of environmental costs

Invasive species, including plants, are considered by many as second only to land clearing activities in terms of threats to biodiversity (Pimentel, 2002). The challenge of including the lost environmental values that result from a plant invasion is demonstrated by the extent to which studies are dominated by cost estimates for agricultural industries and the limited inclusion of environmental values in national cost assessments (see Tables 2.2 and 2.3).

Odom *et al.* (2003a) broached the issue of assessing the value of natural areas when assessing the financial costs of controlling Scotch broom (*Cytisus scoparius*) in

Barrington Tops National Park of New South Wales. They identified $2,308 per species as a minimum value, derived from Queensland government expenditure of $200 million to preserve native vegetation including 26 species per $1 million of expenditure, and $233,220 per species as a maximum, derived from Lockwood and Carberry's contingent valuation of the value of endangered species per household (1998). From this range Odom *et al*. selected $100,000 per species as a midway value to represent the worth of biodiversity protected by weed control efforts. More recent work, using cost data from the WONS study (Thorp and Lynch, 2000), support values in this range with Sinden and Griffith (2007) estimating the annual value of protection of an individual threatened species as $64,830.

Hester *et al* (2006) have reviewed studies on the relationship between the invading plants and the loss of ecosystem services in natural environments. They concluded that perfect knowledge about the relationship between invading plants and loss of services and valuation of the loss of services is as yet unobtainable and few studies have analysed these relationships. As a response, Hester *et al* developed a simulation model which examines the range of relationships that might be feasible. The functional relationship between plant invasions and the loss of ecosystem services is shown in the following equation.

$$S = \frac{qN}{1 + \left(\left(\frac{q}{m}\right) - 1\right)N} \qquad 0 < q \leq 10 \qquad \textbf{(4.7)}$$

In this rectangular hyperbola function, S, is the quantity of ecosystem service output per hectare, q and m are parameters and N is the weed density (invasive plants per hectare). The parameter q describes the damage relationship where values of $q>1$ represent immediate, rapid reductions in the output of services, while values of $q<1$ represent initially smaller reductions in outputs followed by greater marginal falls as the maximum invasion is approached. When $q = 1.0$, the marginal decline in the output of services is constant.

Despite difficulties quantifying the value of lost ecosystem services, there is no doubt that invasive plants present a significant threat to many natural environments. As

such, any analysis to consider the potential net benefits of a plant introduction would necessarily require some account for such losses. The recent work by Hester *et al.* (2006), who has progressed a framework for estimating the cost of changes in the output of ecosystem services, and Sinden and Griffith (2007), who have estimated the value of protecting species from invasive plants, are among efforts that are likely to assist in meeting the increasingly apparent need for assessment of costs in natural areas.

4.2.2 Bioeconomic applications to the problem of salinity

The considerable time lags associated with changes in ground water systems (Smitt *et al.*, 2003; Stirzaker *et al.*, 2000), as well as uncertainty regarding impacts and the many possible land use combinations (Cacho *et al.*, 2004), warrants *ex ante* analysis of the likely impacts of management strategies. Bioeconomic modelling can help to identify appropriate management options, and combinations of options, when resources are limited.

A range of models has been developed to assess salinity and salinity management from an *ex ante* perspective. These include both equilibrium (steady state) or dynamic (transient) models. The dynamic approach is more appropriate as it predicts the change in the system in response to changes in practices (Hertzler and Barton, 1992) and dominates current approaches. Hertzler and Barton (1992) identified varying approaches to such modelling, including process-based numerical simulations, water-balance models, and statistically-derived approximations of cause and effect. The appropriate choice of approach depends on the accuracy and detail required, availability of information to build the model and availability of data to calibrate the model; as well as identification of a successful interface between the biophysical and the economic models that is acceptable to hydrologists and economists (Cacho *et al.*, 2004).

A large number of biophysical models have been developed and adopted by various agencies. These include the BC2C (Biophysical Capacity to Change) developed by CSIRO using an earlier model developed by ABARE, namely SALSA (Evans *et al.*, 2004), SOILEC developed by Oram *et al* (1989), Flowtube, Soilflux, Modflow,

FEFLOW, WEC-C, WAVES (Zhang *et al.*, 1997) and a model being developed by the CRC PBMDS (Beverly, 2004). Difficulties arise with the use of models of salinity processes because models can require a large amount of data (Hodgson and Hatton, 2003), relationships between vegetation and the water table are complex (van Buren and Pannell, 1999) and there will be tradeoffs between simulation time and process description (Beverly, 2004).

Further difficulties arise when costs and benefits associated with salinisation and its management are considered. However, in the context of limited resources, or limited funding, an economic approach to identification of management options is appropriate.

An economic approach allows consideration of management options on the basis of effectiveness at the margin. Despite this, only one of the nine commonly used models reviewed by Beverly (2004) incorporates economic assessment of landscape intervention strategies. Graham *et al.* (2004) identify the size of complex hydrology models as the factor preventing their incorporation with extensive economic models. Consequently, where an economic analysis has been attached to a hydrological model, the analysis has tended to be a simplified benefit-cost or net present value approach. Mullen (2001) considers this to be a concern with approaches to land degradation in general.

Economic approaches have been applied at both the farm and catchment or regional level. Many studies have been site specific though a number have been more general or descriptive in their focus. The choice of model type has been dependent on the level of accuracy and detail required, the availability of information to build the model and the data to estimate parameters and calibrate the model (Cacho *et al.*, 2004). A further factor in the choice of model type is the extent to which spillovers, or externalities, are associated with the problem. Table 4.1 provides a selection of economic applications to salinity and its management together with their scope, analysis technique and study aims.

Table 4.1 Salinity and its management - a chronological selection of economic studies

Authors	Scope	Analysis Technique	Study Aims
Salerian (1991)	Two farm	Dynamic optimisation	To explore the effects of land use on salinisation in a Western Australian subcatchment.
Hertzler & Barton (1992)	Two farm	Dynamic optimisation	Analysis of salinity emergence in south-west Western Australia.
Greiner (1994)	Farm	Whole-farm optimisation	Developed MoFEDS model to investigate best farm management strategies for farmers in the Liverpool Plains who face salinisation threat.
Greiner (1998)	Catchment	Dynamic optimal control	Developed **S**patial optimisation **M**odel for **A**nalysing **C**atchment management (SMAC) to allow quantification of externalities and identification of social optimal levels of salinisation and catchment management strategies. Application to Liverpool Plains.
Mueller *et al*. (1999)	Farm	Dynamic optimal control	Studied the cost of switching between land uses and the optimal time for switching using phase farming as a strategy for reducing salinisation.
Bathgate and Pannell (2000)	Farm	Multi-period linear programming	Applies the MIDAS model to the South Coast of Western Australia and incorporates lucerne rotations. No off-site effects or impacts from the spread of salinity are captured.
Cacho *et al.* (2001)	Catchment	Dynamic optimal control	Non-site specific model developed to analyse R&D and subsidisation policies under normative assumptions, with respect to farm forestry.
Greiner & Cacho (2001)	Catchment	Dynamic optimal control	Application of the SMAC model, in the Liverpool Plains, to consider salinisation as a stock externality, its non-point source characteristics and the isolation paradox.
Heaney *et al.* (2001)	Basin	Dynamic simulation with optimisation elements	Hydroeconomic model developed to evaluate the costs and benefits of salinity mitigation options involving land and water changes in 25 catchments of the Murray Darling Basin over 100 years.
Hajkowixz and Young (2002)	Catchment	Simulation	Benefit-cost analysis of six revegetation scenarios in the Lower Eyre Peninsula of South Australia.
Kingwell *et al.* (2003)	Farm	Dynamic Optimisation	Evaluation of perennial plant options in case study areas across Australia's cropping regions - local case studies to consider profit at full equity with and without management, to accompany a review of existing economic assessments of salinity management options.
Abadi and Cooper (2004)	Paddock	Simulation	A case study of alley farming with oil mallees in Western Australia using the *Imagine* framework (a Visual Basic/Microsoft Excel Spreadsheet tool providing partial budgeting economic analysis).

Primarily, the examples shown in Table 4.1 can be classified as optimisation or simulation approaches to *ex ante* evaluation of salinity management. Simulation models allow investigation of what is likely to happen when a set course of action is taken (Mullen, 2001). As such, simulation allows detailed examination of discrete or continuous management options while optimisation identifies the most favourable strategy, but at the expense of model detail in some cases. Mullen has highlighted the generation of shadow prices as a particular advantage of optimisation techniques, especially with respect to land degradation analyses (2001).

Included in Table 4.1 are a number of farm-level model examples. The value of farm level models is widely accepted. They can assess the feasibility and economic viability of proposed strategies (Greiner and Cacho, 2001) which is especially important when final adoption decisions rest with farmers. Farm models can provide useful information on likely adoption, economic incentives and appropriate policies (Bathgate and Pannell, 2000). Further, if spillovers or externalities are not of central concern or significant, aggregation of farm models may be the most appropriate way to consider a catchment problem (Bathgate and Pannell, 2000).

The value of catchment models is also recognised but again there are difficulties associated with the inclusion of hydro-economic components at this level. Cacho *et al.* (2004) identify the process of salinisation itself and human decision making characteristics as the keys to the challenges facing catchment scale modelling. Specific issues for consideration include that (Cacho *et al.*, 2004):

- land use practices are a function of soil and climatic conditions but also prices and farmer preferences;
- land use systems are not homogeneous across catchments;
- salinity management is just another of many considerations in resource allocation decisions;
- treating recharge as a surrogate of salinity is unsatisfactory as the depth of the water table is the actual determinant and as such a link has to be established between the volume of recharge across a catchment and the depth of the watertable in salinity-prone areas;
- threshold effects apply in relation to increasing salinisation and land use performance with later stages of salinisation being irreversible; and

- salinisation is a dynamic process with inherent time lags between cause and effect.

The range of hydro-economic models reviewed in Table 4.1 includes a number of different approaches to the assessment of salinity management options and a key problem inherent to salinity, externalities. There is debate as to the significance of externalities in this context (Bathgate and Pannell, 2000; Greiner and Cacho, 2001; Pannell *et al.*, 2001; van Buren and Pannell, 1999). This debate has however largely been limited to consideration of externalities resulting directly from changes in off-site salinisation. This includes costs and benefits resulting from increases and decreases in salinity from management practices on-site.

Not included in this debate are potential externalities related to management practices that may **not** be manifest in the form of increases or decreases in salinity itself. These of course will vary by management practice and may be costs or benefits as a result and may be as dynamic and as important in the Australia landscape as the impact of salinity itself.

The study examples shown in Table 4.1 include as their focus, the assessment of salinity. There are examples where the problem of salinity has been considered alongside invasive plants in the context of farm management where both have been incorporated as constraints on production. Doole (2007) for example assessed the value of lucerne with respect to recharge reduction and management of the herbicide-resistant crop weed, annual ryegrass. The analysis included a multi-period linear programming model of optimal land-use with respect to the water table coupled with the RIM (Resistance Integration Model) model to undertake bioeconomic simulation of optimal weed management strategies. The analysis provided assessment of optimal strategies for on-farm management of existing weed infestations and the watertable. An earlier but similar application was that of O'Connell (2003), who considered the use of non-commercial trees on recharge and farm profits where invasive plants were a constraint on the whole-farm model. Such analyses have used bioeconomic approaches to assess salinity management strategies subject to the constraint of existing invasive plant populations, and have had a farm or paddock focus.

4.3 Summary

Consideration of plants introduced to limit recharge associated with salinisation of Australian landscapes presents the opportunity to offer an approach to biological introduction decisions which can accommodate both potential benefits and potential costs. Namely, there is the opportunity to consider a plant a weed when…

"…the expected costs associated with the plant's introduction, including expected negative externalities, EXCEED the expected benefits from the plant, including expected positive externalities."

The economic surplus concept within a benefit-cost framework approach to this undertaking offers the chance to simultaneously incorporate two surplus shifts within a single analysis. In order to fully investigate this approach, a modelling technique which allows incorporation of the spatial and temporal elements as well as the inherent uncertainty is required. Bioeconomic modelling has been used widely for such applications.

Bioeconomic approaches to the study of introduced plant management and control have become increasingly sophisticated. Analyses have included considerations such as growth and spread functions, uncertainty, and quantification of impacts, including, in some cases lost environmental values. However few have incorporated all of these considerations quantitatively.

A number of hydrological and hydro-economic models have been developed to design dryland salinity management strategies and assess options. Analyses such as those Hertzler and Barton (1992), Greiner (1994; 1998; 1996; 1997), Cacho and Greiner (2001), Kingwell *et al.*(2003) and Abadi and Cooper (2004) demonstrate the range of approaches taken in hydro-economic modelling. These examples demonstrate the need for analysis of salinity management to incorporate the dynamic processes which may occur over an extended period. The incorporation of externalities, or not, has been a key point of issue with respect to these approaches. Where incorporated, externalities have been limited to changes in off-site salinisation. Non-salinity externalities have not been considered in research published to date.

5 The Model

5.1 Overview of the model

The method and model presented in this chapter are based on a rationalised version of the decision criterion developed in the previous chapter. Namely, accept a new plant for introduction if:

$$\sum_{i=1}^{I}\sum_{j=1}^{J}\sum_{t=1}^{T}[(B_{i_t})p_{ri} - (C_{Aj_t} + C_{Nj_t})p_{wj})(1+r)^{-t} > 0 \qquad \textbf{(5.1)}$$

where:

i = state of nature of salinity control (recharge reduction), i = [effective, somewhat effective,…ineffective].

j = state of nature of plant invasion, j = [no invasion, some invasion….severe invasion].

B_i = benefit of reduced recharge in the subcatchment where plant is introduced

C_{Aj} = cost of invasiveness in agricultural areas of the surrounding catchment

C_{Nj} = cost of invasiveness in natural areas of the surrounding catchment

p_{ri} = probability of salinity outcome i

p_{wj} = probability of invasion outcome j

t = period where there are T total periods

r = the discount rate

The conceptual approach to the problem developed in Chapter 4 establishes the foundation for considering a large number of possible scenarios over a large number of affected areas. Equation 5.1 is a simplified version that considers one area where benefits are captured, a subcatchment, and two aggregated external areas: agricultural areas (A) of the surrounding catchment and natural areas (N) in the surrounding catchment.

Figure 5.1 illustrates how the decision criterion represented in Equation 5.1 is implemented in this study. The approach comprises two sub-models: A Dynamic Programming (DP) sub-model shown on the left; and a plant population simulation sub-model shown on the right. The DP sub-model is used to estimate the present value of benefits in a subcatchment. The benefits in the subcatchment (B) are estimated from the value of farm profits from the optimal enterprise mix in the subcatchment with the introduced plant, $(V^{p}{}_{t}(w_t))$, and without the plant $(V^{np}{}_{t}(w_t))$. The plant population sub-model is based on a Leslie matrix and is used to estimate the costs of invasiveness in the catchment for both agricultural areas (C_A) and natural areas (C_N).

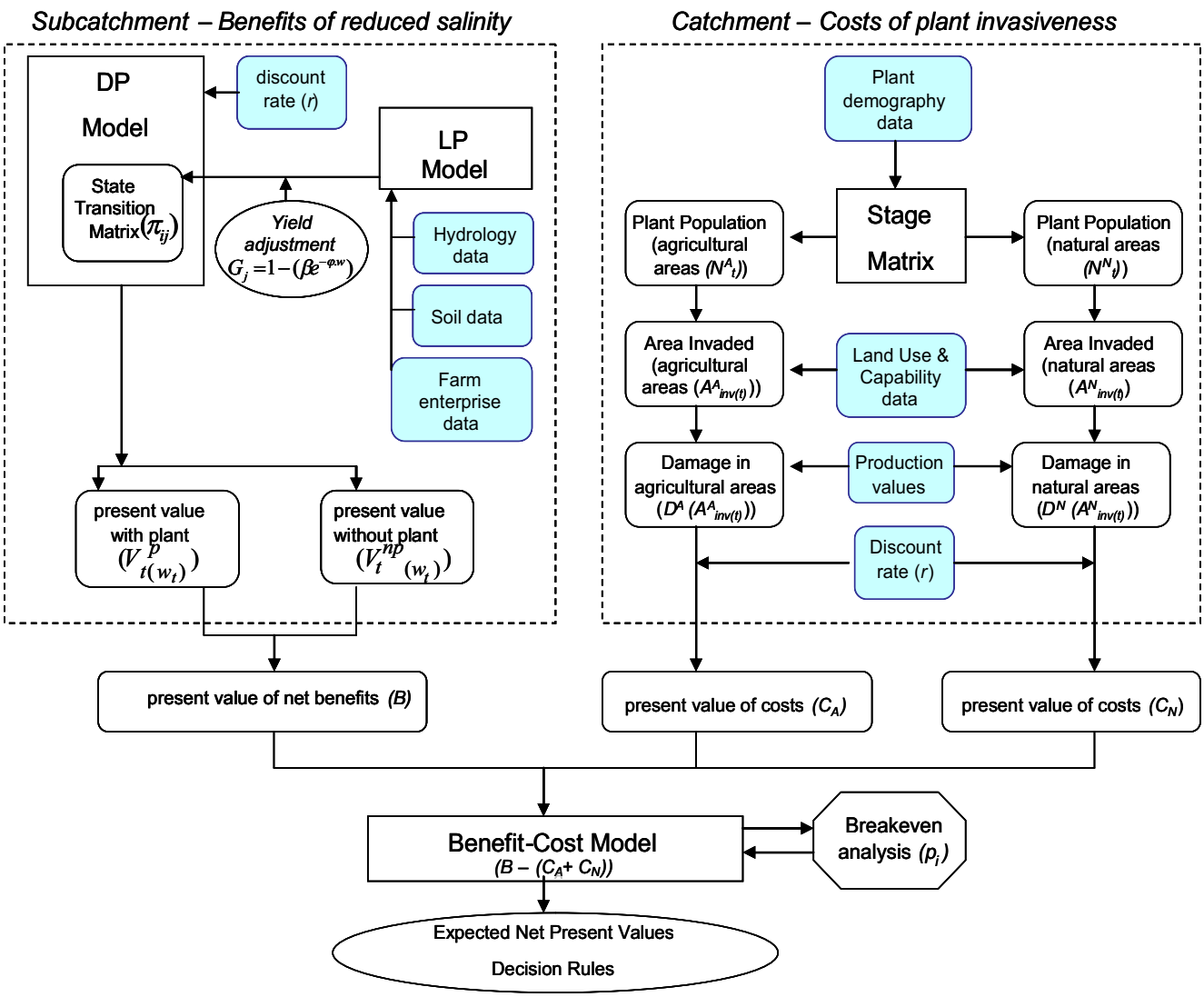


Figure 5.1 The model – biophysical and economic interactions

The estimates of present values are then brought together within a benefit-cost model, where breakeven analysis is undertaken to estimate the conditions under which introduction of the plant is acceptable. The mathematical models employed for each sub-model are now detailed.

5.2 Subcatchment sub-model – benefits of reduced recharge

The subcatchment is represented by an average farm. The objective is to maximise the present value of profits of the farm (V) in the presence of salinity emergence caused by recharge.

The Dynamic Programming (DP) sub-model before the plant has been introduced is

$$V_t^{np}(w_t) = \max(\pi_t(w_t, X_t) + V_{t+1}^{np}(w_{t+1}) \cdot \delta) \qquad \textbf{(5.2)}$$

subject to

$$w_{t+1} = f(w_t, X_t) \qquad \textbf{(5.3)}$$

$$w_0 = \textit{given, at time zero} \qquad \textbf{(5.4)}$$

Where π_t is the profit of the farm in time period t; w_t is the water table depth; X_t is a vector of J farm enterprise options; δ is the discount factor $(1+r)^{-1}$, where r is the discount rate.

Equation 5.3 is the state transition of the water table, which is a function of the farm enterprise mix and the water table depth in the previous period and is captured in the difference equation:

$$w_{t+1} = w_t + \Delta w_t \qquad \textbf{(5.5)}$$

where:

$$\Delta w_t = \frac{\sum_{j=1}^{J} k_j \cdot x_{j(t)} - d_t}{L \cdot \theta} \qquad \textbf{(5.6)}$$

The change in the water table (Δw_t) is a product of the recharge k_j associated with enterprise j; x_j is the area of enterprise j; and d_t the total amount of discharge that drains from the farm's groundwater system in period t. The total area of land is L and the coefficient θ translates the recharge into depth of the groundwater table (Cacho *et al.*, 2001; Greiner, 1994).

The maximum farm profit and associated enterprise mix of a given state transition is estimated by a linear programming (LP) model which includes recharge as well as resource constraints specific to the representative farm. The objective of the LP model is

$$Maximise\, \pi_t = \sum_{j=1}^{J} (p_j \cdot y_{j(w_t)} - c_j) x_j \qquad \textbf{(5.7)}$$

subject to

$$\sum_{j=1}^{J} a_{ij} x_j \le b_i \qquad \text{for all } i = 1, \ldots\ldots I \qquad \textbf{(5.8)}$$

$$\sum_{j}^{J} k_j x_j = \Delta\overline{w}_t + d_t \qquad \textbf{(5.9)}$$

$$X_t \geq 0 \qquad \textbf{(5.10)}$$

where:

a_{ij} = amount of the *i*th resource required by the *j*th enterprise option

b_i = supply of the *i*th resource,

$\Delta\overline{w}_t$ = the target water table change for a given state transition

p_j = revenue for the *jth* enterprise

y_j = adjusted yield for the *jth* enterprise based on the current watertable depth (w_t)

c_j = cost of production for the *jth* enterprise

The yields of enterprise *j* are given by:

$$y_j = \bar{y}_j(1 - \beta_j e^{\varphi_j w_t}) \qquad \textbf{(5.11)}$$

where $\bar{y}_j$ is the expected yield in the absence of salinity and β_j and φ_j are parameters. The expression in brackets is the yield-loss factor associated with rising water tables that lead to salinity (Cacho *et al.*, 2001).

Consider now that a new low-recharge pasture enterprise option is available. The new vector of farm enterprise options is Z_t and this decision vector replaces X_t in the model (5.2) to (5.11). The new recursive equation becomes:

$$V_t^p(w_t) = \max(\pi_t(w_t, Z_t) + V_{t+1}^p(w_{t+1}).\delta) \qquad \textbf{(5.12)}$$

The benefits of the introduction of the new pasture enterprise *B* for a given initial watertable depth can now be estimated, where 0 denotes time zero, as

$$B = V_0{}^{p}(w_0) - V_0^{np}(w_0) \qquad \textbf{(5.13)}$$

5.3 Catchment sub-model – costs of plant invasiveness

Now consider that the subcatchment is surrounded by a catchment with two distinct areas, agricultural lands and natural areas. The cost of the plant invading the surrounding catchment is estimated as

$$C_A + C_N \qquad \textbf{(5.14)}$$

where:

C_A = present value of costs of invasiveness of the plant in Agricultural areas.
C_N = present value of costs of invasiveness of the plant in Natural areas.

First consider the cost in the agricultural area. The damage of the plant invasion in the single discrete area is estimated as

$$D_t^A = (b_t{}^A(A_{inv(t)}^A).g_A) \qquad \textbf{(5.15)}$$

where

$$b_t^A = b_0 - b_t \qquad \textbf{(5.16)}$$

$$b_t = \frac{qA^A_{inv(t)}}{1+\left(\left(\frac{q}{m_b}\right)-1\right)A^A_{inv(t)}} \qquad 0 \le q \le 10 \qquad \textbf{(5.17)}$$

$$b_0 = given,\ at\ time\ zero \qquad \textbf{(5.18)}$$

D^A_t is the damage resulting from the invasion in time *t,* as a function of the size of the invasion ($A^A_{inv(t)}$), b^A_t is the quantity of lost output and g_A the value of output per hectare invaded. The output with no invasive plants, b_0 is given, and the output with invasive plants at time *t*, b_t, is estimated using a rectangular hyberbolic function (Equation 5.17) from Hester *et al* (2006). The size of the invasion ($A^A_{inv(t)}$) is expressed as a proportion of the total area at risk ($A^A_{\max}$).

The total cost of the invasion in agricultural areas is estimated as

$$C_A = \sum_{t=1}^{T}(D^A_t \cdot A^A_{inv(t)} \cdot A^A_{\max}) \cdot (1+r)^{-t} \qquad \textbf{(5.19)}$$

The cost of the plant's invasion in natural areas is estimated as using an analogous mathematical model:

$$C_N = \sum_{t=1}^{T}(D^N_t \cdot A^N_{inv(t)} \cdot A^N_{\max}) \cdot (1+r)^{-t} \qquad \textbf{(5.20)}$$

Where D^N_t , $A^N_{inv(t)}$ and $A^N_{\max}$ are the damage in natural areas, the proportion of natural area invaded and the total natural areas that could be invaded respectively and are estimated using an approach identical to that for agricultural areas.

5.4 Numerical application of the model

The numerical model was implemented in Matlab© (Mathworks, 2006) and Excel ©.

Following is a summary of the methodology employed to implement the model and its components as described.

1. A linear programming (LP) model was developed which describes the subcatchment's agricultural production for a given level of the watertable. The model identifies land use choices that maximise farm profit subject to a number of constraints.
2. The LP model was then embedded within a Dynamic Programming (DP) model. Iterative runs of the LP model were undertaken to estimate the state transition matrix from which the recursive equation of the DP optimises the state path.
3. A plant population simulation sub-model was developed for two external areas where plant invasion might occur: agricultural and natural areas. The plant population sub-model incorporates a number of stochastic elements which reflect variation in plant demographics.
4. The plant population simulations and the outputs of the DP sub-model, for a number of scenarios, were combined within Excel©.
5. Breakeven analysis was then undertaken to assess the probabilities associated with plant spread and recharge reduction, to identify the range of conditions under which the plant might be introduced.

5.4.1 Implementing the dynamic programming salinity sub-model

The benefits of the introduction of the new varieties of birdsfoot trefoil (*Lotus corniculatus* L.) were estimated as the difference in the net present value of farm profits from the optimal enterprise mix with the new plant and without the new pasture plant. The LP model selects the optimal land use activities subject to feasible changes in the key state variable, the watertable depth.

The LP incorporates eight land use options and two livestock options. Optimisation is subject to nine constraints: four seasonal stock feed requirements, four land constraints and recharge to the groundwater system. Yield of each of the land use options is a function of the watertable depth. Equation 5.11 is applied to generate the LP matrix for any given watertable depth.

Repeated solution of the LP using the yield adjustment functions at different watertable depths allows the development of a state transition matrix: the key element of the dynamic programming model. The state transition matrix incorporates all potential transitions of the watertable depth, within defined limits and step units. The numerical DP model generates two matrices associated with possible state transitions within an annual time step: A reward matrix (**R**) and an optimal enterprise-mix matrix (**X***). These two matrices are generated through repeated solution of the LP model as explained above. The matrix **R** has dimensions $M \times M$ and matrix **X*** has dimensions $M \times M \times J$ where M is the number of discrete states for which the model is solved and J is the numbers of enterprises in the LP model (Equations 5.7-5.11). Figure 5.2 shows a representation of **R.**

Depth (m)	0.50	0.51	0.52	0.53	0.54	0.55	0.56	0.57	etc
0.50	π_{11}	π_{12}	π_{13}	.	.	.	.	.	.
0.51	π_{21}	π_{22}	.	.	.	.	.	.	.
0.52	π_{31}	.	.	.	.	.	.	.	.
0.53	.	.	.	.	.	.	.	.	.
0.54	.	.	.	.	.	.	.	.	.
0.55	.	.	.	.	.	.	.	.	.
etc	.	.	.	.	.	.	.	.	.

Figure 5.2 Illustration of reward matrix, **R**

Element π_{mn} represents the net annual profit (reward) associated with the transition from state m to state n, this reward is obtained through the optimal management

strategy represented by the vector X_{mn} within the associated matrix **X***. The vector X_{mn} contains J elements, which represent the enterprise mix that achieves the state transition $m \rightarrow n$ while obtaining the maximum possible net profit, subject to the technical constraints of the LP. The two matrices **R** and **X*** are used by the DP algorithm to solve the recursive equation (5.2) before introduction of the new plant. After introduction of the new plant, equation (5.12) replaces (5.2) and **Z*** replaces **X***. The recursive equations are solved for a planning horizon of T years. The transition matrices were created for discrete values of the watertable depth (state) ranging from 0.5m to 15m at increments of 10mm, for a total number (M) of 1,451 possible state values.

Outputs of the DP sub-model include the optimal state path, the optimal trajectory of enterprise mixes and the present value of benefits for any initial value of the state variable. The DP sub-model (including its LP) was implemented in Matlab©.

5.4.2 Implementing the plant population simulation sub-model

The Leslie Matrix is the most commonly adopted approach to describe population growth numerically and has been widely adopted to describe plant life cycles. Named after P.H. Leslie who formalised its use in the 1940's, the matrix is a discrete and age-structured model which simulates the changes to a population over time (Leslie, 1945).

The area of the invasion (A_{inv}^{A}) of the plant population in this study was modeled using a variation of the original Leslie matrix based on life stages rather than age groups. The structure of the population at any time t is described by a vector, N_t, which contains the number of individuals in each of n stages of the life cycle of the plant.

$$N_t = \begin{bmatrix} n_{1t} \\ n_{2t} \\ \cdots \\ n_{nt} \end{bmatrix} \qquad \textbf{(5.21)}$$

The Stage Matrix, *H*, is square and has the same number of rows and columns as the population vector has elements. The elements of the Stage Matrix (*H*) describe the rate of change in each of the life stages in each time step (Caswell, 2001). In the same way that Cacho *et al.* (2006) describe a stage matrix which drives a weed population with four stages, the following equation describes a stage matrix for a plant population with a five stages.

$$H = \begin{bmatrix} 0 & 0 & 0 & F_J & F_A \\ P_S & P_S & 0 & 0 & 0 \\ G & G & 0 & 0 & 0 \\ 0 & 0 & P_{J1} & 0 & 0 \\ 0 & 0 & 0 & P_{J2} & P_A \end{bmatrix} \qquad \textbf{(5. 22)}$$

where the stages (rows and columns) represent new seeds, the seed bank, seedlings, juveniles and adults. *F* represents the fecundity of adults (seeds produced per plant per time period: F_J is the number of seeds produced by juvenile plants and F_A the number produced by adult plants), P_S is the proportion of seeds that do not germinate and survive from one time period to the next, *G* is the proportion of seeds that germinate and survive into seedlings, P_{J1} is the proportion of seedlings that survive to become juveniles, P_{J2} the proportion of juveniles that survive to become adults and P_A is the proportion of adults that survive from one time period to the next. For simplicity it is assumed that new seeds and the seeds in the seedbank have the same survival (P_S) and germination (*G*) rates[11].

Population growth over time is represented through a matrix multiplication.

$$N_{t+1} = H_t \cdot N_t \qquad \textbf{(5.23)}$$

The growth rate of the population (λ) is the dominant eigenvalue of *H* and is associated with the intrinsic rate of population increase (*r*) where $\lambda = e^r$ (Caswell, 2001).

[11] Cacho *et al.* (2006) cite evidence that this simplification does not affect the dynamical properties of the matrix model.

The area invaded is estimated based on the size of the plants in each life stage, represented in a vector *S*. The area invaded is:

$$A^{A}_{inv(t)} = N_t' \cdot S / 10{,}000 \qquad \textbf{(5.24)}$$

where *S* has the same dimensions as *N*, represents the area (m^2) occupied per plant and is divided by 10,000 to transform the expression into an expression per hectare.

Density dependence is incorporated to the model by assuming that the population growth becomes zero when the whole area is covered by weeds and a steady state is reached. As per Cacho *et al.* (2006) an interpolative approach to identify a growth path between exponential growth in the early stages of the plant invasion and the steady state of a mature invasion has been used.

The growth path of the invasion over time is simulated using two stage matrices: one for the 'beginning' of an invasion (H_0) when the growth rate $\lambda > 1$ and one for the 'end' of the invasion (H_∞) when the growth rate $\lambda = 1$. Interpolation between H_0 and H_∞ is based on the density of the population at the given time t, as explained in Cacho *et al.* (2006).

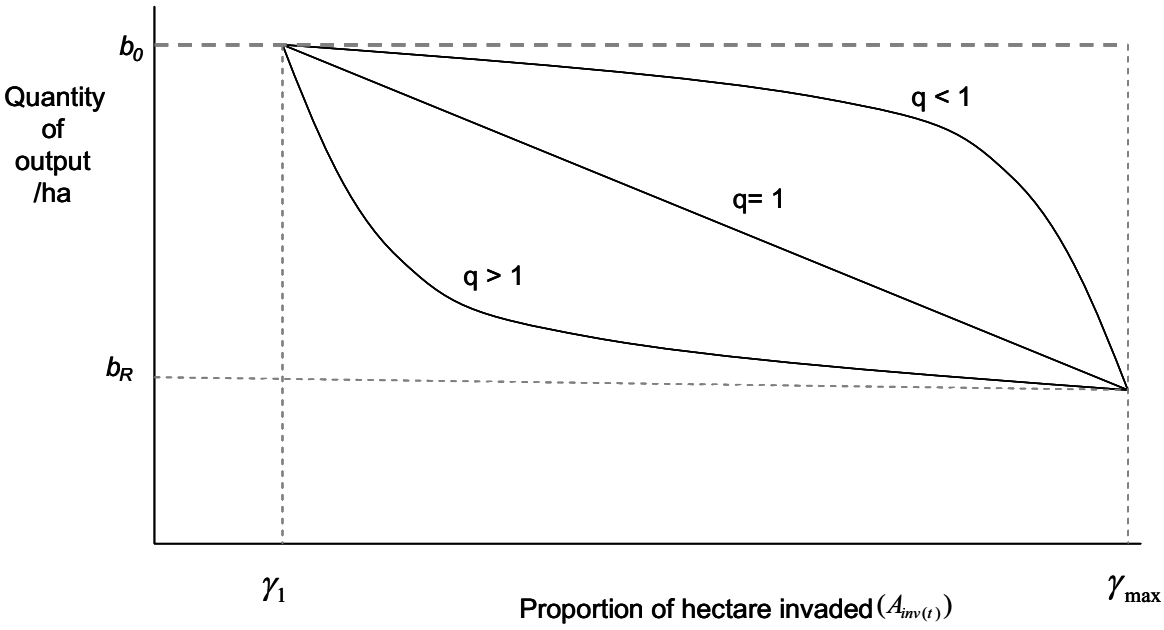
The parameter values in the stage matrix are uncertain. Monte Carlo simulations using beta distributions were undertaken in the analysis, allowing the growth path to reflect random variation in the key demographic parameters: germination; survival; and plant fecundity.

The beta distribution was selected for the Monte Carlo simulations because of its flexibility and ease of definition (Cacho *et al.*, 2006). A covariance matrix was incorporated using the Copulas technique (Perkins and Lane, 2003) to allow consideration of any stochastic dependencies which may exist between the demographic parameters.

The plant invasion is simulated on a per-hectare basis, assuming a homogeneous invasion. The proportion of the 'average' hectare invaded in each time period $(A^{A}_{inv(t)})$ is also the proportion of the total area at risk covered by the invasion. So the net area of agricultural land invaded at time t is estimated as

$$TA^{A}_{inv(t)} = A^{A}_{inv(t)} . A^{A}_{\max} \qquad \textbf{(5.25)}$$

The losses associated with the invasion were calculated based on equations 5.19 and 5.20 for agriculture and natural areas respectively. These loss functions are based on a rectangular damage function (5.17). The shape of the function is illustrated in Figure 5.3 for different values of q. Values of $q > 1$ represent immediate, rapid reductions in the output of services, while values of $q < 1$ represent initially smaller reductions in outputs followed by greater marginal falls as the maximum invasion is approached. When $q = 1$, the marginal decline in the output of services is constant.



Source: Adapted from Hester *et al* (2006)

Figure 5.3 Damage function (Equation 5.17)

Thresholds for the beginning of the decline in ecosystem or agricultural output and the maximum damage have been incorporated in relation to the proportion of the hectare invaded. These are shown in Figure 5.3 as b_0 (value initially) and b_R (value remaining at full invasion) which correspond to the invasion thresholds, γ_1 and $\gamma_{\max}$ respectively. At each time step, the deficit between the total value originally available and that available following the invasion of the plant is calculated $(b_0 - b_t)$ such that a stream of marginal losses over time is estimated as in Equation 5.15.

Outputs of the model include the weed population growth path for both agricultural and natural areas, the stream of costs associated with the invasion over time and the net present value of the invasion. These values are calculated for a range of plant demographic parameters and, for the stochastic analysis, the distribution of costs.

The data used in the application of this model are now presented.

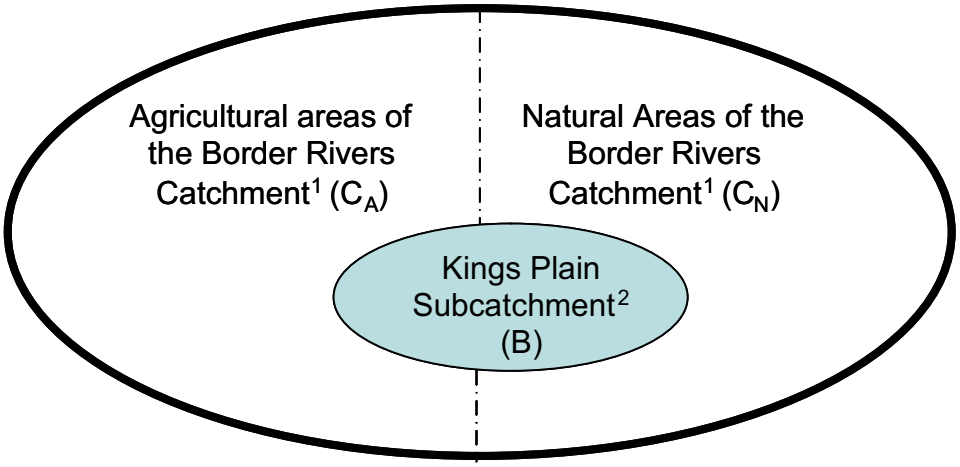
6 Data for a case study application

Implementation of the bioeconomic model developed in Chapter 5 requires parameter values to be determined for a particular situation. This allows demonstration of the framework for biological introduction decisions and identification of decision rules for the development and introduction of new plants. The potential introduction of new varieties of *Lotus corniculatus* L. to the Kings Plain Subcatchment of the Border Rivers Catchment of northern New South Wales is used as the case study. This chapter details the data used in the analysis

To identify a suitable case study plant for this study a questionnaire (see Appendix B) was developed and distributed to sixteen plant breeders known to be working with new pasture plants with the potential to reduce recharge of groundwater systems. From the responses received, *Lotus corniculatus* L., more commonly known as birdsfoot trefoil, was selected for assessment. Selection was based on availability of data and location of potential suitable case study sites.

The Kings Plain Subcatchment (KPS) on the north-east slopes of New South Wales was selected as the case study area for introduction because it suits the intended application within the birdsfoot trefoil (BFT) breeding program in terms of location, and is within the region where the plant's adoption might be encouraged. The KPS is located north east of Inverell within the NSW section of the Border Rivers Catchment (BRC) of northern New South Wales. The analysis is undertaken at a Catchment level because natural resource management in New South Wales predominately occurs through Catchment Management Authorities (CMAs), and subcatchments and catchments represent discrete areas for which the problem of externalities can be considered.

An adapted version of Figure 4.5 is shown below (Figure 6.1) to put the case study in the context of the conceptual model from Chapter 4 and the empirical model from Chapter 5.



[1] Costs are estimated using the plant population simulation sub-model
[2] Benefits are estimated using the DP salinity sub-model

Figure 6.1 Case study area in the context of the conceptual and empirical models

This chapter provides a description of *Lotus corniculatus* L. along with the data used to estimate the benefits of its introduction as a pasture in the Kings Plain Subcatchment, and to simulate its potential invasion if dispersed into the surrounding areas of the Border Rivers Catchment. This is followed by descriptions of the case study area, the agricultural system that might benefit from the addition of a new deep-rooted perennial pasture option and the extent and value of areas at risk of invasion if the plant is dispersed in the surrounding catchment.

6.1 *Lotus corniculatus* L.

Lotus corniculatus L. is known as birdsfoot trefoil due to the shape of its umbels (collection of seed pods) (McLaughlin and Clarke, 1989). Herein referred to interchangeably as *L. corniculatus* L. or birdsfoot trefoil (BFT), the plant is one of some 180 annual and perennial species which comprise the *Lotus* genus (Ayres *et al.*, 2006a). *Lotus* is native to Europe, North Africa and parts of Asia; though there is significant agronomic use in North and South America, Europe and Australasia (Ayres *et al.*, 2007). In the United States, birdsfoot trefoil accounts for the largest proportion of the *Lotus* species used agriculturally, with around 1 million hectares of the pasture in use (Blumenthal and McGraw, 1999).

Birdsfoot trefoil (BFT) is a perennial legume pasture plant with a taproot similar to lucerne (*Medicago sativa* L.). It is an erect and prostrate outcrossing species which has been found to be an excellent non-bloating forage which has softer stems and higher carbohydrate content than lucerne and clover (Bennett, 2003a). Further, it is productive on infertile soils (Ayres *et al.*, 2007; Bennett, 2003a; Blumenthal and McGraw, 1999) ranging from poorly drained to drought soils and in acid to mildly alkaline conditions (Bennett, 2003a). It is often referred to as 'poor-land lucerne' based on its ability to do well on soils which are more acid, drier, prone to water-logging and less fertile than those in which lucerne thrives (Blumenthal and McGraw, 1999).

There have been attempts to use birdsfoot trefoil in Australian pasture systems. These have been limited in the first instance by the need for specific rhizobia required for this leguminous species (Ayres *et al.*, 2008) and secondly by insufficient photoperiod (daylight hours) at areas of low latitude to allow full expression of the plant's reproductive processes (Ayres *et al.*, 2007; Blumenthal and McGraw, 1999). These factors have contributed to the pasture having poor persistence and thus low adoption. One commercial cultivar, Grasslands Goldie, developed in New Zealand, has been available commercially in Australia together with an effective inoculant. Still, adoption of Grasslands Goldie has been restricted to pasture demonstrations (Ayres *et al.*, 2008) because the variety's performance has remained limited by photoperiod and therefore has not been a persistent pasture.

Birdsfoot trefoil was identified by the Cooperative Research Centre for Plant-Based Management of Dryland Salinity (CRC PBMDS) as a perennial pasture plant with the potential to reduce excess water recharge of groundwater systems that leads to salinisation of Australian landscapes (see Table 3.2). So that this plant might offer a viable pasture alternative with persistence, breeding efforts have been focused on selection of lines for enhanced seedling recruitment and population density (Kelman and Ayres, 2004) and, more specifically, adapting cultivars to a shorter photoperiod to improve flowering and seed production (Ayres *et al.*, 2008). Success in the program has led to the development of three new varieties. These present Australian farmers with new perennial legume options adapted to low soil fertility and highly variable climatic conditions and which may also help farmers manage recharge of groundwater

systems (Ayres *et al.*, 2007). The new birdsfoot trefoil cultivars which have provisional Plant Breeders Rights (PBR) protection are 'Phoenix', 'Venture' and 'Matador' (Ayres *et al.*, 2008). Table 6.1 summarises the characteristics of the new varieties with respect to their environmental tolerances.

Table 6.1 Summary characteristics for new varieties of *L. corniculatus* L.

Parameter	Tolerance
Minimum pH	4.8
Average annual rainfall limits (mm per annum)	650 – 1000
Most active growing season	spring - autumn
Land class preference	Coastal hill country, Tablelands, Slopes
Soil fertility requirements	Low
Toxicity tolerances or intolerances	Tolerant of soil acidity and low phosphate soils
Rhizobial requirements	Requires specific commercial inoculant
Production practices	Seed requires scarification and inoculation

Source: Ayres *et al* (2006a) Ayres *et al* (2006b) and pers comm., Ayres and Lane (2007)

The release of new, more persistent varieties of BFT means the potential invasiveness of these new pasture plants should now be considered. Therefore they provide a pertinent case study to consider the economic framework developed to assess potential new plant introductions which may present benefits, such as reduced recharge, but also costs, such as those associated with invasiveness.

An understanding of how the new varieties of BFT will perform as a pasture (productive capacity, costs of establishment, ability to limit recharge) and the demographic features which will govern its potential to be invasive, is necessary for this assessment. Each of these is provided in the following sections with the data and assumptions used in the later analysis. The growth habit and vegetative morphology varies between the three new varieties, but all express improved seed yield and agronomic performance from the original variety, Grasslands Goldie (Ayres *et al.*, 2008). The parameters reported are representative of the new varieties collectively for the purpose of this study (pers comm., Ayres and Lane, 2007).

6.1.1 Pasture production

Two methods for incorporating birdsfoot trefoil into a pasture system include:

1. direct drill binary seed mix (e.g. birdsfoot trefoil and tall fescue) into herbicide treated 'run-down' sward; and
2. broadcast (fertiliser spinner or aerial application) application into native pasture.

Successful establishment of BFT pasture includes the need to prepare the seed through inoculation (coating with a commercially sourced specific rhizobium) and scarification (the softening or abrasion of the seeds in preparation for planting to increase germination) as well as conducive site conditions (addition of fertiliser and, if appropriate, cultivation or herbicide application). The productive capacity of BFT under each of the two methods has been estimated for the Kings Plain Subcatchment. The productive capacity of BFT will be a key determinant of the extent to which BFT might be adopted on farm.

Pasture production can be measured in terms of metabolisable energy (ME) produced on a per hectare basis. Alford *et al* (2004) represented pasture production in terms of ME, and matched it to the ME requirements of the animal enterprises included within a linear programming model developed for the Northern Tablelands of NSW. Total metabolisable energy production (TME) per hectare is estimated as a product of the quantity or yield of pasture produced per pasture type (kilograms of dry matter produced per hectare, kg DM/ha) and the quality of the pasture produced (ME measured as the megajoules of metabolizable energy per kilogram of pasture dry matter, MJ/kg DM)). This relationship is shown in Equation 6.1.

$$\text{TME} = \text{ME (MJ/kg DM)} \times \text{yield (kg DM/ha)} \qquad \textbf{(6.1)}$$

Estimation of these parameters on a monthly basis allows incorporation of the seasonality of pasture production into whole-farm models and this approach from Alford *et al* (2004) has been adopted for this study.

In consultation (pers comm., Ayres, 2006) and based on field trials of the Centre for Perennial Grazing Systems, Glen Innes, TME estimates for BFT were obtained for the north east slopes of New South Wales. A summary of these estimates is shown in Table 6.2.

Table 6.2 Birdsfoot trefoil pasture production (TME / ha)

	BFT in Tall Fescue Pasture	**BFT in Native Pasture**
Summer Total	24,216	17,083
Autumn Total	24,487	22,143
Winter Total	10,484	4,475
Spring Total	34,680	25,894
Annual Total	***93,867***	***69,594***

Source: Pers comm., Dr John Ayres, Centre for Perennial Grazing Systems, Glen Innes, (2006)

Tables detailing available dry matter and metabolisable energy produced by month, can be found in Appendix C.

Of total pasture produced only a proportion is ever utilised by stock. Pasture utilisation will vary considerably by enterprise, management practices, pasture type and climate (Alcock, 2006). Estimated pasture utilisation that is achievable on NSW grazing properties is estimated to range from 20 percent to more than 50 percent (Alcock, 2006; King and Hutchinson, 1995; MLA, 2004). As such the amount of pasture effectively available is somewhere between 20 to 50 percent of the estimated total production shown in Table 6.2. The pasture utilisation rate has been set at 40 percent which is within this range and at the higher end of the range to reflect improving grazing practices over time.

6.1.2 Costs of pasture establishment

The financial aspects of the establishment, management or profitability of new or existing varieties of BFT as a pasture in Australia have not been documented. This study requires representative gross margins for the linear programming model and so a gross margin has been developed for the establishment of BFT pastures for each of the methods described in the previous section. These are summarised in Table 6.3, with details provided in Appendix D.

Table 6.3 Birdsfoot trefoil establishment gross margins ($/ha)

	1) direct drill **BFT and tall fescue** into herbicide treated 'run-down' sward	**2)** broadcast (spinner or aerial) application of **BFT into native pasture**
Establishment	- $173.11	- $152.24
Annual (when spread over 5 yr stand life)	- $34.62	- $30.45

Source: Developed in consultation with Bob McGufficke, District Agronomist, NSW DPI, (2006). For full gross margin see Appendix D.

Expectations are for new varieties of birdsfoot trefoil to have a stand life approaching or exceeding 10 years (pers comm., Ayres and Lane, 2007) such that the gross margins in Table 6.3 could be considered conservative regarding the costs of establishment spread over the life of the pasture.

6.1.3 Recharge

The attraction of BFT is that it offers a pasture option with reductions in recharge of the ground water table similar to lucerne, in areas where lucerne is not well adapted. Reductions of recharge to between 0 – 3 percent under species such as BFT, down from 6 – 11 percent under annual crops and pastures, have been recorded in trials in the 400 – 750 mm rainfall zone of NSW (Bennett *et al.*, 2002). Table 6.4 shows estimates of recharge under BFT, by soil type of the Kings Plain subcatchment. Recharge is measured in millimetres per square metre (mm/m^2).

Table 6.4 Recharge under birdsfoot trefoil pasture in the Kings Plain Subcatchment, North East Slopes, NSW (mm/m^2)

	Birdsfoot Trefoil in Tall Fescue	Birdsfoot Trefoil in Native Pasture
Deep black cracking clays	5-20	10-30
Deep structured red clay loans	10-30	10-40
Stony sandy loams	20-50	30-60
Yellow & red texture contrast	10-30	20-50
Average	**10-30**	**17.5-45**

Source: Pers comm., Dr Sean Murphy, Farming Systems Hydrologist, Tamworth Agricultural Institute, NSW Department of Primary Industries, December (2006). Reviewed by Ms Carol Harris Senior Research Agronomist, NSW Department of Primary Industries, Centre for Perennial Grazing Systems, Glen Innes and found to be reasonable, December (2006). Comparable estimates of recharge in the Kings Plain Subcatchment for other land uses are provided in Table 6.13

6.1.4 Evidence of invasiveness

A review of literature has found no academic studies relating to the invasiveness of birdsfoot trefoil[12]. This includes both internationally where BFT is a widely used pasture plant and in Australia where there has been limited introduction of BFT. This has been confirmed by researchers both in Australia (pers comm., Ayres, 2006; Kelman, 2006) and internationally (pers comm., Bueselinck, 2008). There are no formally documented conclusions in this regard but there is anecdotal evidence and documentation of characteristics that suggest that transfer of birdsfoot trefoil seeds into new areas will be limited and there will be factors that limit the ability of it to establish, even if seeds are transferred.

The primary vectors for transfer would include agricultural machinery, the faeces of grazing animals, hay that might be cut from pastures that include birdsfoot trefoil and, potentially, in flood waters (pers comm., Bueselinck, 2008). The small seeds are not adapted to be dispersed in the wind or to attach to clothing or animals and are not favoured by birds or other ranging animals (pers comm., Bueselinck, 2008). Further, the quality of seed produced by birdsfoot trefoil varies considerably throughout the

[12] The United States Department of Agriculture (http://plant-materials.nrcs.usda.gov/) and Invasive Species Specialist Group (http://www.issg.org) note that *L. corniculatus* L. may become invasive in some regions or habitats without appropriate management.

year (McGraw *et al.*, 1986) such that any dispersal events would need to coincide with good-quality seed production to ensure dispersal of viable seeds. However, it should be noted that seeds have the potential for longevity in the soil (Ayres and Lane, 2007).

Even if seeds are transferred, population establishment will be limited. Particularly relevant to the establishment of leguminous plant populations is the need for effective nodulation for successful establishment (Ayres *et al.*, 2006a) . Birdsfoot trefoil has a requirement for specific rhizobia and its effective nodulation is essential for seedling vigour and subsequent yield performance (Hughes and MacDonald, 1951). Therefore the potential for BFT to establish in unintended areas is reduced because the specific rhizobia are unlikely to be present. Birdsfoot trefoil also tends to have a high proportion of hard seed such that germination can be delayed unless the seeds are naturally scarified. Further, the plant has not proven to be a strong competitor, with the only instances of the plant exhibiting any invasive tendencies being on disturbed vacant areas, such as building sites, where any plant competition has been removed (pers comm., Bueselinck, 2008). Finally, BFT can be prone to foliar and root diseases which further reduces overall plant performance and limits the seedling and juvenile rates of survival (Bueselinck *et al.*, 2005). This is exacerbated in the absence of effective nodulation. These factors all limit the potential establishment of new populations.

However, there is also some anecdotal evidence to suggest that there may be spread of the plant at the periphery of any birdsfoot trefoil pastures that are planted. On a field trip to the Glen Innes Centre for Perennial Pastures in October 2007 it was observed that surrounding a 10 m by 40 m trial plot of Goldie Grasslands birdsfoot trefoil there were 15 discernible patches of birdsfoot trefoil plants outside the trial area, each in the order of $4m^2$. The implied growth in the area of this population from this observation is 1.5 percent annually over the 10 years. The trial had been grazed and subject to similar conditions as a commercial pasture.

Based on documented and anecdotal evidence, there is little to suggest the new varieties of BFT will be invasive. BFT invasiveness has however not been addressed

directly in the literature in Australia or internationally. Further, the concern here is with new varieties with enhanced reproductive capacity and potential persistence.

6.1.5 Population dynamics

Table 6.5 provides estimates of some of the parameters which describe the demography and dynamics of the new varieties of birdsfoot trefoil (BFT) that might be expected when seeds are dispersed outside of an intentionally planted pasture of BFT (i.e. uncultivated, unscarified seed, without specific rhizobia, without fertiliser).

Table 6.5 Demographic parameters of new varieties of *L. corniculatus* L.

Parameter	Estimate
Seedling survival (seedlings that make it to juvenile stage) [a]	10%
Longevity of seeds in the soil seedbank	10 + years
Seeds produced by mature plant per annum	2,875
Seeds produced by juvenile plant per annum	200
Years to plant maturity	2
Plant longevity	10+ years
Size of Mature plants	0.04 m^2 [b]
Ratio of size of Mature plants: Juveniles	30

Source: Pers comm., Ayres and Lane (2007)
[a] The value for this parameter is considered indicative of those that might be observed when the seeds are dispersed outside of an intentional pasture planting. That is, it does not reflect the performance of the plant as a pasture when planted intentionally. [b] This size implies 25 adult plants per m^2. In the case where birdsfoot trefoil is grown with another pasture such as tall fescue it is assumed that in a square metre there might be one-third birdsfoot trefoil plants to account for the tall fescue plants and other recruited native species. This implies 80,000 birdsfoot trefoil plants per hectare.

Values for the key parameters that describe the population dynamics are difficult to estimate. In any case it can be difficult to elicit values for parameters such as germination, but this is especially true for a new variety in an unintended area of establishment. Therefore instead of directly estimating these values, the population growth rate was estimated and the values for the parameters were determined via

reverse induction. This is possible because the population growth rate (λ) is described by the dominant eigenvalue of the stage matrix (H in Equation 5.22) and is related to the intrinsic rate of increase (r) by the function $\lambda = e^r$ (Caswell, 2001; Cacho *et al.*, 2006).

On the basis of the evidence observed on the field trip to the Centre for Perennial Grazing Systems in October 2007 (see Section 6.1.4), the annual growth rate was estimated to be $\lambda = 1.015$. If the new varieties of birdsfoot trefoil are twice as persistent as Goldie Grasslands, then the implied growth rate would be $\lambda = 1.03$. The implied growth rate on the basis of the field trial observations, however, underestimates the true population growth rate because it is based on the observed area of adults only. Therefore, the population growth rate is likely to be higher than 1.015 for Goldie Grasslands and higher than 1.030 for new varieties that might be twice as persistent.

A modified version of the density-dependent plant simulation was written so that the eigenvalue, λ, could be specified along with parameters for which values are more easily observable (Table 6.5). The values for germination, juvenile survival and adult mortality were induced for the steady state point of an invasion over time following the technique explained by Cacho *et al.* (2006). On the assumption that population growth might be at least twice as high as the existing variety, a growth rate of 1.0325 was set for the base simulations as a conservative estimate of the growth rate. The stage matrix parameters, including the implied parameter values calculated for the population from this growth rate, are shown in Table 6.6.

Table 6.6 Stage matrix parameters (Equation 5.22)

Parameter description	Value
Proportion of new seeds entering the seedbank[a] (P_S)	0.501190
Proportion of seeds retained in the seedbank[a] (P_S)	0.501190
Germination rate of new seeds[a] (G)	0.003048
Germination rate of seeds in the seedbank[a] (G)	0.003048
Seeding survival (proportion of seedlings that survive to juvenile stage) (P_{J1})	0.125000
Juvenile survival (proportion of juveniles that survive to adulthood) (P_{J2})	0.273350
Annual adult survival (P_A)	0.421700
Fecundity of juveniles (F_J)	200
Fecundity of adults (F_A)	2,875

[a] As per Cacho *et al.* (2006) the rates of germination and seeds in the seedbank have been set to be the same for both new seeds and seeds produced in previous years.

6.2 The Border Rivers Catchment

The NSW section of the Border Rivers Catchment (BRC) is managed together with the Gwydir River Catchment by the Border Rivers-Gwydir Catchment Management Authority (BRG CMA). Only the BRC is considered in the plant population modelling in this analysis because a complete data set (land-use and land capability) was not available for the Gwydir section of the management area. Figure 6.2 shows a map of the BRC.

The BRC covers some 2,418,212 hectares and extends from the Queensland border near Goodiwindi in the north, to just east of Tenterfield and Glen Innes in the east, just north of Moree and just south of Inverell in the south and to Mungindi in the west. The major river of the catchment, as shown in Figure 6.2, is the Macintyre. Other rivers include the Beardy, Boomi, Dumaresq, Mole and Severn Rivers.

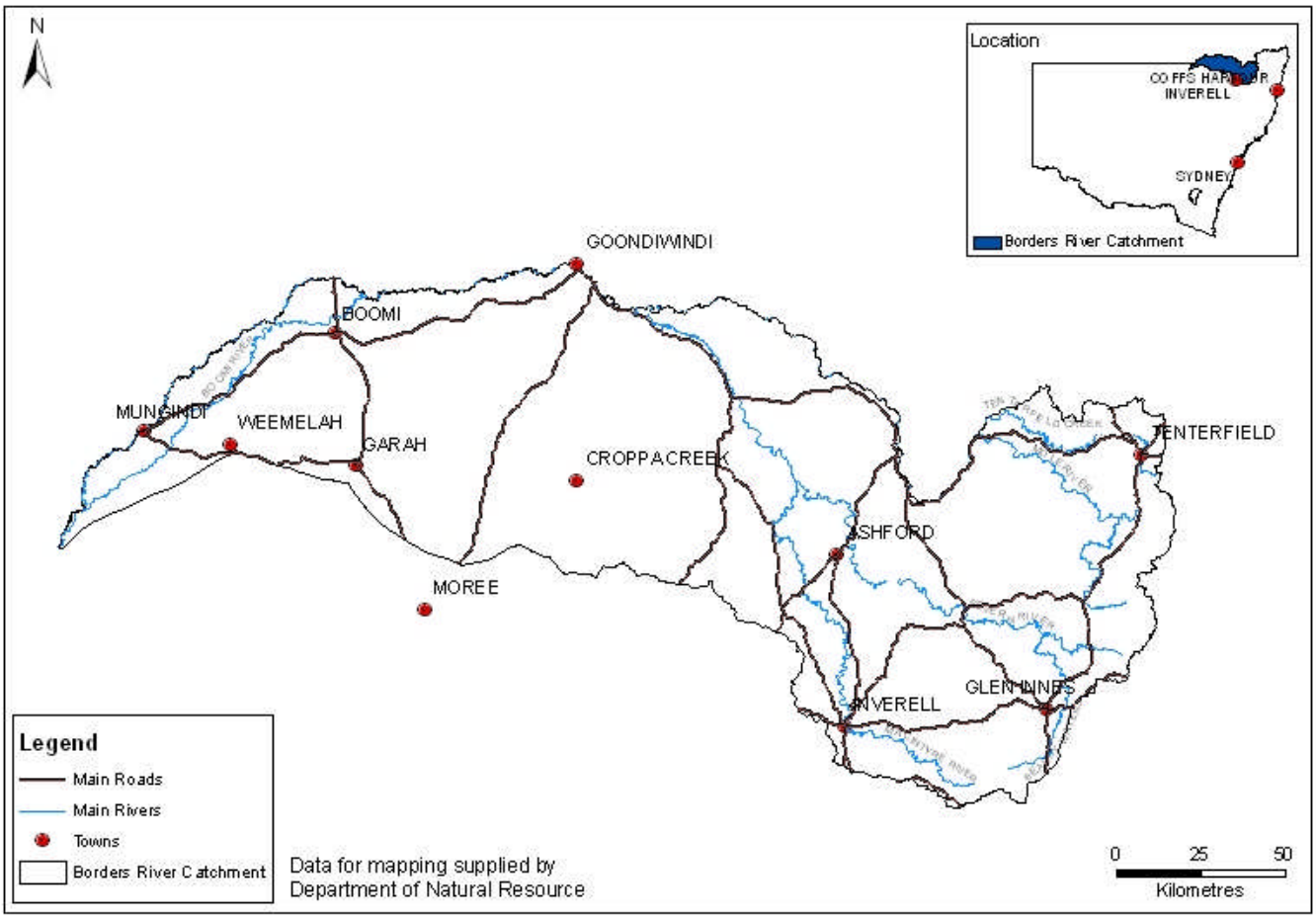


Figure 6.2 Case study area: Border Rivers Catchment

The BRC is a diverse region. In the east, the catchment is characterized by higher altitude areas consisting of patches of extensively forested areas, where clearing has occurred largely on the river flats and adjacent slopes. Grazing of sheep and cattle are the primary agricultural enterprises. As the catchment extends west, there is more open forest, shrub-land and grassy plains. Significant areas have been cleared for broadacre dryland and irrigated farming. In contrast to the east, the agricultural enterprise mix in the west is dominated by cropping, including cereals, cotton and a range of other summer and winter broadacre crops. The slopes in between are characterized by grazing with some opportunistic cropping (BRG CMA, 2006).

Limiting the extent of salinity in the Catchment is a priority for the BRG Catchment Management Authority (CMA). Specifically, the CMA aims for there to be no increase in the extent and severity of dryland salinity in the catchment, and includes

among its management targets an aim to increase the current area of deep-rooted perennial vegetation by 10 percent in recharge areas of the Border Rivers-Gwydir catchment by 2015 (BRG CMA, 2006). Further, the CMA has aims in relation to the management of biodiversity including minimization of weed invasions and halting the increase in numbers of new weeds in the catchment.

Table 6.7 provides a summary the broad characteristics of the BRG catchment relevant to this study.

Table 6.7 Border Rivers Catchment – A summary of characteristics

Descriptive Features			**Source & Comments**
Total area	2,418,212 ha		Calculated from data supplied by DNR,(2006)
Local Government Areas	All of the Glen Innes LGA and parts of the following LGAs: Moree Plains, Yallaroi, Inverell, Severn, Tenterfield.		2001 Census
Natural Areas (State Forests, National Parks, wetlands)	519,889 ha of timbered area		Includes all 50 ha mapped areas where the crown canopy cover (ccc) exceeds 50% and areas where ccc exceeds 20% and it is judged to be non-agricultural in its use. Calculated from data supplied by DNR, (2006)
Endangered vegetation	at least 54 species		54 species identified in the Inverell-Yallaroi LGAs, (DLWC, 2002).
Land use	Cropping	653,487 ha	Prepared from data supplied by DNR, (2006).
	Horticulture	262 ha	
	Mining	4,922 ha	
	Pasture	1,236,687 ha	
	Timber	519,889 ha	
	Urban	2,966 ha	
Land capability	Classes 1, 2, & 3	1,235,650 ha	Calculated from data supplied by DNR, (2006). Class definitions can be found in Appendix E.
	Classes 4 & 5	435,301 ha	
	Classes 6	407,292 ha	
	Classes 7 & 8	291,439 ha	

6.2.1 Areas at risk

Understanding the maximum potential area that a plant may invade is the key first step to understanding the extent of its likely impact. Land-use data[13] have been used to identify the potential areas of adaptation for BFT in the BRC. The extent of all agricultural and natural areas is summarised in Table 6.8. The proportion of the total area of BRC that might be at risk of BFT invasion has been estimated as that proportion of the Catchment that has at least the minimum annual average rainfall required for BFT establishment. This is also shown in Table 6.8.

Table 6.8 Potential area at risk of birdsfoot trefoil invasion in the Border Rivers Catchment (ha)

	Agricultural Land	Natural Areas [c]	Other (mining, urban) [d]	Total
Total external areas in the Catchment [a]	1,890,435	519,889	7,888	2,418,212
Estimated area suitable to BFT adaptation [b]	1,260,290	346,593	-	1,606,883

[a] Land-use data supplied by the NSW Department of Natural Resources, September, 2006.
[b] Estimated on the basis that two-thirds of the Catchment area has an average annual rainfall of at least 650mm (New South Wales Government, 2007).
[c] The total timbered area, as per Table 6.7, has been used as a proxy for natural areas in the absence of complete mapping of the areas of national parks, state forests, reserves, wetlands and other natural areas. This proxy may over or underestimate the total number of hectares of natural lands.
[d] Other areas have not been included in the analysis undertaken in this study.

BFT is likely to adapt to some soil types, some land use areas and areas with different topography, and according to climatic variation, differently even within the relatively small area being used in this analysis. There is now a range of tools that can be used for to estimate where a plant might invade. These include SimClim as refined by the CRC PBMDS (pers comm.,Crawford and Imhof, 2006) and CLIMATE (Pheloung, 2006; Stone *et al.*, 2008). These models were not accessible at the time of the analysis for this project but could be adopted for future analyses. Programs such as these have the capacity to more closely define the potential area of adaptation based on climatic, geographical and geological variations within an area. Without access to

these models, the estimated area of agricultural and natural lands to which BFT might adapt, as shown in Table 6.8, were adopted as the maximum potential areas at risk in the simulation modelling for this analysis.

6.2.2 Production values

For the purpose of assessing the impact of a potential invasion of BFT in the catchment it is necessary to identify values associated with land uses across the catchment. Table 6.9 outlines the annual values placed on output in the agricultural and natural areas of the Catchment prior to the invasion of BFT.

Table 6.9 Agricultural & ecosystem production values for the Border Rivers Catchment

Land use	Estimated annual value per hectare	Basis of estimation / source
Agriculture	$221	Average value of agriculture in the Northern Statistical District of NSW found by dividing the total annual value of agricultural production by the total reported hectares of agricultural production ABS (2001)
Natural Areas	$6.70	Sinden and Griffith (2007) estimated the annual value per lost endangered species to be $64,830. On the basis of there being at least 54 endangered species in the Catchment (DLWC, 2002), the total annual value was apportioned across the total hectares of natural lands in the Catchment to derive a conservative per hectare estimate of the value of output in natural areas.

The losses in the Catchment are estimated as the annual reduction in the values shown in Table 6.9 as the invasion of birdfoot trefoil proceeds over time. The damage functions used for this analysis (Equation 5.17) are shown in Figure 6.3 where output is now shown in terms dollars per hectare.

The value of q, which describes the shape of the damage functions, and thus the marginal losses, as the density of a plant invasion increases is assumed to be less than 1. This is consistent with findings that later stages of plant invasions are reflected by

[13] Land-use and land capability data correlate at high levels of aggregation such that either could be used, and in this case land use data have been used due to ease of mapping.

$q > 1$ (van Wilgen *et al.*, 2004) and those of new invasions or those in earlier stages reflected by $0<q<1$ (Sinden *et al.*, 2007). A value of $q = 0.5$ has been adopted for the estimation of losses to agriculture and natural areas but a range of values was tested for their impact on the analysis.

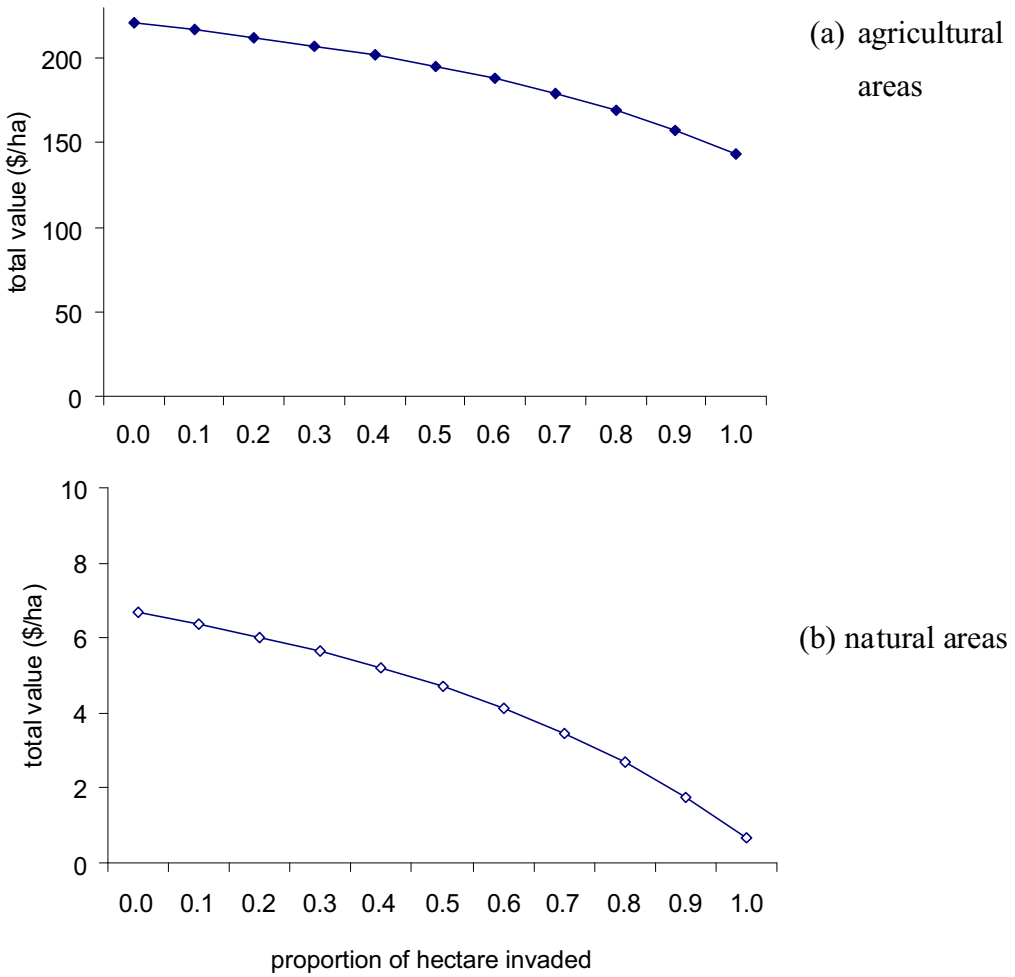


Figure 6.3 Damage loss functions expressed in dollar terms – *(a) agricultural and (b) natural areas*

These damage functions require that lower loss thresholds be specified. At the theoretical extreme, 100 percent of cropping production could be lost at full invasion of the birdsfoot trefoil in cropping areas. In pasture areas losses could be assumed to be zero as it is a useful addition to the pasture. This, as a weighted average across the cropping and pasture areas of the catchment where 65 percent of the agricultural land is used for pastures and 35 percent for cropping (DNR, 2006), equates to an average loss in agricultural areas of 35 percent at full invasion. So the lower bound on the

damage function for agricultural areas is $144 per hectare per annum (65 percent of the initial value of output).

For the purpose of the analysis in natural areas, it is assumed that 90 percent of the total number of endangered species in the area could be lost at full invasion of a plant species. So the lower bound of the damage function for natural areas is $0.70 (10 percent of the initial value of output).

6.3 Kings Plain Subcatchment

The Kings Plain Subcatchment (KPS) is a small subcatchment, just 60 km long and just over 20 km at its widest, located north-east of Inverell and part of the slopes area of the BRC. A map of the subcatchment is shown in Figure 6.4.

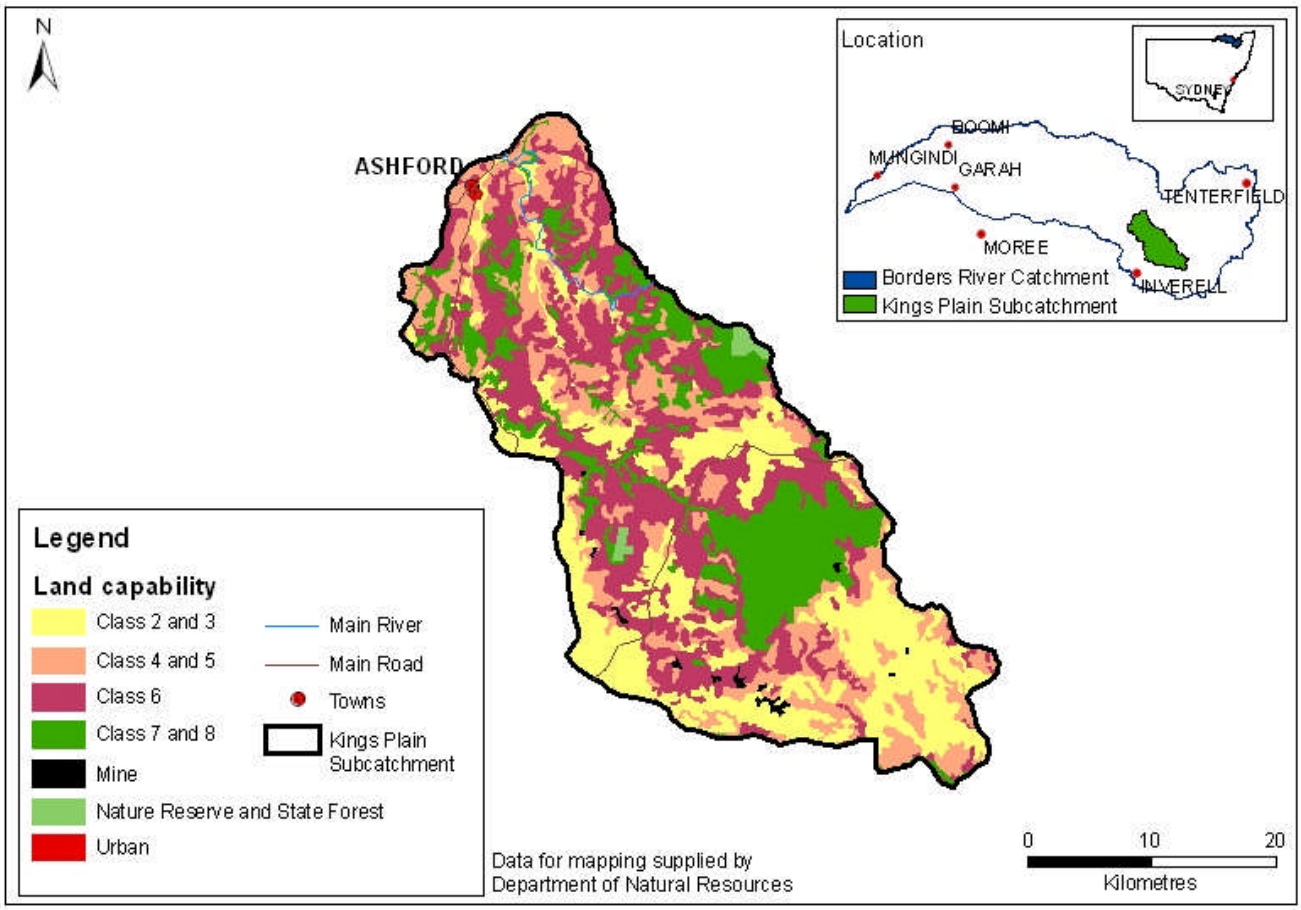


Figure 6.4 Kings Plain Subcatchment land capability

KPS is 115,980 ha and is characterised by undulating hills and open slopes with areas of open forest interspersed with patches of more dense forest. Agricultural producers are the primary landholders in the subcatchment. There are around 250 farming establishments[14] which average around 480 ha in size, but vary in the range of 200 ha to well over 2000 ha (pers comm., McGufficke, 2006). The LP matrix for the representative farm used in this study is presented in Appendix F. The matrix includes production and recharge data for BFT, as outlined in earlier sections of this Chapter, and data to describe all other land-use production and constraints as detailed in this section.

Grazing dominates the local enterprises, with a mixture of both sheep and cattle. Local pastures range from unimproved native pastures consisting primarily of Wallaby grass (*Austrodanthonia bipartite*), Red grass (*Bothriochloa macra*) and Plains grass (*Austrostipa aristiglumis*)[15] through to improved lucerne, fescue and clover-based pastures. Grazing is accompanied by some opportunistic cropping of wheat, sorghum and other winter cropping options, while forage oats may be cropped to supplement the grazing enterprise. The capability of land in the KPS is shown also in Figure 6.4

The breakdown of area in the KPS by land class is provided in Table 6.10. Detailed definitions of the land classes can be found in Appendix E. Broadly, classes 1-3 are suitable for regular cultivation, classes 4-5 are suitable for grazing and occasional cultivation, class 6 is not suitable for cultivation, class 7 is land best protected by timber and class 8 is unsuitable for agricultural production. In general, soil productivity declines with class.

There are 93,145 hectares of agricultural land in the KPS according to this data (classes 1- 6) and this is set as the total area of the KPS on which benefits of BFT

[14] The Northern New England Rural Lands Protection Board levies 295 properties for the Division (I) which encompasses the KPS and is slightly larger than the KPS. The total area of properties in Division I is 138,997 hectares and an average of 471 hectares are levied land holding (pers comm., Debbie Cuneen, Northern New England RLP Board, May 2007).

[15] There would possibly be 15 to 20 native grass species in the area but these are 3 of the most important species (Pers comm., Bob McGufficke, District Agronomist, Inverell. November 2006).

introduction are calculated. Classes 5 – 8 are used to represent the pasture areas where lucerne is not suitable, class 3 is the area where lucerne would be the preferable improved pasture option and Class 4 where lucerne and BFT could both be good options. On this basis, adoption of lucerne is constrained to 50 percent of suitable agricultural land area in the LP model. Similarly, cropping options are constrained in the LP to 30 percent of the area because classes 1-3 represent in the order of 30 percent of the Subcatchment's agricultural area.

Table 6.10 Land area by capability in the Kings Plain Subcatchment

	All lands[a]	Agriculture[a]	Lucerne[b]	Cropping[b]
Class 1 and 2	5,393	5,393	5,393	5,393
Class 3	24,287	24,287	24,287	24,287
Class 4	17,379	17,379	17,379	
Class 5	9,711	9,711		
Class 6	36,374	36,374		
Class 7	17,971			
Class 8	3,347			
Other (mining, urban areas etc)	1,518			
Hectares	**115,980**	**93,145**	**47,060**	**29,681**
			% of Agricultural lands	
			51	32

[a.] Data supplied by DNR (2006), [b.] Determined on the basis of the land class definitions (see Appendix E (DNR, 2006)).

6.3.1 Values of production

Table 6.11 details the gross margins associated with typical agricultural enterprise options and supporting pasture options within the KPS.

Table 6.11 Values of production for agricultural enterprises in the Kings Plain Subcatchment ($)

Enterprise	Represented by gross margin for:	Revenues	Costs	Gross Margin
Forage oats	Dryland forage oats, North East NSW	0	170	-170/ha
Existing improved pasture	Dryland lucerne[a]	0	35	- 35/ha[b]
Native pasture	Unimproved native pasture.	0	0	0.00/ha
Cropping rotation	Wheat/wheat/sorghum/sorghum/wheat dryland rotation, North East NSW.	537	360	177/ha[c]
Fallow	Cultivation cost.	0	11	- 11/ha
Sheep	21 micron Merino Ewe/Merino Ram flock	73	25	48/hd
Cattle	Growing out early weaned calves 160-340kg	566	105	461/hd

Source: NSW DPI (2005-06). [a.] Dryland lucerne & subclover gross margin for the Southern zone used in the absence of a NSW DPI gross margin for dryland lucerne for the North East. [b]. Establishment costs annualised over 5 years. [c]. Annualised for the 5 year rotation (3 years wheat - 2 years sorghum: 2 years short fallow dryland wheat after wheat ($81/ha), 2 years dryland grain sorghum ($250/ha), 1 year long fallow dryland wheat after sorghum ($220/ha)). The value of those enterprises shown to have a negative gross margin is captured in the transfers (of pasture availability) within the LP model (see Appendix F).

Together with the enterprise options shown in Table 6.11, a high-density tree option is also included in the LP model. A gross margin of -$33 is used for the high-density tree option based on work by Hill (2004) with respect to the costs and benefits of tree planting for salinity control.

6.3.2 Pasture: production and consumption

Table 6.12 summarises the metabolisable energy produced seasonally by the range of livestock feed sources included in Table 6.11. These energy values are comparable to the production estimates for BFT provided in Table 6.2. Also detailed in Table 6.12 are the seasonal feed requirements of the livestock enterprises (consumption).

Tables detailing available dry matter and metabolisable energy produced by month for each pasture can be found in Appendix G, together with the source of the data. A pasture utilisation rate of 40 percent, consistent with that set for BFT pasture (see section 6.1.1) was used for all pasture options.

Table 6.12 Pasture production and consumption values for a representative farm in the Kings Plain Subcatchment

	Production (TME/ha) [a]			Consumption (TME/BU [b])	
	Forage Oats	**Lucerne**	**Native Pasture**	**Sheep [c]**	**Cattle [d]**
Summer	0	24,997	29,700	1,930	15,725
Autumn	22,379	22,718	14,775	1,505	15,004
Winter	36,158	13,426	4,784	1,441	13,674
Spring	36,503	29,377	19,189	2,332	15,458
Annual Total	***95,040***	***90,518***	***68,448***	***7,208***	***59,861***

[a] See Appendix G. [b] Breeding unit, includes the entire number of stock in the enterprise expressed per breeding female. [c] Self-replacing Merino flock breeding unit (Alford *et al.*, 2004). [d] Heavy feeder steer breeding unit equivalent (Alford *et al.*, 2004).

6.3.3 Recharge by land use option

The land use choices in the LP model are constrained by the allowable recharge set by the DP to represent a given state transition (see section 5.4.1). Estimates of recharge for the range of land use options are shown in Table 6.13. The average values for each land use option across soil types have been adopted in this analysis.
High-density trees are assumed to have zero recharge.

Table 6.13 Recharge by land use option for the Kings Plain Subcatchment (mm/m^2)

Soil Type	**Lucerne**	**Wheat**	**Fallow**	**Forage Oats**	**Sorghum**	**Native Pasture**
Deep black cracking clays	0-5	40-60	80+	30-50	20-50	0-10
Deep structured red clay loams	5-10	50-80	100+	40-70	40-60	0-30
Stony sandy loams	10-25	60-100	120+	60-80	50-80	30-60
Yellow & red texture contrast	10-20	50-80	100+	40-70	40-60	20-50
Average	**6-15**	**50-80**	**100+**	**40-70**	**35-60**	**35-45**

Source: Pers comm., Dr Sean Murphy, Farming Systems Hydrologist, Tamworth Agricultural Institute, NSW Department of Primary Industries, December (2006). Reviewed by Ms Carol Harris Senior Research Agronomist, NSW Department of Primary Industries, Centre for Perennial Grazing Systems, Glen Innes and found to be reasonable, December (2006).

6.3.4 Depth of the watertable

The hydrology of the KPS is governed by the interaction of successive volcanic layers, deposited within the Permian (250-300 million years ago) and Tertiary (38 million years ago) periods and interspaced by long periods of landscape erosion. The groundwater systems are localized, low yielding, of moderate quality and relatively recent, recharging deeper aquifers to the west and south whilst discharging into surface flow in the north west. The elevation of Kings Plains within the landscape, and the underlying felsic volcanic material falling to the south west and overlain by more porous and much younger basaltic material, means that the movement of groundwater within the catchment is difficult to predict (pers comm., Blair, 2007; Blair, unpub).

At the southern end of the catchment the water table has been reported to be as close as 60cm to the surface. This depth increases toward the middle of the plain as the alluvial black cracking clays increase in depth over the hardrock felsic[16] deposits of the Emmaville Volcanics. Where this felsic material occurs as bedrock highs, shallow groundwater systems are negligible; however, deeper systems between 2-10 m may occur away from these areas, particularly associated with current drainage and the deeper basalt (pers comm., Blair, 2007; Blair, unpub).

Overall, it is difficult to characterize the depth of the watertable for the entire subcatchment. However, consideration of depths ranging from around 15 m up to 0.5 m below the surface accommodates the reported variance in the subcatchment. Bore data for the Subcatchment (New South Wales Government, 2007) has been sourced to provide a proxy of the range of watertable depths throughout the subcatchment. This is reported with the results in Chapter 7.

Change in the depth of the watertable results not only from recharge entering the system, but also through groundwater that is discharged from the system to rivers and other groundwater systems. Estimates place the discharge from groundwater systems at between 0.5 – 10 mm/m^2 per annum (Stirzaker *et al.*, 2000), with estimates for

[16] **Felsic** is a term used in geology to refer to silicate minerals, magmas, and rocks which are enriched in the lighter elements such as silica, oxygen, aluminium, sodium and potassium.

other locations in northern NSW being at the higher end of this range (Sun and Cornish, 2006) . This parameter has been set at 10mm/m^2 per annum for this analysis but a range of values is tested.

The parameter θ, which translates total net recharge into a change in the watertable depth (see Equation 5.6), has been set at 150 mm/m^2. This value was used by Greiner (1994) and was also adopted by Cacho *et al.*(2001).

6.3.5 The impact of a rising watertable on land-use options

There is currently an estimated 760 ha of salinity-affected land in the KPS (DNR, 2006). That is, less than 1 percent of the subcatchment has identifiable impacts of excess recharge and rising ground water tables. There is the potential for this area to increase with the continuation of land use practices which contribute excess recharge to the ground water tables in the subcatchment.

The impact of a rising watertable will vary with the tolerance of each plant type. The impact of the rising water table is incorporated within the model according to the yield loss function from Cacho *et al.* (2001) as described in Equation 5.11. The variables, β and φ, vary for each of the crop options in the model to reflect the tolerance of each to salinity (Table 6.14). The sensitivity of the results to these factors was tested.

Table 6.14 Yield adjustment factors

	β	φ	**Source**
Wheat rotation	1.326	0.564	Estimates used for Wheat by Cacho *et al.* (2001)
Forage oats	1.193	0.62	Assumed 10% more tolerant than wheat in base analysis
Lucerne pasture	1.193	0.62	Assumed 10% more tolerant than wheat in base analysis
BFT pasture	1.061	0.677	Assumed 20% more tolerant than wheat in base analysis
Native pasture	1.061	0.677	Assumed 20% more tolerant than wheat in base analysis

6.4 Probabilities

To assess the expected values for the costs and benefits of the introduction of birdsfoot trefoil, the probability that they are realised must be considered. These are values for the parameters p_{ri} and p_{wj} in the decision criteria (see Equation 5.1).

The likelihood that the plant invades each area will depend on the probability that plant material or seeds make it to the area as well as the subsequent probability that the plant then establishes itself in the new area. In turn, each of these is dependent on a range of distance and location specific factors. The approach of Glauber and Narrod (2003) (as reviewed in 4.2.1.1) can be applied to this problem as shown below in Equation 6.2 where there are '*n*' factors influencing the establishment of the plant in a new area.

$$p^* = 1 - (1 - p_1)(1 - p_2)(1 - p_2)(1 - p_3).....(1 - p_n) \qquad \textbf{(6.2)}$$

Where (for example) p_1= probability seeds are transported with stock

p_2= probability seeds are transported in the river

p_3= probability seeds settle in a suitable soil

p_4 = probability seeds are not eaten by predators

p_5= probability sufficient rainfall for germination occurs

etc

For any given plant and any given location many factors are likely to influence establishment. It is difficult to obtain the data required to confidently identify each of these factors and therefore it is difficult to obtain reasonable estimates of the probabilities to calculate p_i. While the framework presented by Glauber and Narrod provides a sound basis for considering the factors which will influence the probability of spread, computation of p^* in this way is unlikely to provide a pragmatic addition to

the decision model being established because of the difficulty determining $p_1 ... p_n$. The Delphi technique provides one avenue for the estimation of p^*. The Delphi technique incorporates a 'planned program of sequential, individual interrogations' (Hardaker *et al.*, 2004) which allows, and relies on, anonymity, controlled feedback, reassessment and group response. Rather than the need to discover the probability associated with each and every factor that might be included in a spread probability equation such as 6.2, Delphi offers the opportunity for experts to estimate p^* through their implicit calculation of p_1 through p_n. While the Delphi technique is a potential avenue for investigating probabilities associated with the spread of invasive plants, it does have a high resource input and therefore potentially does not provide the pragmatic approach desired.

It is generally recognised that the likelihood of introduced organisms becoming invasive is low (Caley *et al.*, 2006). For introduced species generally Smith *et al.* (1999) identified the range of 0.01 percent to 17 percent, with a likely value of 2 percent, for the prior probability that a newly introduced plant species would become invasive. This range could be used as the probability that the costs of invasiveness are realised (p^* in Equation 6.2 above). The value of basing predictions of invasiveness on prior invasions when it relates to the same species in a different environment is recognised (Reichard, 2001; Stone *et al.*, 2008). These base rates, are however, in no way specific to the new varieties of birdsfoot trefoil and adopting these parameters would ignore the many factors that contribute to whether birdsfoot trefoil is invasive. Therefore, an approach which includes breakeven analysis together with stochastic parameters is used.

Breakeven analysis is one form of sensitivity test used in benefit-cost analysis (Sinden and Thampapillai, 1995). It provides for a focus on a particular variable by determining the value which results in the net benefit of a project being zero. That is, the point at which the project breaks even. Break-even analysis allows estimation of the required probability that the costs of the plant's introduction (i.e. costs of its invasiveness) are realised to be equal to the benefits of its introduction (i.e. benefits of salinity reduction). As such, it considers the question of likelihood of invasion from an alternate view – is a breakeven probability of p_w a reasonable expectation? Or, is it reasonable to expect that a particular number of seeds will be dispersed into external

areas? In this analysis, breakeven analysis is applied to a range of factors, but in particular to the number of seeds dispersed into external areas. This is complemented by the incorporation of stochastic demographic factors.

6.5 Summary

A range of data sources has been used to populate the model developed for the analysis. Sourcing the data has illustrated difficulties in adopting the proposed decision framework for new plant introductions. That is, in many cases data are not readily available. For the purpose of this analysis, a range of methods has been used to circumvent the data gaps, including adopting alternative and proxy data, expert opinion, implementing breakeven analysis and incorporating stochastic variables which describe the range in which unknown variables are likely to fall.

The results of the analysis using this data set are now presented in Chapters 7 and 8.

7 Net benefits of the introduction of birdsfoot trefoil: deterministic analysis

The results of a deterministic analysis of the impacts of the introduction of birdsfoot trefoil to the Kings Plain Subcatchment (KPS) and potential invasion of the surrounding Border Rivers Catchment (BRC) are presented in this chapter.

Estimated benefits within the KPS, from solution of the salinity dynamic programming (DP) sub-model are presented first. This is followed by the estimated costs in the surrounding BRC, found from solution of the plant simulation sub-model. The results of the two sub-models are then brought together to consider the net benefits of the introduction of birdsfoot trefoil.

Results have been obtained for the parameters presented in Chapter 6, with a discount rate of 7 percent consistent with recommendations of NSW Treasury (1999).

7.1 Reduced salinity in the Kings Plain Subcatchment

7.1.1 Benefits for a range of watertable depths

Analysis presented in this section shows that there are benefits to be gained from the introduction of birdsfoot trefoil (BFT), and that benefits will vary considerably according to the initial state of the watertable.

The present values of farm profits over 50 years, with and without BFT for watertable depths ranging from 0.5 to 15 m below the surface are shown in Figure 7.1 on the following page. The results were generated by solving the DP model with BFT and without BFT as a land use option in the LP. These results reflect the lower farm productivity where the watertable is close to the surface and salinisation adversely impacts plant growth. This impact is evident irrespective of birdsfoot trefoil's introduction.

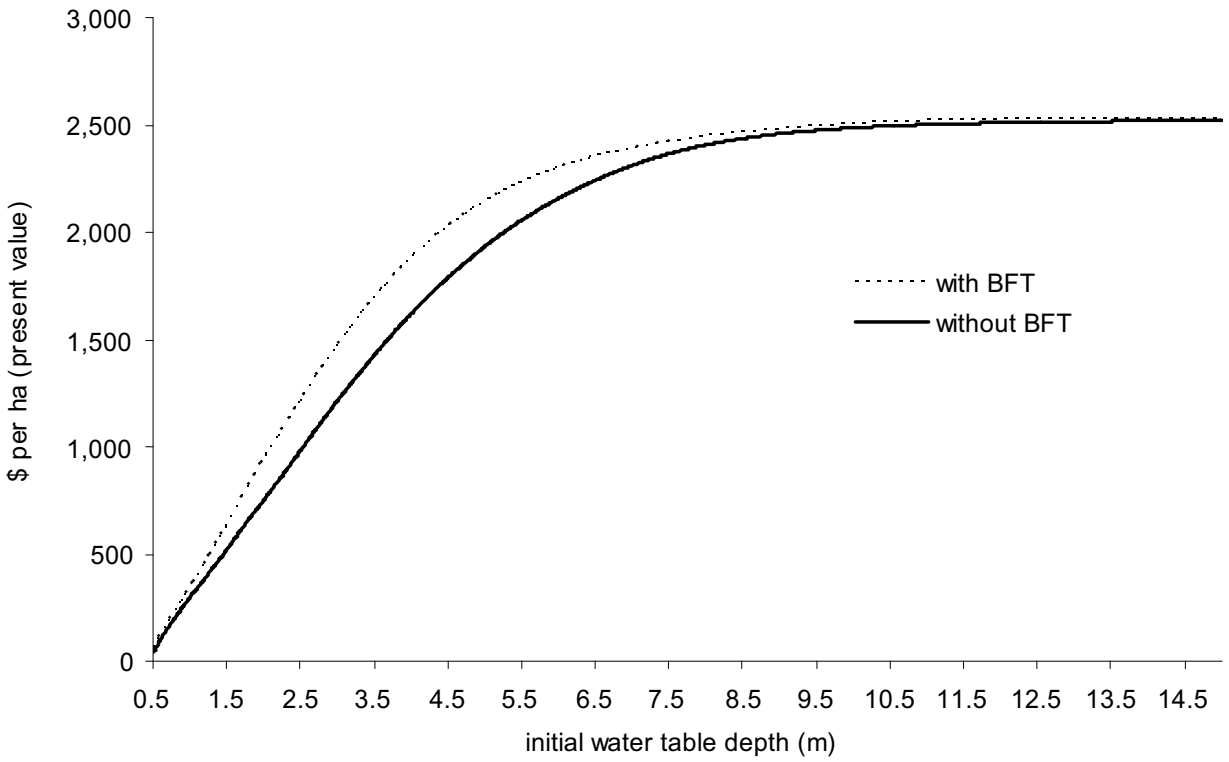


Figure 7.1 Present value of profits under optimal management with and without BFT for a range of initial watertable depths ($/ha, discounted at 7 per cent over 50 years)

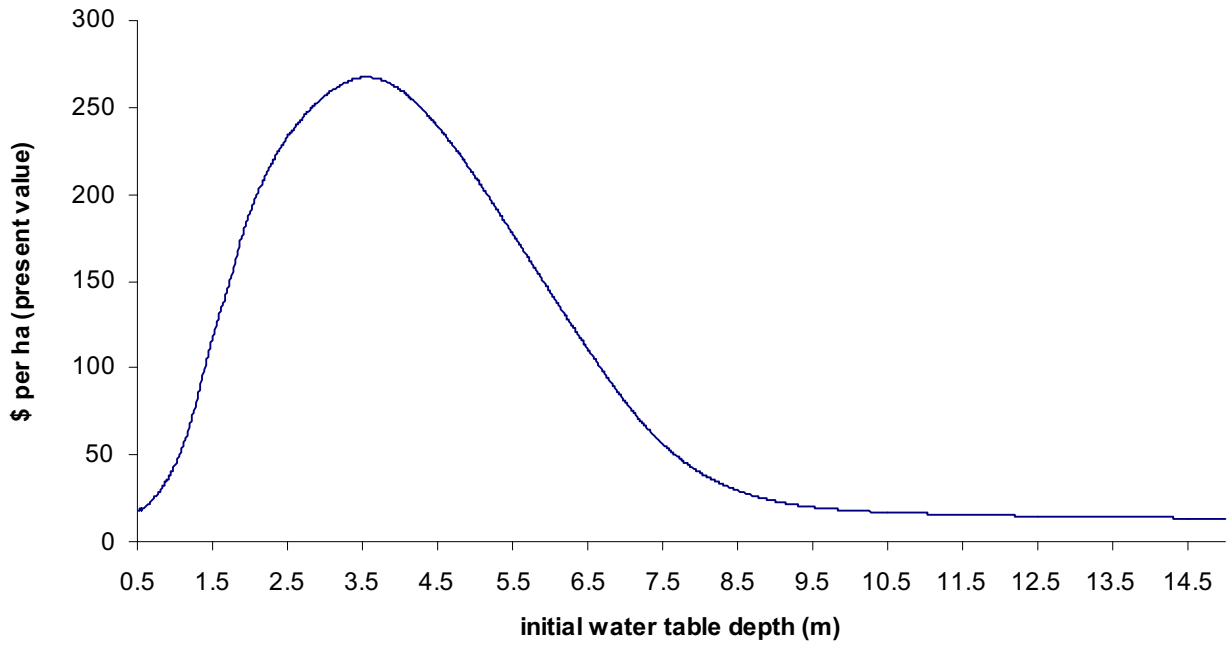


Figure 7.2 Present value of benefits from the introduction of BFT for a range of initial watertable depths ($/ha, discounted at 7 per cent over 50 years)

The benefit of the introduction of BFT is estimated as the net difference in farm profit with and without the plant[17]. The present values of benefits from the introduction of BFT are illustrated in Figure 7.2. The benefits increase to a maximum of $270 per hectare, where the initial watertable depth is 3.5 m, and then decrease as the initial watertable depth increases further.

Mullen (2001) has highlighted the generation of shadow prices as a particular advantage of optimisation techniques applied to land degradation analyses. The shadow price of a resource is the marginal increase in the benefit per unit increase in the resource. In the absence of an established market for the resource, the imputed shadow price is an indicator of its marginal value (Chiang, 1984). The shadow price of a resource increases as the resource becomes scarce and is zero when the resource is not a limiting factor of production. Here, the shadow price of a metre increase in the watertable depth is estimated as the marginal change in the present value of farm profits for a one metre change in the watertable depth, all else constant (i.e. the slope of Figure 7.1). The shadow price of the watertable is shown in Figure 7.3.

The shadow price varies with depth and is highest at watertable depths between 2 and 4 m. Figure 7.3 also shows that the introduction of BFT increases the value of a change in the watertable at lesser depths: birdsfoot trefoil provides an option for increasing the productive capacity of the land at lesser depths such that a one metre increase in the depth of the watertable has a greater net impact on the present value of farm profits than it would have in the absence of birdsfoot trefoil.

Beyond 4 m in depth, increases in watertable depth are less valuable with BFT than without. As the watertable approaches 15 m, the value of watertable reductions approaches zero. This indicates that an initial watertable depth of 15 m does not place a constraint on profits over a 50-year planning horizon, independently of whether BFT is introduced.

[17] 'Benefits' rather than 'net benefits' is used in this instance to limit confusion that might be created when 'net benefits' are later discussed in reference to the overall benefits from the plant's introduction net of the costs of the plant's invasiveness.

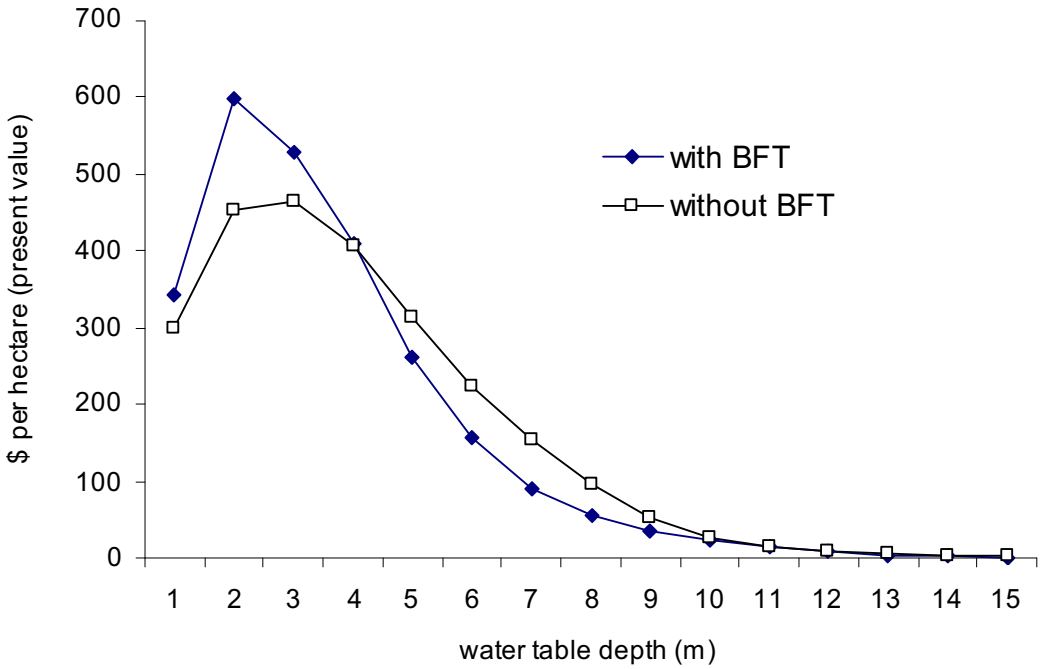


Figure 7.3 Shadow price of the watertable with and without birdsfoot trefoil ($/ha)

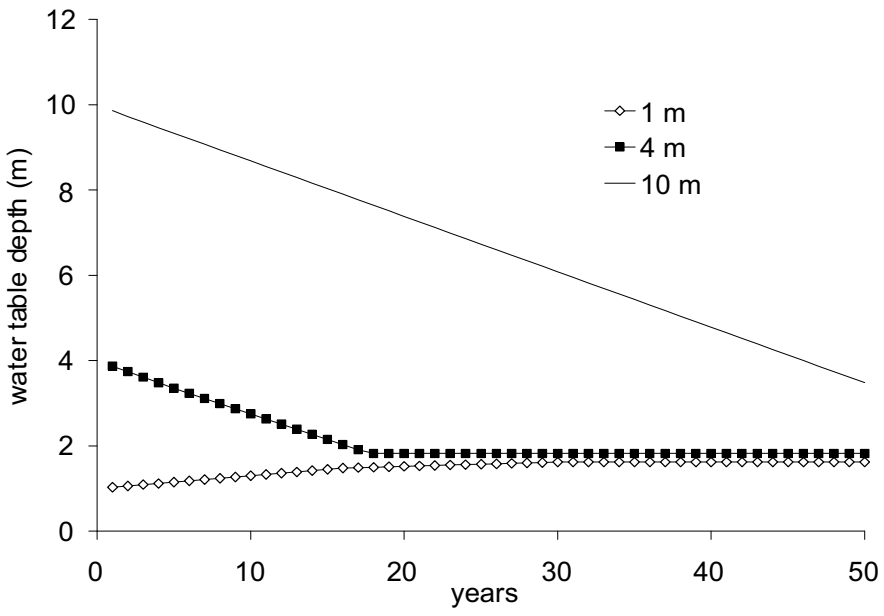
In the KPS where the average watertable is likely to be between 3 – 6 metres (see Chapter 6), the marginal value of a metre reduction in the watertable is likely to range from around $200 - $450 per ha without birdsfoot trefoil and from around $150 - $600 per ha with birdsfoot trefoil. While these values are relatively small as present values over 50 years, it could be argued that these marginal values illustrate that at some initial watertable depths, farmers as individual decision makers may have less incentive to preserve the watertable with a new plant such as BFT than without it. This however relies on farmers being perfectly informed about the depth of their watertable across the farm, which is unlikely given the uncertainty related to watertables and their management.

The analysis in Figures 7.1 to 7.3 shows that there are benefits from the introduction of BFT, and that these benefits will vary with the initial watertable depth. More detailed results are now presented for three initial watertable depths to illustrate the optimal decision rules and associated optimal state paths.

7.1.2 Optimal state paths

Initial watertable depths of 1 m, 4 m and 10 m below the surface have been selected to illustrate the optimal state paths and optimal decision rules. The optimal state paths identified for both cases without BFT and with BFT are shown in Figure 7.4.

(a) without birdsfoot trefoil



(b) with birdsfoot trefoil

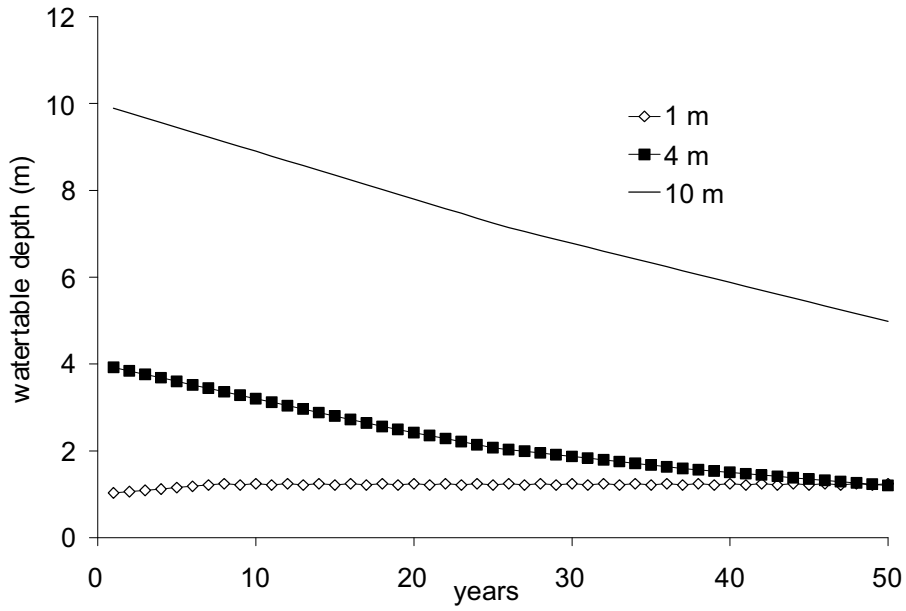


Figure 7.4 Optimal state paths for three initial watertable depths

Figure 7.4 (a) shows that, from an initial watertable depth of 1 m, the optimal strategy is to increase the depth of the watertable such that it reaches a steady state of 1.6 m in year 30, while from an initial watertable depth of 4 m a steady state of 1.8 m is reached rapidly in year 18. From an initial depth of 10 m, the state path reaches 4 m after 50 years.

Figure 7.4 (b) shows that from an initial watertable depth of 1 m, the optimal strategy with BFT is also to increase the watertable depth. The optimal state path reaches an optimal steady state that oscillates around 1.21 – 1.24 m after year 7. From initial depths of 4 and 10 m, a steady state is not reached in the analysis period. The states in year 50 are 1.2 m and 6.0 m respectively.

These results indicate that, where the initial watertable depth is greater than 2 m, it is optimal to decrease its depth over time and for initial depths less than 2 m it is optimal to increase the depth over time. These results hold whether BFT is introduced or not. The overall effect of birdsfoot trefoil from any given initial watertable depth is in the timing of changes to the state path as well as the steady state reached. That is, the benefits of the introduction of BFT arise from a combination of impacts.

Firstly, there is an issue of timing. The introduction of BFT decreases the rate at which the steady state is approached. Secondly, there is a difference in the steady states reached. The introduction of BFT allows optimal steady states where the watertable is closer to the surface. This reflects that, as a pasture, BFT both delays rising watertables and allows more profitable management when watertables are closer to the surface.

7.1.3 Optimal enterprise mix

The optimal state paths are estimated on the basis of the available land-use options and the constraints placed on the LP model, which is controlled through the DP model. The land-use mix associated with an initial watertable depth of 4 m and without birdsfoot trefoil is shown in Figure 7.5.

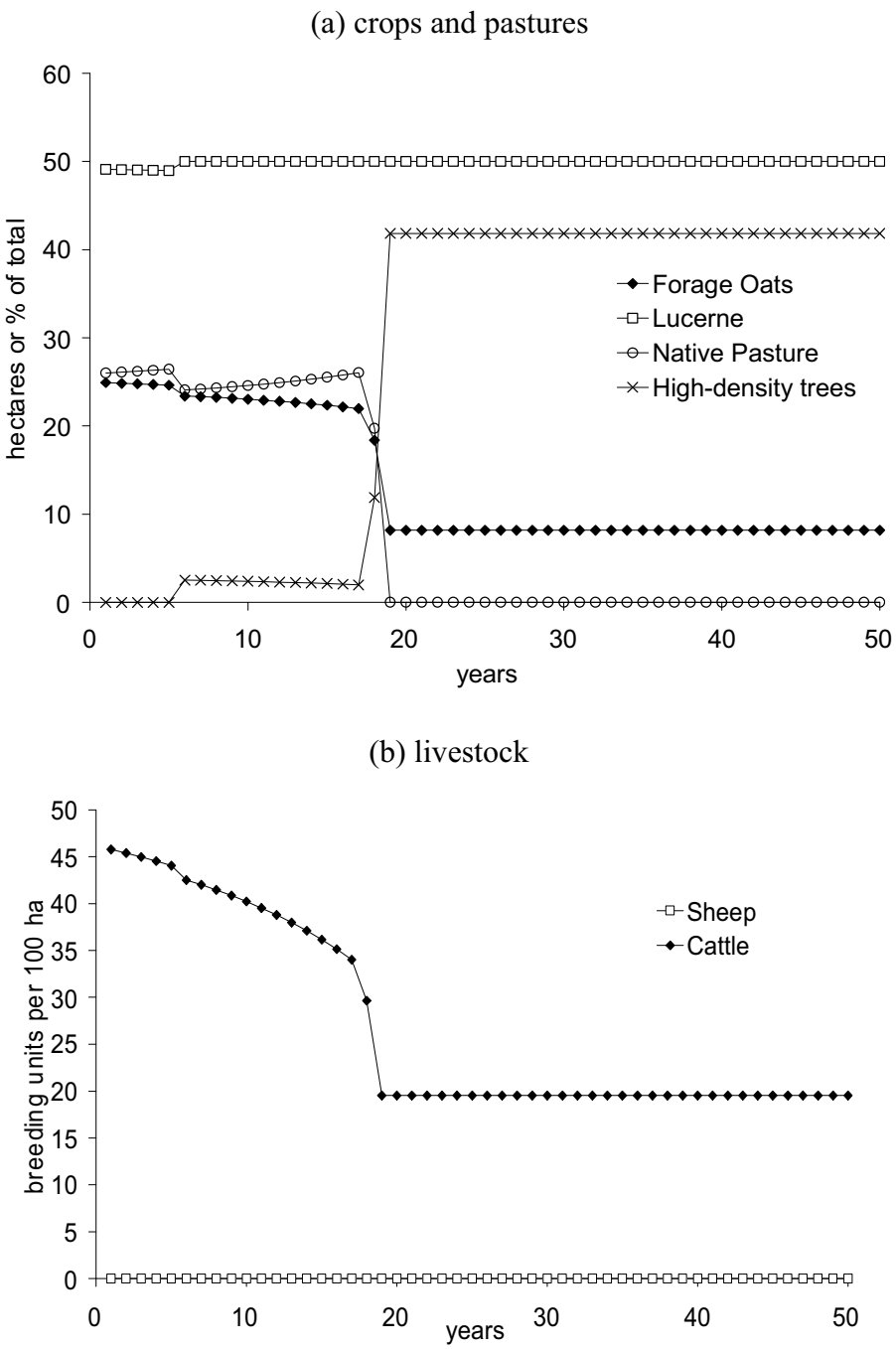


Figure 7.5 Optimal enterprise mix for an initial watertable depth of 4 m – *without birdsfoot trefoil*

Figure 7.5 (a) illustrates a high (close to 50 percent) reliance on lucerne for the entire planning period, accompanied by a varying mix of high-density trees, native pastures and forage oats in the early stages of the planning horizon. From year 19, the optimal mix settles to just over 40 percent high-density trees and just below 10 percent forage oats. The optimal solution for an initial depth of 4 m, without BFT, includes cattle as

the only livestock as shown in Figure 7.5 (b). Consistent with the optimal state path and the optimal land use mix, the figure shows that there is an initially high number of cattle breeding units (46 per 100 ha) in the optimal solution and that this declines to a steady state of 20 breeding units per 100 ha by year 19. The decline in carrying capacity occurs because the productivity of pasture options falls as the watertable rises, and because land-use options with lower recharge but less pasture value, such as high-density trees, are selected to mitigate further increases in the watertable.

Comparable results for the case where birdsfoot trefoil is introduced are shown in Figures 7.6 (a) and (b) on the following page. Shown in Figure 7.6 (a) is the same reliance on lucerne. Birdsfoot trefoil enters the solution for just over 30 percent of the area and increases to between 40 and 50 percent later in the planning period. The increasing use of birdsfoot trefoil comes from the substitution of forage oats which enters the solution at just under 20 percent up until year 18 and then decrease to eventually exit the solution in year 38. The exit of forage oats coincides with the introduction of 5 ha of high-density trees.

Figure 7.6 (b) shows that with BFT again the optimal solution includes cattle as the only livestock enterprise, and again the stocking rate decreases over the planning period in line with the falling productivity of the land as the watertable rises. The decline in stocking rate with BFT is more gradual but falls to 18 breeding units by year 50, compared to 20 without birdsfoot trefoil, in year 50.

Now that the benefits associated with the optimal state paths for a range of initial watertable depths have been estimated for a farm representative of those in the KPS, the total benefits for the entire Subcatchment are considered next.

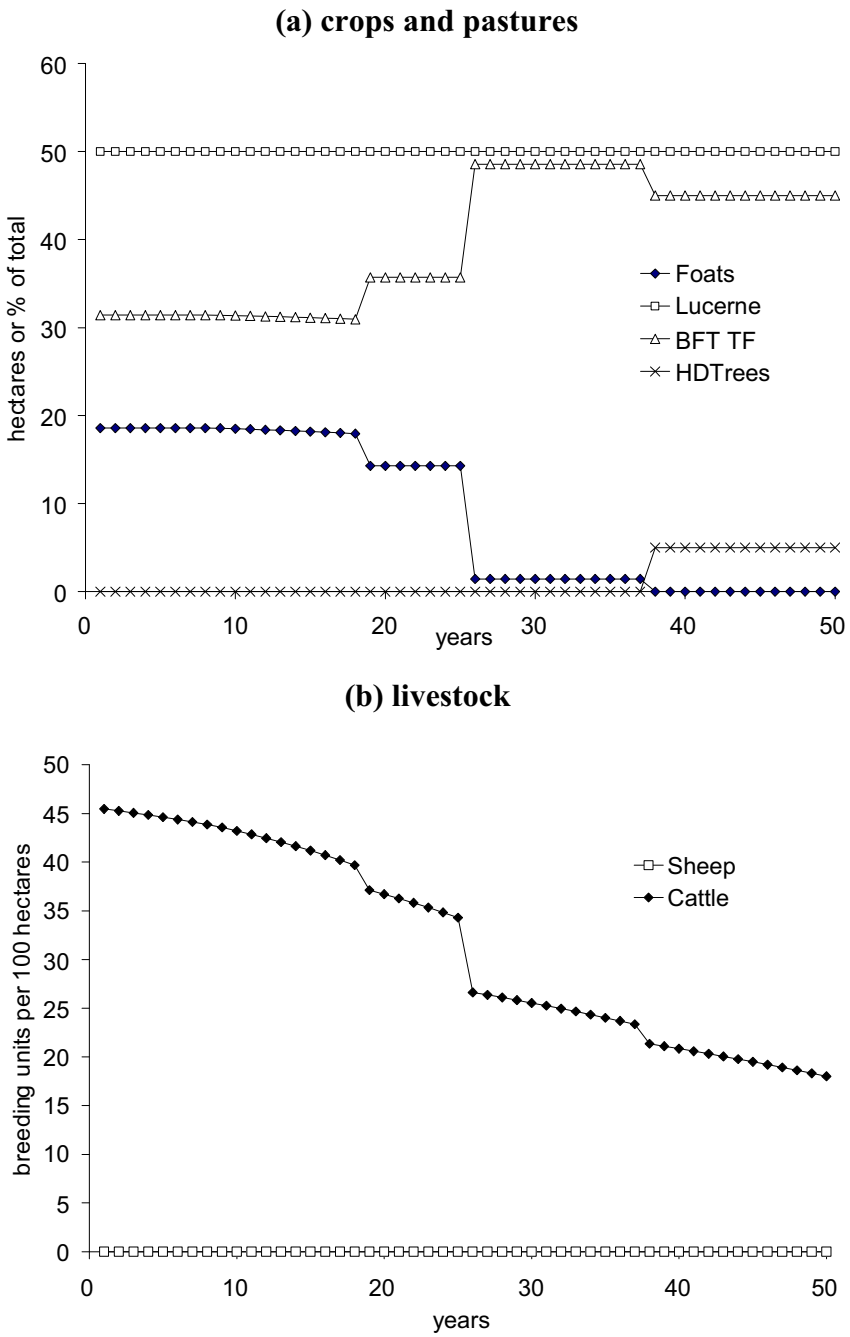


Figure 7.6 Optimal land use for an initial watertable depth of 4 m – *with birdsfoot trefoil*

7.1.4 Subcatchment benefits of the introduction of BFT

The KPS has 93,145 hectares of agricultural land (DNR, 2006). Extrapolating the per-hectare benefits of the introduction of BFT across all agricultural lands in the KPS provides an estimate of the subcatchment benefits from the introduction of BFT

(Figure 7.7). The subcatchment benefits have the same functional form as the benefits on a per-hectare basis (see Figure 7.1) where the subcatchment is assumed to have a homogeneous watertable depth throughout. The benefit for the range of initial watertable extends from $1.6 million at 0.5 m up to $24.9 million at 3.5 m and then down to $1.2 million at 15 m.

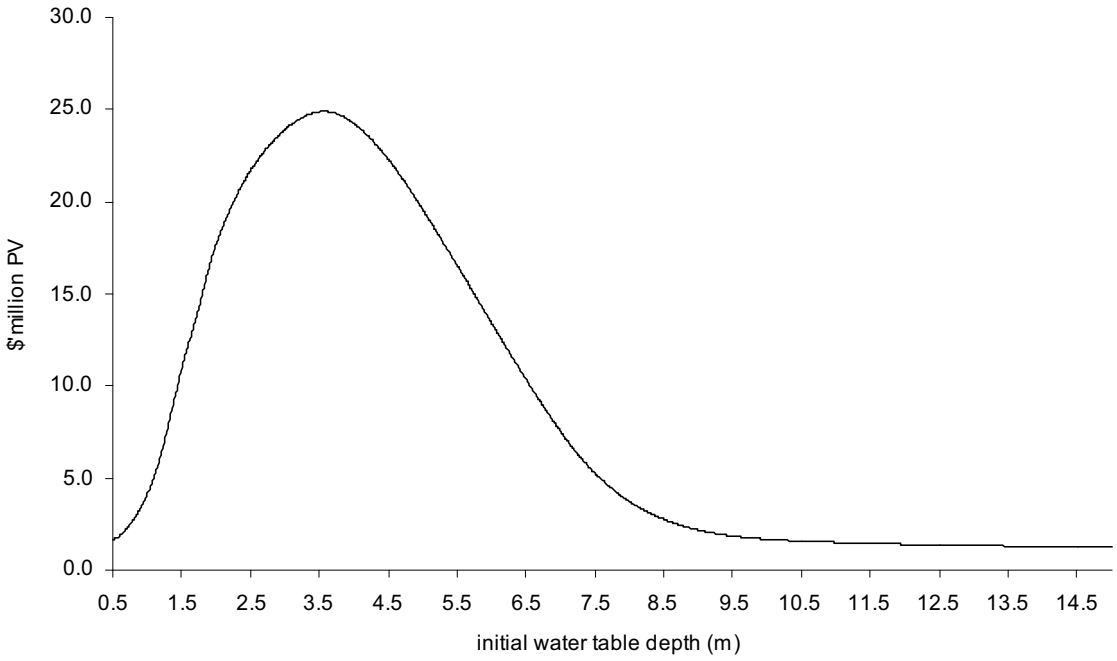


Figure 7.7 Subcatchment benefits of birdsfoot trefoil introduction over 50 years – *for a range of depths (assuming a homogeneous depth throughout the catchment)*

Watertable depths generally vary considerably throughout subcatchments and catchments and this applies to the KPS, where the watertable is reported to range from 0.6 m to more than 30 m (pers comm., Blair, 2007; Blair, unpub). Therefore the assumption of a homogeneous watertable depth in the subcatchment must be relaxed.

Data on the spatial variation in depth are required to estimate the benefits of BFT introduction to the Kings Plain Subcatchment. There is no such comprehensive data set for the KPS so a sample of the depths of the standing water level (SWL) reported for bores across the KPS has been used as a proxy to estimate the indicative depth of the watertable throughout the catchment. Bore data have been sourced from the NSW

Natural Resource Atlas (New South Wales Government, 2007). Table 7.1 shows the benefits from birdsfoot trefoil introduction in the KPS on the basis of the watertable depth throughout the subcatchment as estimated using these bore data. On this basis benefits are estimated to be in the order of $8.8 million in present value over 50 years.

Table 7.1 Total benefits of birdsfoot trefoil in the KPS

Depth (m below the surface)	**Estimated proportion of the catchment (%) (a)**	**Benefit ($/ha) (b)**	**Weighted average benefit ($) (c)**
1	5	45	207,701
2	5	191	887,310
3	15	257	3,589,339
4	2	260	403,799
5	3	210	652,011
6	12	144	1,562,492
7	10	81	751,446
8	2	40	61,878
9	5	23	108,759
10	0	18	-
11	0	16	-
12	7	15	93,825
13	0	14	-
14	2	14	21,503
15 & greater	33	14	421,604
TOTAL			**$8.8 million**

(a) From a total of 99 bores records (New South Wales Government, 2007), 60 included Standing Water Level (SWL) data. Five percent were found to have SWL depths from 1 -2 m, 5 percent between 2 -3 m, 15 percent between 3-4 m and so on. This has been used a proxy for the depth of the water table in the subcatchment. Care should be taken in the interpretation however as i) the bores are located throughout the subcatchment but not all locations are represented, ii) some records are dated such that water tables may have risen since records were taken, and iii) most records pre-date the current drought such that levels may be lower than recorded.
(b) See analysis results shown in Figure 7.2
(c) The weighted average benefit is estimated as (a) multiplied by the total size of KPS (93,145 ha), and then multiplied by (b).

When considering the total benefits for the Subcatchment, it is necessary to consider the implications of the optimal decision rules for the entire catchment. For an average

watertable depth of 4 m for example, the optimal land use for the subcatchment (based on the indicative average land use decision rules over the planning period) includes:

- 8,350 ha of forage oats;
- 46,573 ha of lucerne;
- 36,913 ha of birdsfoot trefoil/tall fescue pasture;
- 98 ha of native pasture; and
- 1,211 ha of high-density trees.

This extrapolation brings with it the need to consider the feasibility of the optimal decisions identified.

7.1.5 Feasibility of the optimal solution

In particular, it is pertinent to consider three questions: the feasibility that half of the subcatchment will be sown to lucerne; whether 36,000 hectares of BFT would achieve the change in recharge required to limit salinity; and whether it is realistic to reduce the area of native pasture to 98 ha considering the potential loss of biodiversity?

The model has been constrained to limit the proportion of farming land at which lucerne can enter the optimal solution. This limit of 50 percent provides recognition of the key limitation of lucerne as a low-recharge pasture option: its unsuitability to soils with low fertility. The optimal solution at the steady state maximises lucerne use subject to this constraint and so reflects the superior characteristics of lucerne in the areas to which it is suited. The KPS is primarily a grazing area so a high proportion of pasture as part of the optimal solution is not unexpected from a cropping-pasture mix perspective. It is unlikely however that lucerne would ever be planted on 50 percent of the land in the subcatchment, because in some locations its salinity-reduction benefits would be a minor consideration from the point of view of private landholders who plant lucerne as a feed source. Broadly, however, the feasibility of lucerne being grown on half of the Subcatchment is not a crucial point for this analysis, because the benefits are assessed as the difference in financial returns for optimal decision rules with and without birdsfoot trefoil. Without

birdsfoot trefoil, lucerne still enters the optimal decision rule for half of the agricultural land.

In the steady state, the optimal decision rule identifies that 38 percent of the land area be planted to the new pasture birdsfoot trefoil. In practice, landholders may or may not adopt BFT on 38 percent of their land. There has been some concern regarding the required scale and location of tree and pasture plantings to achieve changes in watertable depths (George *et al.*, 1999). This analysis assumes the optimal area will be sufficient in scale, and appropriately located, to achieve the changes represented in the model. The analysis is undertaken with the same assumptions with and without birdsfoot trefoil such that the benefits are measured on the ability of birdsfoot trefoil relative to other land uses.

Is it feasible to have only 98 hectares of native pasture? This result should be viewed in the context of the entire subcatchment. Only agricultural areas have been modelled in this analysis, such that more than 22,000 ha of natural grassland and forest of the subcatchment are not included in land use change prescribed by the model. Additional consideration of the role for government in encouraging adoption would, however, be required in the instance that native pastures were displaced, especially with respect to state legislation (e.g. NSW Native Vegetation legislation).

Finally, and most importantly, it needs to be noted that the optimal decision rules for a depth of 4 m (i.e. those discussed above), or any particular depth, will not apply throughout a subcatchment, unless the watertable is at the same depth throughout. For the KPS, and subcatchments in general, the watertable depth varies (See Table 7.1) such that a different set of decision rules will apply for each water table depth. Therefore, over an entire subcatchment the optimal decision rules will include a large range of land-use combinations that reflect the nature of the specific subcatchment.

Sensitivity of the present value of benefits of BFT introduction to plant, environment and market factors is now presented.

7.1.6 Effect of changes in the technical parameters on the optimal solution

The sensitivity of the optimal solution to changes in a range of technical parameters has been tested. These parameters include those that describe BFT (pasture production, recharge), the sensitivity of land use options to salinity (β, φ) and the hydrology of the catchment (discharge). These parameters have been tested to assess the stability of the benefits estimated for the Kings Plain Subcatchment and also to identify how benefits might change when considered in the context of other subcatchments, or the characteristics of other plants that might be introduced.

Figure 7.8 summarises the results of sensitivity analysis. The optimal solution, the highest net present value of benefits for a given watertable depth, is most sensitive to the amount of pasture produced by birdsfoot trefoil and the amount of recharge that the pasture plant allows past the root zone. The benefits were especially responsive for initial watertable depths in the vicinity of 3.5 m. As the amount of recharge falls (Figure 7.8 b) and pasture produced increases (Figure 7.8 c), benefits increase markedly.

The elasticities of the optimal solution where the benefits are highest ($w_0 = 3.5$) were estimated for the two parameters found to be the most influential (Table 7.2). The estimated benefits were found to be most sensitive to the amount of pasture produced by the plant, with elasticities greater than 2. In particular, the solution is sensitive to pasture production when the capacity of the plant to limit recharge is lower (ie, recharge to the watertable increases). This might be an important consideration when introducing birdsfoot trefoil to a catchment with lighter soils (e.g. sandy soils) where recharge is likely to be higher on a mm/m^2 basis. In this situation the benefit of the plant's introduction could be significantly lower. Alternatively, if plant breeders are able to develop a similar pasture plant with 10 percent higher pasture production and 10 percent lower recharge, benefits for an equivalent catchment might exceed $33 million over 50 years in the KPS.

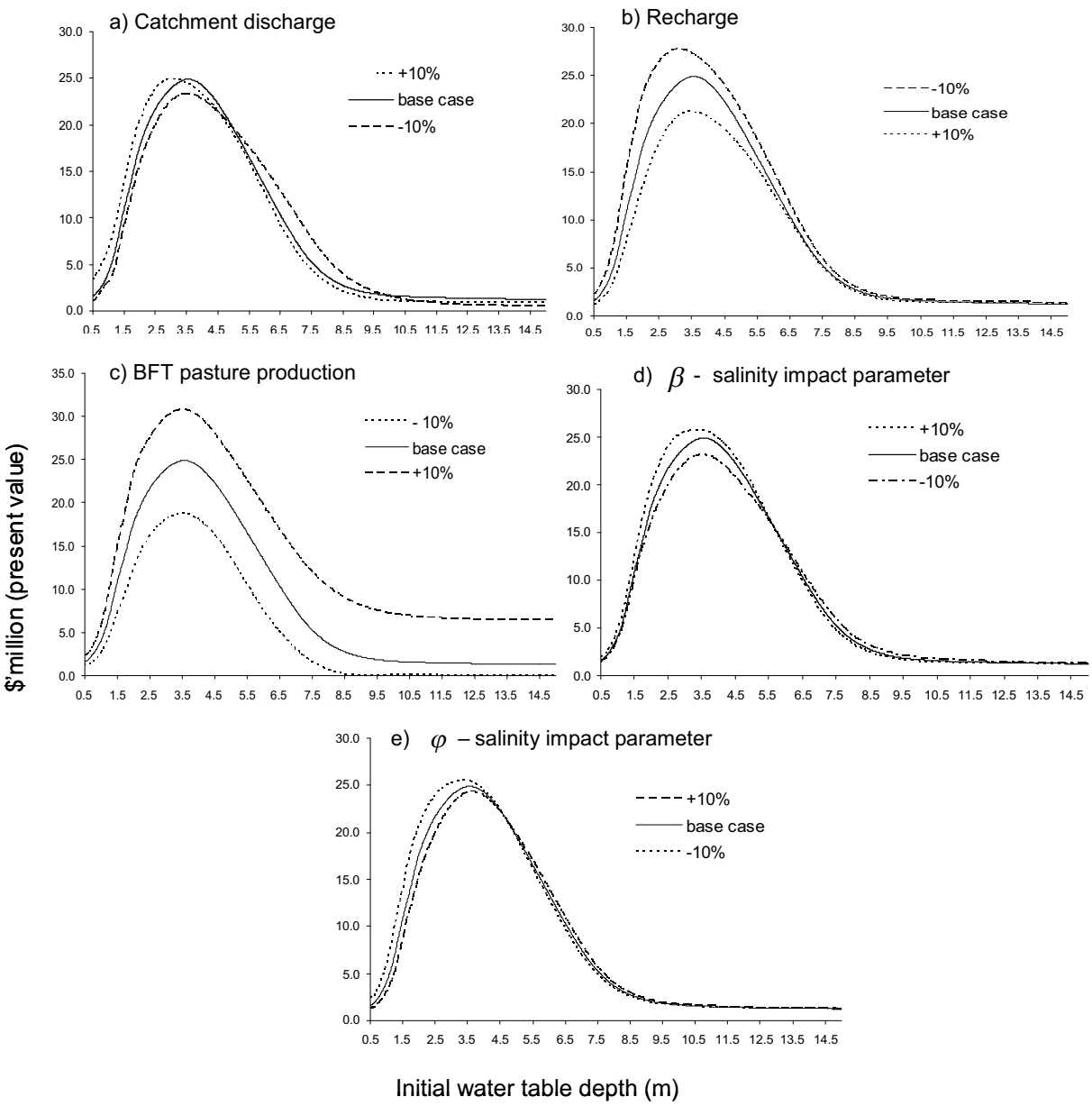


Figure 7.8 Effect of changes in technical parameters on estimated benefits

The sensitivity of the optimal solution to the amount of discharge drained from the watertable was also tested, because this value is difficult to obtain and is not known for the Kings Plain Subcatchment. It is also a value that is likely to vary significantly between catchments. The results of sensitivity analysis, shown in Table 7.3, illustrate that the elasticity with respect to the drainage parameter varies significantly with the initial watertable depth. When the watertable is shallow, the estimated present value of benefits is very sensitive to changes in the estimated discharge from the subcatchment (elasticity = 4.3 at w_0=1), while at depths around 3 - 4 m, the solution is

insensitive (elasticity between 0.11 and 0.22). This illustrates the relative importance of drainage from the catchment within the context of net changes to the watertable (see Equations 5.5 and 5.6). When the initial watertable approaches depths around 10 m and deeper, the optimal solution is inversely related to the discharge from the catchment. This reflects that when discharge rates are high and the watertable deep, the benefits of introducing birdsfoot trefoil are diminished.

Table 7.2 Elasticities of the optimal solution with respect to birdsfoot trefoil recharge and birdsfoot trefoil pasture production *(w_0=3.5)*

	Recharge (mm/m^2)			
Pasture Production	10 % Lower	Base	10% Higher	Elasticity$_{(Rech)}$
	Benefits ($'million PV)			
Low (-10%)	*21.41*	*18.70*	*15.80*	-1.50
Base	*27.21*	***24.90***	*21.24*	-1.20
High (+10%)	*33.16*	*30.78*	*26.85*	-1.03
Elasticity$_{(PProd)}$	2.16	2.43	2.60	

Table 7.3 Effect of subcatchment discharge on optimal solution

	Initial watertable depth (m)			
Subcatchment discharge (mm/m^2)	1	3.5	4	10
	Benefits ($'million PV)			
8	3.07	23.31	22.73	1.55
10	4.14	**24.90**	24.23	1.68
12	6.65	24.4	23.27	1.09
Elasticity	4.32	0.22	0.11	-1.36

Results indicate that discovery or development of profitable pasture plants that might make no, or a negative, contribution to the watertable, rather than just reduce existing contributions would be a superior option. Figure 7.9 illustrates the benefits when the

recharge of birdsfoot trefoil is reduced to 1mm/m^2 (from 20mm/m^2 in the base analysis) such that the net recharge of one hectare of BFT is -9 mm/m^2 (i.e. net of the 10mm/m^2 discharge that is drained from the groundwater system). The benefits increase significantly and, in particular, there are significant benefits to be obtained in areas at greater risk of salinisation (i.e. where the watertable is closer to the surface initially).

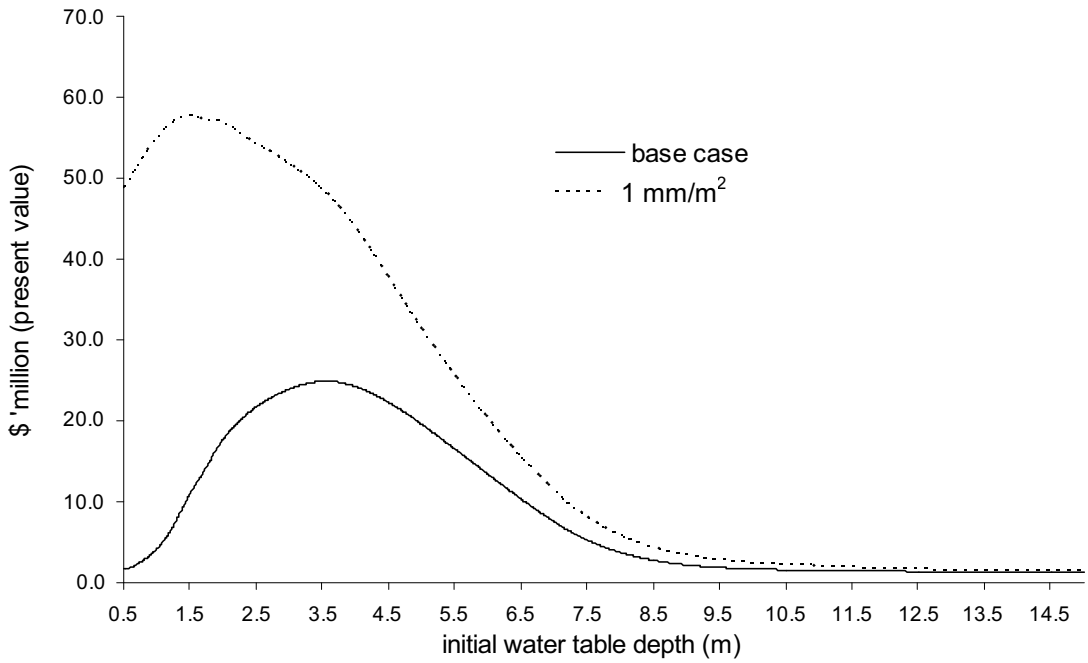


Figure 7.9 Benefits of birdsfoot trefoil – *the case of BFT with recharge of 1mm/m^2*

According to these results, a pasture plant with recharge of just 1mm/m^2 has the potential to contribute benefits in the order of $48 million over 50 years where the initial watertable depth is 3.5 m, and be as high as $57.5 million where the initial watertable depth is 1.5 m. This demonstrates the value to be obtained from research to further reduce the recharge of potential new pasture options.

Figure 7.10 illustrates the optimal state and decision paths that underlie the benefits for an initial watertable depth of 1.5 m. Where pasture plant recharge is sufficiently low to result in a negative contribution to the watertable, there is the opportunity for the optimal state path to move to a greater depth and thus over the planning period lead to higher stock carrying capacities. This optimal state path does require 100 percent of the area to be devoted to BFT for the first 6 years of the planning horizon, a

condition that may not be feasible when considering an entire subcatchment (i.e. 93,145 hectares). This analysis does, however, illustrate the type of benefits that could be achieved by researchers with an aim to identify and breed new plants for introduction with even lower recharge.

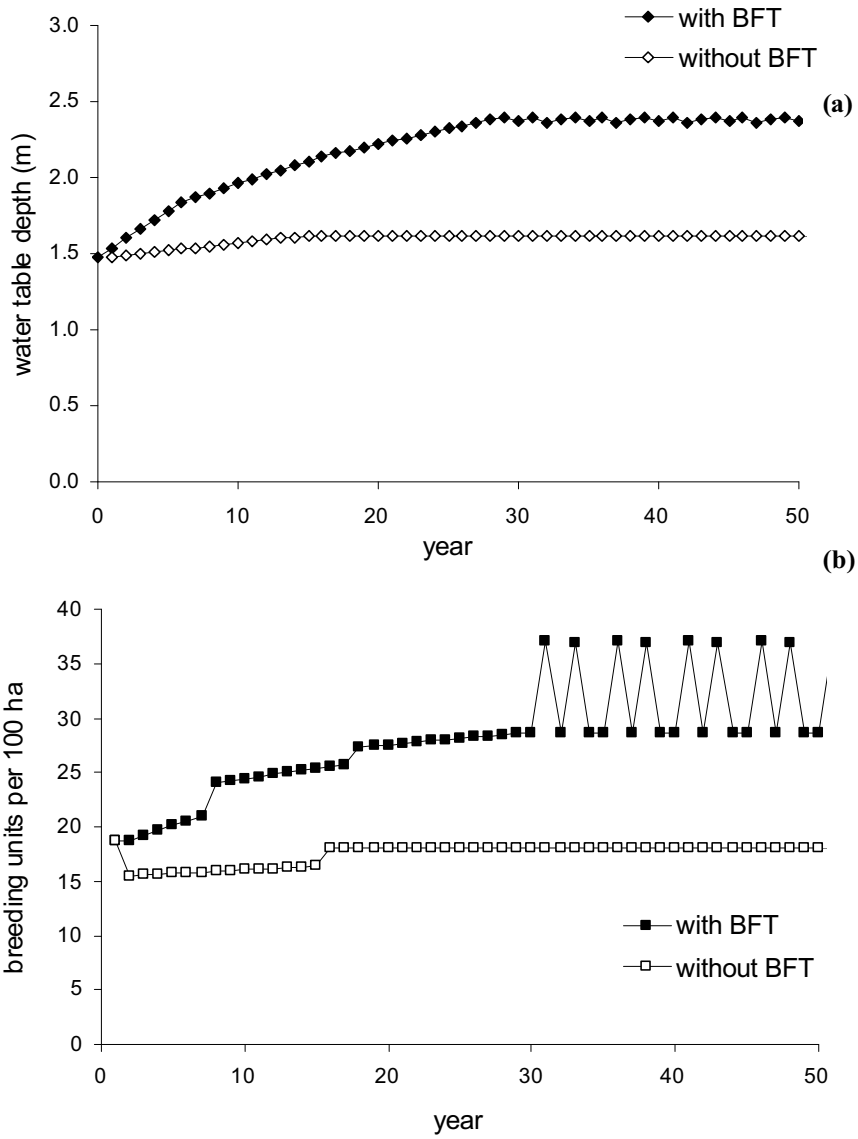


Figure 7.10 Optimal state path (a) and carrying capacity (b) for an initial state of 1.5 m – *the case where BFT has recharge of 1mm/m^2*

7.1.7 Effect of changes to market factors on the optimal solution

This section summarises the results of sensitivity analysis on market factors, including the cost of establishing birdsfoot trefoil, lucerne and high density trees, and the returns from cropping and livestock production as well as the discount rate (Figure 7.11).

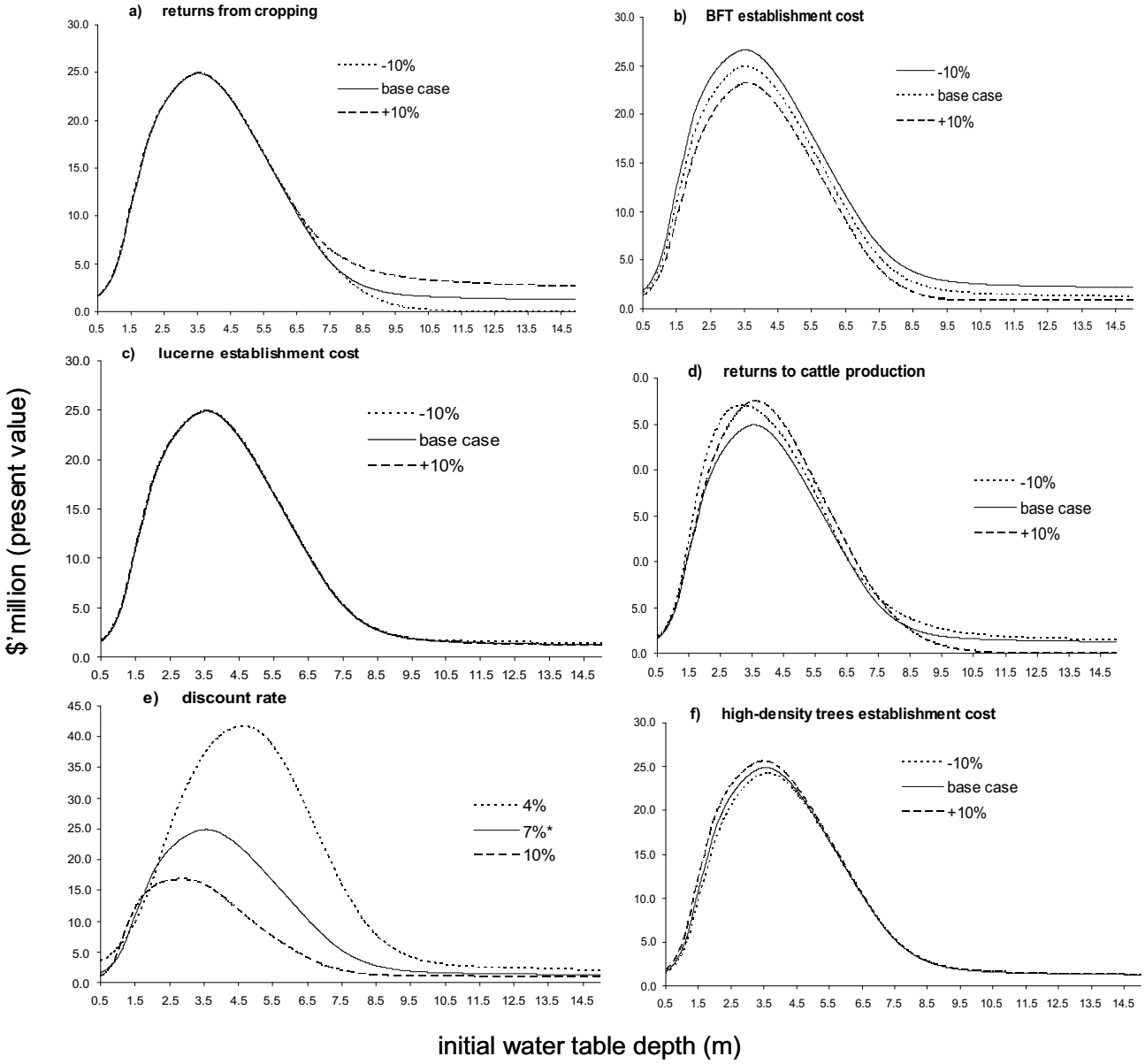


Figure 7.11 Effect of changes in market factors on benefits of birdsfoot trefoil introduction for the range of initial watertable depths

In general, the value of BFT appears to not be sensitive to market prices (Figure 7.11). Analysis of the elasticity of the solution to the cost of BFT establishment (Table 7.4) however, shows that benefits are sensitive to this cost at certain values of initial depths in particular. This reflects that at deeper initial depths there is less incentive to adopt BFT and as such a small change in cost can lead to birdsfoot trefoil moving in and out of the solution. The estimated cost of establishing birdsfoot trefoil has been annualised over five years. Anecdotal evidence suggests that a stand of the new varieties of birdfoot trefoil may only require replacement after 10 years such that annualisation over 10 years might be appropriate (pers comm., Ayres and Lane, 2007). In this instance, the benefit of BFT introduction would be higher at deeper initial depths than estimated in the base case because establishment costs are spread over a longer period.

Table 7.4 Effect of cost of establishing birdsfoot trefoil on the optimal solution

	Initial watertable depth (m)			
Cost of BFT establishment ($/ha)	1	3.5	4	10
	Benefits ($'million PV)			
-10%	*4.91*	*26.63*	*25.86*	2.67
Base	*4.15*	***24.9***	*24.23*	1.68
+10%	*3.45*	*23.18*	*22.6*	0.86
Elasticity	-1.76	-0.69	-0.67	-5.39

Table 7.5 Effect of the discount rate on the optimal solution

	Initial watertable depth (m)			
Discount rate (%)	1	3.55	4	10
	Benefits ($'million PV)			
4	*5.64*	*37.43*	*40.22*	3.32
7	*4.15*	***24.90***	*24.23*	1.68
10	*4.78*	*15.79*	*14.10*	1.02
Elasticity	-0.24	-1.01	-1.26	-1.60

Changes in the discount rate have been shown to have a significant impact on the optimal solution across the range of watertable depths. The size of benefits at discount rates of 4 percent and 10 percent are shown in Figure 7.11 (e) compared to the base results estimated with a discount rate of 7 percent.

The benefits of birdsfoot trefoil are shown to be significantly higher when a lower discount rate is applied. A lower rate, such as 4 percent reflects a higher preference for future benefits and as such is a social rate. A rate of 10 percent is more likely to reflect a commercial rate (i.e. a rate that reflects a farm manager's preference for profit streams earlier in a planning period) and as such an even greater potential for market failure is demonstrated by the difference in benefits between these two rates. As shown in Table 7.5 this disparity is more pronounced as the initial watertable depth increases, which reflects the realisation of the benefits of birdsfoot trefoil later in the planning horizon at such depths.

In light of the current drought, where the prices of agricultural products and grains in particular have increased to unprecedented levels, it is pertinent to investigate what sustained higher prices might mean for the value of a plant introduction. Figure 7.12 shows the impact of a significant increase in grain prices and cattle prices. For illustrative purposes the base price of grains is doubled and the base price of cattle is

increased by 50 percent, resulting in gross margins of $713 per hectare and $744 per head for the cropping rotation and cattle production respectively.

Figure 7.12 illustrates that for initial watertable depths less than 1.7 m, the introduction of BFT would provide no greater benefit if agricultural prices were to rise significantly. At depths greater than 1.7 m, however, the impact is significant. The present value of benefits over 50 years is increased to $20 million for depths of 2 m or greater and approaches $35 million for watertable depths of around 5.5 m. The optimal solution at these depths comprises largely the same enterprise mix as the base case, so the difference is primarily explained by higher prices for the same output.

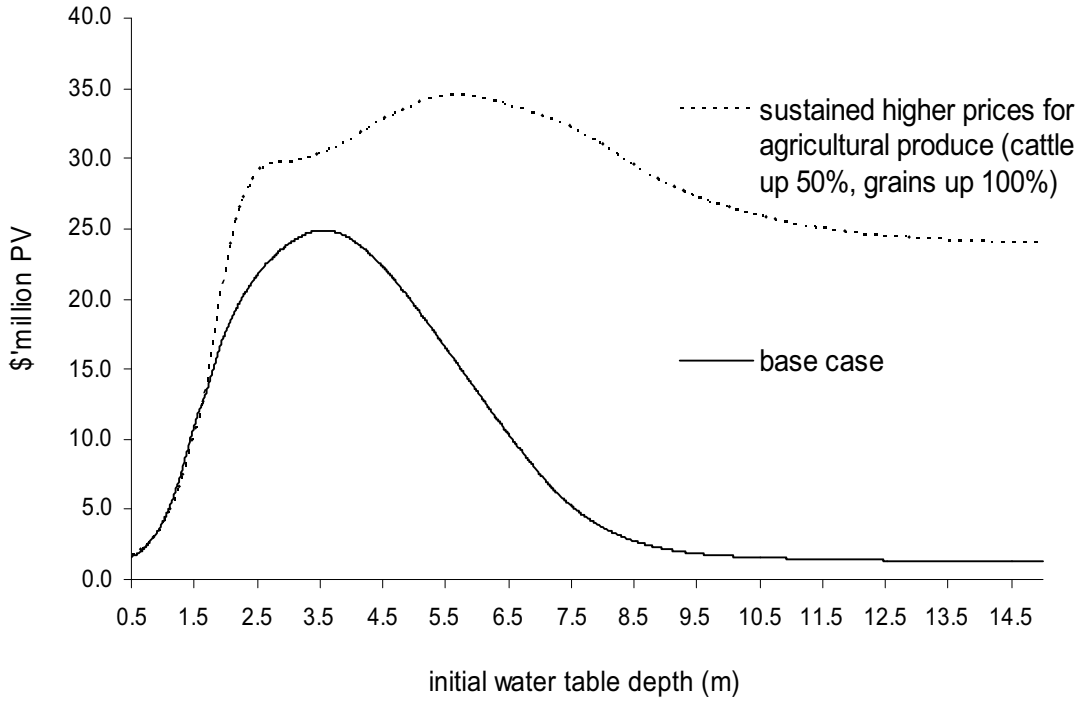


Figure 7.12 Benefits of BFT – *the case of sustained higher agricultural prices*

Higher prices for agricultural products provide an incentive to delay a rising watertable, but they also provide an incentive to capture more profits earlier in the planning horizon at the expense of those later. The competing incentives explain the relatively small differences in the state path with and without BFT as shown in

Figure 7.13, for three initial watertable depths when agricultural prices are high. The small difference again reflects the fact that birdsfoot trefoil provides a more profitable way to manage the watertable and also is profitable when the watertable is shallower.

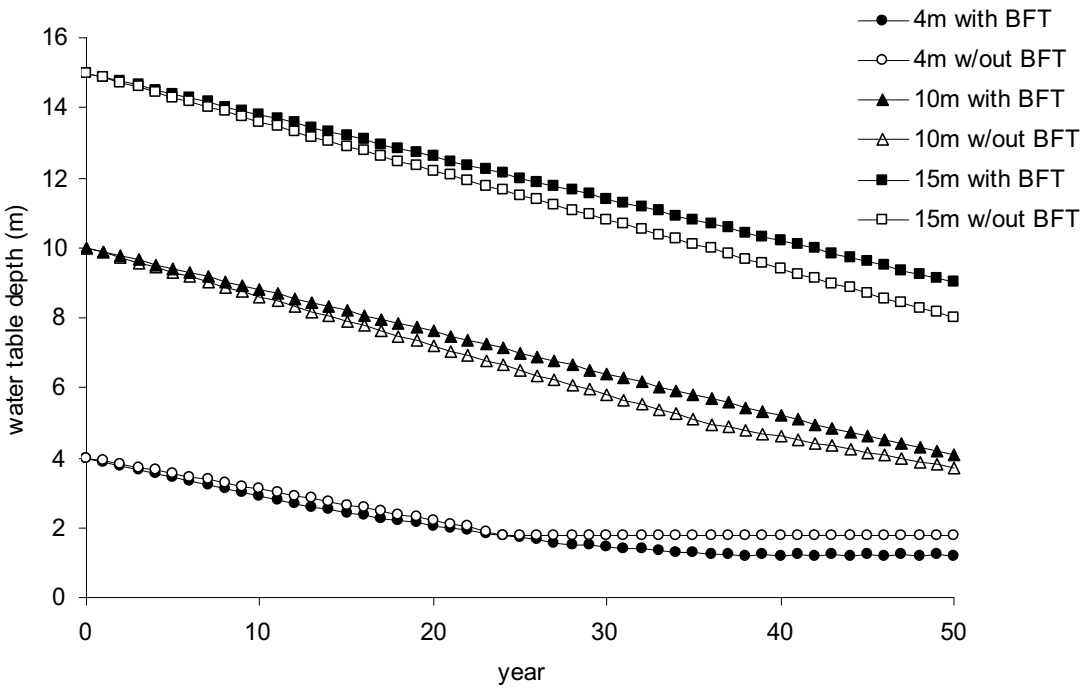


Figure 7.13 Optimal state path over 50 years at various depths – *the case of higher agricultural prices*

The key point shown by the scenario of higher prices is that as the value of production increases, there is also an increase in the value of the watertable and thus the value of a plant which provides an option to delay depletion of the watertable. In light of the potential for climate change to contribute to ongoing droughts, it is pertinent to consider that the benefits of birdsfoot trefoil might be that much higher than those estimated with current prices. However, this should be qualified, because if the Kings Plain Subcatchment is also in drought, there will be lower production and less recharge to manage and therefore less potential value from the introduction of BFT.

7.2 Potential BFT invasion of the Border Rivers Catchment

The costs of the potential invasion of the Border Rivers Catchment (BRC) that surrounds the Kings Plain Subcatchment (KPS) are presented in this section. Firstly, the output of the biophysical simulation of the invasion is presented, followed by the results of a deterministic bioeconomic analysis of the costs in agricultural and natural areas. The deterministic analysis considers firstly the impact of a single initial dispersal of seeds and then ongoing annual dispersals. Sensitivity analysis completes the section.

7.2.1 Biophysical simulation of plant invasion on a per-hectare basis

The simulation of the invasion of the BFT into the catchment is characterised by a dynamic process where there are feedbacks to the initial state variables (new seeds and seeds in the seedbank). The population dynamics in this analysis are based on a stage matrix that simulates the process of the plant invasion using the parameters identified in Chapter 6. The simulation is undertaken initially for the case of one hectare where there is a single dispersal in year 1 of 10,000 seeds dispersed randomly across the hectare. This is then extended to consider an average initial dispersal of 10,000 seeds per hectare across the entire Catchment. The analysis provides insight to the influence that plant features, including fecundity, seed longevity and seedling survival rates, have on the invasion. The estimated costs are shown to vary as a result of the potential variability of these features.

Figure 7.14 shows the simulated growth of the birdsfoot trefoil population over the 50 year period following dispersal. This illustrates the dynamic nature of the modelled invasion. In particular, shown in Figure 7.14 (a) is the number of seeds starting at 10,000 as per the initial dispersal, followed in year 1 by no new seeds but some 5,000 seeds in the seedbank. By year 3 the population includes new seeds as a result of juvenile plants which produce a small amount of seed and a greater number of new seeds in year 4 when the first adult plants begin producing seeds. The corresponding growth in the plant population is illustrated in Figure 7.14 (b).

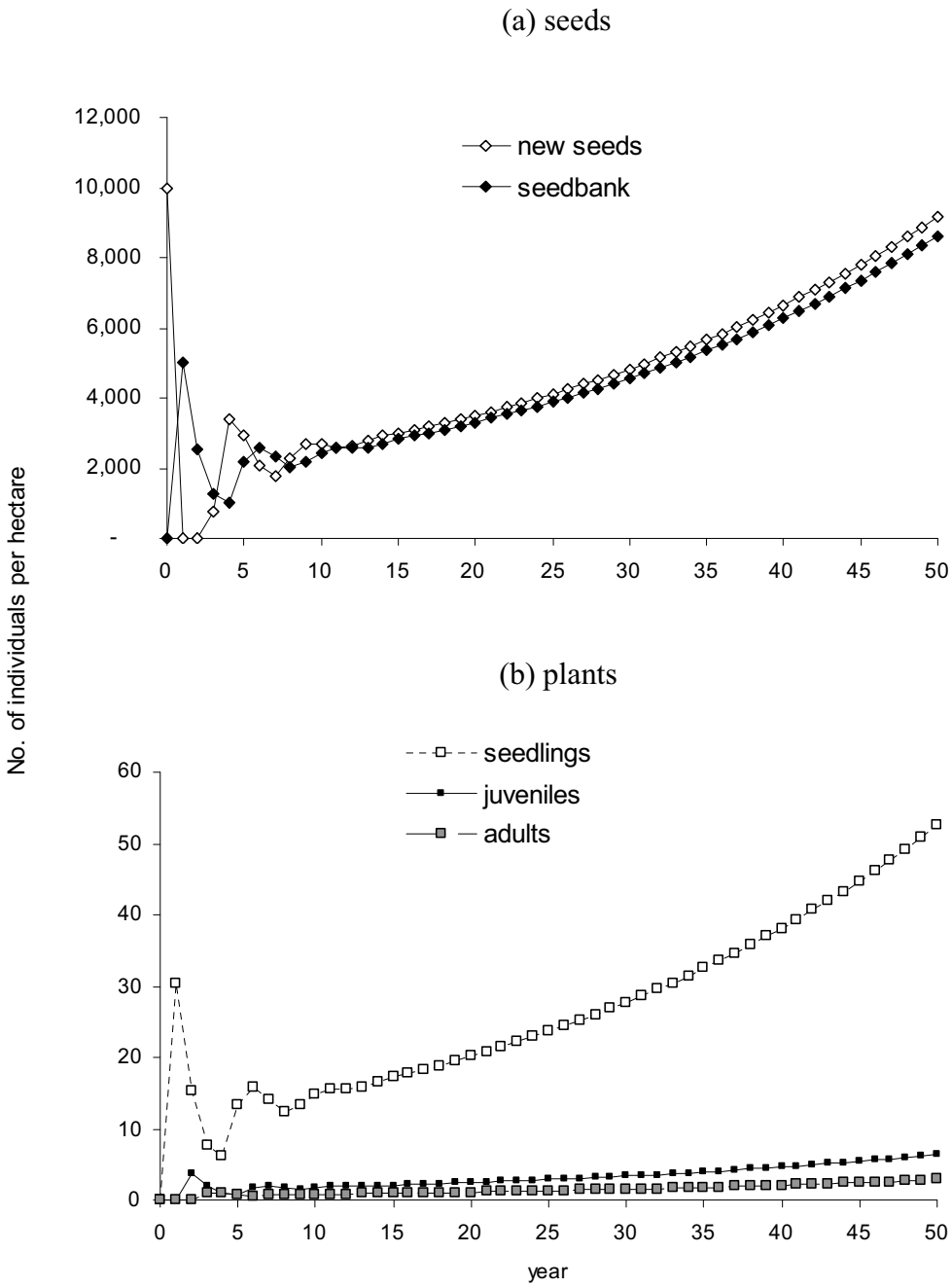


Figure 7.14 Population dynamics of birdsfoot trefoil over 50 years starting with 10,000 seeds per hectare – *no. of individuals per hectare*

The growth of the invasion in terms of area over the same period is shown in Figure 7.15. On the base assumptions applied, the invasion is just 0.018 percent of the hectare, or the equivalent of just 2-3 adult plants, 6 juveniles and around 50 seedlings, after 50 years. In year 50, there are 8,600 seeds per ha in the seedbank and just over 9,000 new seeds produced.

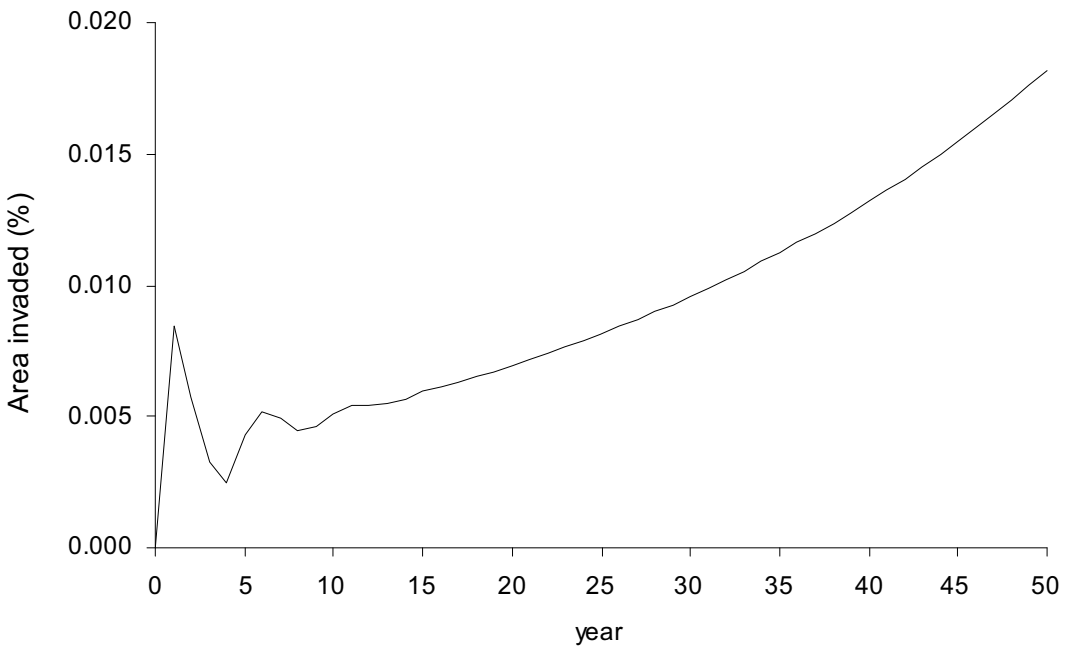


Figure 7.15 Simulated area of invasion over 50 years

But what would the result be if there was a dispersal of 10,000 seeds annually, as might be the case where a plant is grown as a pasture? Figure 7.16 illustrates the growth in area invaded where there are annual dispersals of 10,000 seeds, together with growth for a single initial dispersal for the purpose of comparison. Where there is an annual dispersal of 10,000 seeds, the area invaded after 50 years is 0.48 percent of a hectare. The estimated areas of invasion for both an initial and for ongoing annual dispersals are small after 50 years, consistent with the low potential for BFT to naturally regenerate as discussed in Chapter 6.

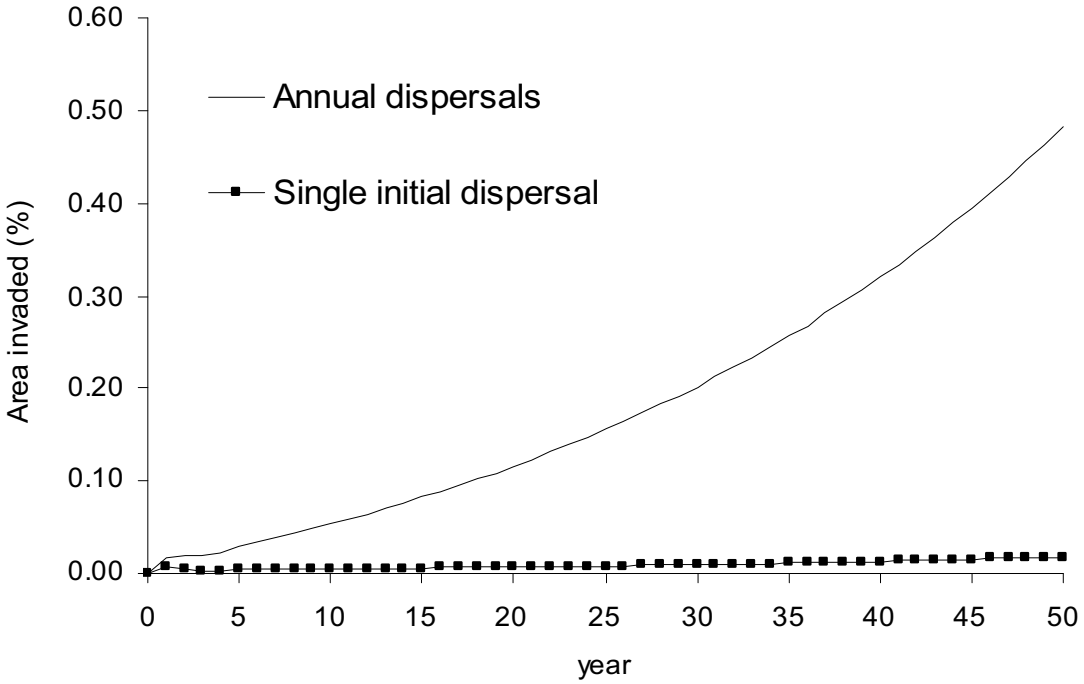


Figure 7.16 Growth in area invaded – *single and annual dispersals of seed*

In this analysis, the area invaded (A_{inv}) is used as both the proportion of the average hectare that is invaded, and a proxy for the proportion of the total number of hectares that could be invaded at any time. As such the above analysis can be interpreted as, by year 50:

- with a single initial dispersal of 10,000 seeds, 0.018 percent of the Catchment is invaded with an average population density of 2-3 adult plants, 6 juveniles and around 50 seedlings per ha; and
- with annual dispersals of 10,000 seeds, 0.48 percent of the Catchment is invaded an average population density of around 1,400 seedlings, 170 juvenile plants and 70 adult plants per ha.

With respect to the BRC and in the case of *annual ongoing dispersal*, birdsfoot trefoil is estimated to have invaded 6,075 hectares of agricultural land by year 50 ($A_{inv(50)}$ multiplied by the total area at risk of invasion) with the size of the invasion per hectare averaging 48 m^2.

Analogous analysis for the natural areas was undertaken with an adjusted Stage matrix (where $\lambda_N = 1.025$). The Stage matrix was adjusted to accommodate a reduced population growth rate. This reflects that populations can generally be expected to be less invasive when disturbance is less (Hobbs, 1991; Hobbs and Humphries, 1995; Sheppard, 2000); as would be the case in natural areas where disturbance is less frequent and less widespread than in agricultural areas subject to regular cultivation and movement of machinery and stock. The results of the simulation in both agricultural and natural lands are provided in Table 7.6.

Table 7.6 Simulated invasion of Border Rivers Catchment – *Summary of biophysical simulation results*

	Agricultural lands		**Natural areas**	
Total area at risk (ha)	1,260,290		346,593	
Population growth rate (λ)	1.0325		1.025	
Simulated invasion of BFT				
Dispersals	*Single initial*	*Annual*	*Single initial*	*Annual*
Net area invaded in Year 50 (ha)	228	6,075	42	1,293
Average size of invasion in each invaded hectare (m^2)	1.8	48.0	1.2	37.0

7.2.2 Bioeconomic results

The costs associated with the invasion of BFT into agricultural and natural areas of the surrounding BRC are estimated using a damage function specific to each area (see Equation 5.17 and Figures 6.3 (a) and (b)). As an area is progressively invaded, the value of its output falls.

Through the application of the damage functions, the costs have been estimated as a present value over 50 years and at a discount rate of 7 percent. The present value of the costs in agricultural areas is estimated to be just over $43,000 in the case of a single initial dispersal of 10,000 seeds and just over $593,000 in the case of ongoing

annual dispersals of 10,000 seeds. Table 7.7 summarises the costs in both agricultural and natural areas.

Table 7.7 Costs in agricultural and natural areas of the Border Rivers Catchment invaded by birdsfoot trefoil (PV over 50 years at 7 percent)

	Agricultural lands (C_A)		**Natural areas (C_N)**		**Total (C)**	
Dispersals	*Single initial 10,000*	*Annual 10,000*	*Single initial 10,000*	*Annual 10,000*	*Single initial 10,000*	*Annual 10,000*
Total cost ($)	43,163	593,102	796	11,137	43,959	604,239
Cost per ha ($/ha invaded in year 50)	188	97	19	9	207	106
Cost per seed dispersed ($)	4.30	1.20	0.08	0.02	4.38	1.22

Of note from Table 7.7 is the lower cost per hectare and per seed for the case of annual ongoing dispersals. This reflects the diminishing losses as the invasion progresses and the discounting of costs later in the analysis period.

7.2.3 Sensitivity analysis

The analysis shown thus far presents the case for low costs from the potential invasion of birdsfoot trefoil in the BRC. The sensitivity of these results has been tested for the case where there are ongoing annual dispersals. Table 7.8 summarises the results from the estimation of the costs of the invasion (both in agricultural and natural areas) given variation in key parameters. In particular, the sensitivity of the results to dispersal, plant characteristics, and market factors are presented.

Table 7.8 Effect of changing parameters on the cost of invasion - *annual dispersals*

Parameters		Elasticity
Plant characteristics	Fecundity (seeds produced annually by plants)	2.4
	Germination rate	2.8
	Seedling survival rate	3.4
Market Factors	Discount rate	-8.6
	Productive values (value of production/biodiversity)	1.0
Damage Parameters	Shape of damage function	1.0
	Lower loss threshold	-1.8

Sensitivity analysis shows the estimated cost of the invasion of birdsfoot trefoil in the BRC to be sensitive to a range of parameters. Estimates of cost are sensitive to the plant's demographic characteristics and the discount rate. Sensitivity to the discount rate highlights the potential for market failure, while sensitivity to plant demographic characteristics highlights the need for scientific evaluation to inform economic analyses of such a plant invasion. This analysis has been undertaken on the basis of dispersal of an average of 10,000 seeds into each hectare annually, but the number of seeds dispersed is a key unknown parameter.

7.3 Net benefits of an introduction

This section considers the net benefits of the introduction of birdsfoot trefoil after taking account of the external costs of plant invasiveness in the surrounding catchment.

7.3.1 Benefit-cost analysis

The decision criterion for this deterministic model would be

$$(B-(C_A+C_N)>0 \qquad \textbf{(7.1)}$$

That is, a plant should be introduced when the discounted benefits of the plant's introduction minus the discounted external costs in agricultural and natural areas is greater than zero.

For the case of one initial dispersal of 10,000 seeds, B = \$8.8 million and $C_A + C_N$ = \$43,959, such that the net benefits of the introduction of BFT to KPS is greater than zero. This result also holds for the case of annual dispersals of 10,000 seeds, where $C_A + C_N$ = \$604,239 and the net benefit is \$8.2 million.

On the basis of the assumptions and parameters used (including the conservative assumption that the population growth rate of the new varieties of BFT would be twice that observed for existing varieties) the introduction of birdsfoot trefoil should be approved for introduction.

A benefit-cost ratio (BCR) is considered desirable when it is greater than one (Sinden and Thampapillai, 1995) and as such under both circumstances the plant's introduction should be considered desirable: the BCRs are 200 and 15 for the case of a single initial dispersal and the case of annual ongoing dispersals respectively.

7.3.2 Breakeven analysis

Breakeven analysis allows consideration of factors which may alter the net benefits estimated in an analysis, and as such change the conclusion with regard to the desirability of, in this case, the plant's introduction.

In the case of ongoing annual dispersals, the average annual dispersal would need to increase by 15 times, to 150,000 seeds per hectare in the Catchment, for the estimated costs to equal the estimated benefits. With the possibility of some 36,000 hectares of birdsfoot trefoil planted in the subcatchment (see Section 7.1), that is just under 3 percent of all seeds from the Subcatchment that would need to be dispersed into the Catchment every year. As shown in Table 7.9 this is the equivalent to requiring the

dispersal of some 6.7 million seeds per hectare of planted pasture into the surrounding Catchment.

Table 7.9 Birdsfoot trefoil seeds required to enable a range of potential seed dispersal events in the Border Rivers Catchment

Seeds potentially dispersed in Border Rivers Catchment (as modeled in analysis)		Equivalent number of seeds required to be produced in Kings Plain Subcatchment	
Number per hectare per annum	**Equivalent annual catchment total [(a)] ('million)**	**Per hectare of pasture per annum ('million) [(b)]**	**Proportion of total seeds produced per annum [(c)]**
10,000	16,069	0.45	0.19%
15,000	24,103	0.67	0.28%
20,000	32,138	0.90	0.37%
50,000	80,344	2.23	0.93%
100,000	160,688	4.46	1.86%
150,000	241,032	6.70	2.79%

(a) The total number of seeds for dispersal is the number of seeds potentially dispersed per hectare multiplied by the total number of hectares at risk in the Catchment (1.6 million). The equivalent number of seeds required for dispersal is reported as the number per hectare of pasture planted in Subcatchment **(b)** and as a proportion of all seeds produced by the pasture in the Subcatchment **(c)**. The total number of seeds produced in the KPS by 36,000 hectares is 8.20×10^{12} per annum (36,000 hectares multiplied by 80,000 adult BFT plants/hectare (as per Table 6.5) multiplied by 2,875 seeds/adult plant).

Dispersal of 3 percent of all seeds produced in the Kings Plain Subcatchment does not appear unreasonable, nor a possibility that can be ignored. However, as discussed in Chapter 6, BFT has limited vectors for transfer, such that dispersal of BFT beyond the immediate boundaries, in the volumes required at breakeven would appear limited.

Another question of interest is how many additional hectares would need to be at risk in the surrounding catchment for the costs to outweigh the benefits. Results indicate that almost 19 million ha of agricultural land would need to be at risk of invasion before the costs outweigh the benefits at base parameter values. This is again unlikely given that birdsfoot trefoil has environmental limitations that would prevent it from invading such a large amount of land (see Table 6.1). The potential for a plant like birdsfoot trefoil, with limited vectors for transfer, is a further factor reducing how

reasonable it is to consider such a large at risk area, particularly within the analysis period investigated in this study. Costs associated with invasion of areas in subsequent years, after year 50, would be sufficiently discounted to not make a significant difference to a decision based on the criterion developed in this study.

7.3.3 Sensitivity analysis: costs in natural areas

The cost of the invasion of birdsfoot trefoil in the surrounding Border Rivers Catchment has been assessed for two discrete kinds of areas: agricultural areas and natural areas. The costs in natural areas were found to be minor compared to those in agricultural areas so that assessment without costs in natural areas might be justified. For the case of annual dispersals of 10,000 seeds per hectare, the present value of costs is just over $11,000 as compared to a present value of almost $600,000 in agricultural areas. To change this ranking, the value of biodiversity or the natural area at risk would have to increase substantially.

The natural area at risk from invasion by the new plant is just less than 350,000 hectares in the base analysis: one hectare of natural land at risk of invasion for every 4 hectares of grazed or cropped agricultural land at risk of invasion (Table 6.8). There is the potential for this ratio to be quite different in other Catchments such that the potential costs in natural areas may not be a minor contributor to total costs. Under the base analysis and assumptions used in this study, the ratio of hectares of natural lands to hectares of agricultural lands would need to be more than twelve to one before the value of costs in natural areas would approach the value of costs in agricultural lands. Even with this ratio of natural lands to agricultural lands there would still be net benefits from the introduction as estimated under the base case in this analysis: the total costs would not breakeven with the estimated total benefits. However, the implication is that an assessment might require inclusion of costs in natural areas where agricultural lands at risk are located in areas with an abundance of natural lands at risk.

If the damage caused by the plant was catastrophic, the ranking may also change. The susceptibility of natural areas populated with well adapted native species is likely to be low and this has been reflected in the choice of damage function in natural areas. The damage function is shown in Figure 7.17, where the value of q describes the shape of the function. A value of q = 0.5 was assumed for the analysis. This value reflects an assumption of increasing marginal loss: smaller initial losses followed by relatively larger losses as the invasion proceeds. If the native flora or fauna of the area were particularly susceptible to a new invasion of birdsfoot trefoil then the value for q would be higher. That is, if the introduced plant were especially competitive at low levels of invasion or perhaps it were poisonous to native fauna then a much higher value of q would be appropriate.

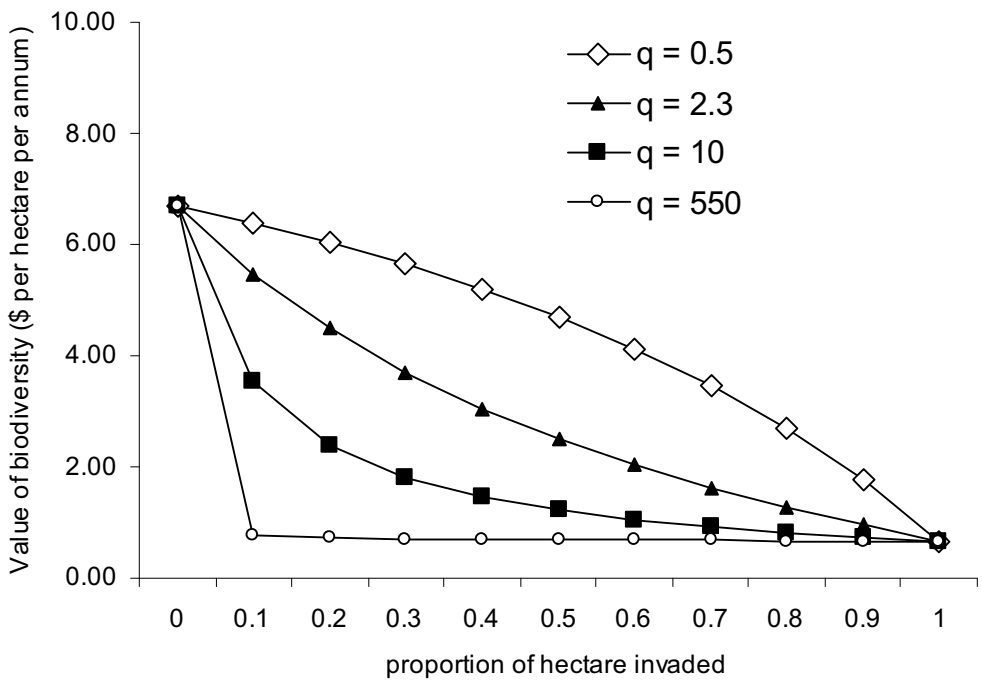


Figure 7.17 The damage function for a range of values of q (the susceptibility of natural species increases with q)

Figure 7.17 illustrates the annual loss in value of biodiversity flowing from natural areas at various values of q. Given the base assumptions used in this analysis, the value of q would need to increase from 0.5 to 2.3 for the costs in natural areas to approach a value equal those estimated for agricultural areas. The breakeven value of q for natural areas was calculated at 550. As shown in Figure 7.17, q = 550 implies

immediate catastrophic losses of biodiversity at very low levels of invasion. There is no evidence that Australian native species are particularly susceptible to birdsfoot trefoil.

The assumption that natural areas are homogeneous may also affect the estimated costs and thus relative costs in agricultural and natural areas. The base analysis in this study assumes an even spread of endangered species throughout the natural areas, such that the value of biodiversity is homogeneous throughout. If however, species of high value are concentrated in one part of the Catchment, then there is a range of costs that might be estimated for natural areas.

At one extreme, this means that the loss of value could be $0 if the natural areas invaded have no scarcity value. At the other extreme, the costs could be much higher if the area invaded is home to key valuable species not found elsewhere. In this instance, the annual value of biodiversity per hectare would not be apportioned throughout the natural area as in the base case, but rather be the value used to estimate the total losses in that particular hectare. That is, the value at risk in that hectare would be $64,830 per species per year, for a total of $3.5 million annually[18]. If the invasion were concentrated in that one hectare of the Catchment then the present value of invasion costs in natural areas would be $18,000 as compared to $11,000 when dispersed homogeneously throughout all natural areas of the Catchment. The number of endangered species in such a concentrated area at risk would still need to increase more than fivefold before the costs in natural areas would approach the costs in agricultural areas, and still there would be net benefits from the introduction.

For this case, the finding that the burden of costs will be primarily borne in agricultural areas is shown by these analyses to be robust and that the subsequent focus on them in this study is sound. More generally, however, it demonstrates that for a more competitive new plant, for native flora and fauna that is susceptible and in areas with a higher ratio of natural to agricultural lands, potential for the burden of

[18] $64,830 per species p.a. was estimated as the value of protecting one threatened species (Sinden and Griffith G.R. (2007)). A conservative estimate of the total value of endangered species at risk was found by multiplying this value by 54, the number of endangered species reported to be at risk in one part of the Border Rivers Catchment, the Inverell-Yallaroi Local Government Areas (DLWC 2002).

costs to be concentrated in natural areas is greater and so is the need for costs in these areas to be considered.

7.4 Summary

The results of the deterministic analysis indicate there to be benefits from the introduction of birdsfoot trefoil to the Kings Plain Subcatchment, and that these are largely unchanged when net of the external costs of a possible invasion of the surrounding Border Rivers Catchment. In particular, it has been found that

> "…the costs associated with the plant's introduction, including negative externalities DO NOT EXCEED the benefits from the plant, including positive externalities."

On the basis of the deterministic analysis presented in this chapter, entry of the plant should be endorsed. The analysis has shown the sensitivity of the result to various parameters and cases where costs may outweigh benefits. Further, it has revealed potential market failures and the need for policy to accompany a decision to introduce such a new plant.

The criterion as stated above is not complete as it does not include uncertainty. Uncertain events include the number of seeds that will be dispersed, the growth rate of the population and how effectively salinity is managed by a new plant. Uncertainty is introduced into the decision criterion in the next chapter.

8 Introducing Uncertainty

The dynamic processes modelled to estimate the benefits and costs of introducing birdsfoot trefoil in Chapter 7 are inherently uncertain. In this chapter, the uncertainty of the plant's effectiveness in reducing recharge and uncertainty of its potential to become invasive are incorporated into the analysis, which then provides the data for complete consideration of the decision criterion as developed in Chapter 5.

Two approaches are adopted to incorporate uncertainty. The range of benefits of reduced recharge that might result from the introduction of birdsfoot trefoil is estimated, and stochastic analysis is applied to the potential plant invasion by incorporating beta distributions around key demographic parameters. These two approaches were adopted to reduce the numerical complexity of simultaneously analysing both problems stochastically.

The outputs of each of these approaches to considering uncertainty are presented separately. The range of benefits of reducing recharge is presented first, followed by the estimated stochastic cost function. Analysis in both cases is again undertaken over 50 years with a discount rate of 7 percent. These results are then examined together to consider the net benefits of the introduction under uncertainty.

8.1 Range of benefits of reduced recharge

The benefits to be gained from the introduction of birdsfoot trefoil (BFT) are sensitive to assumptions with respect to the plant itself and the nature of the catchment (Chapter 7). In particular, the estimates were found to be sensitive to the cost of establishing a new pasture, the productive capacity of the new pasture, the capacity of the plant to limit drainage beyond its root zone and the capacity of the subcatchment itself to drain discharge.

The benefits of introduction have been estimated for a range of values for each of these parameters (Table 8.1), all else constant. The results of this analysis are shown in Figure 8.1 and are discussed in turn.

Table 8.1 Parameter values adopted to consider the range of benefits of birdsfoot trefoil introduction

Parameter	Low end of range	High end of range	Source/notes
BFT establishment cost	$37 per ha	$32 per ha	Assumes cost of seed (as primary unknown in the gross margin calculations) is doubled in the worst case scenario and reduced by 50 percent in best case.
BFT pasture production	Reduced by 25 percent	Equivalent to Lucerne in production quantity and timing	Arbitrary assumption.
BFT recharge	30mm/m^2	10mm/m^2	Range reported in Table 6.1.3 of Chapter 6.
Catchment drainage	0.5 mm/m^2	10 mm/m^2	Range reported by Stirzaker *et al.* (2000)

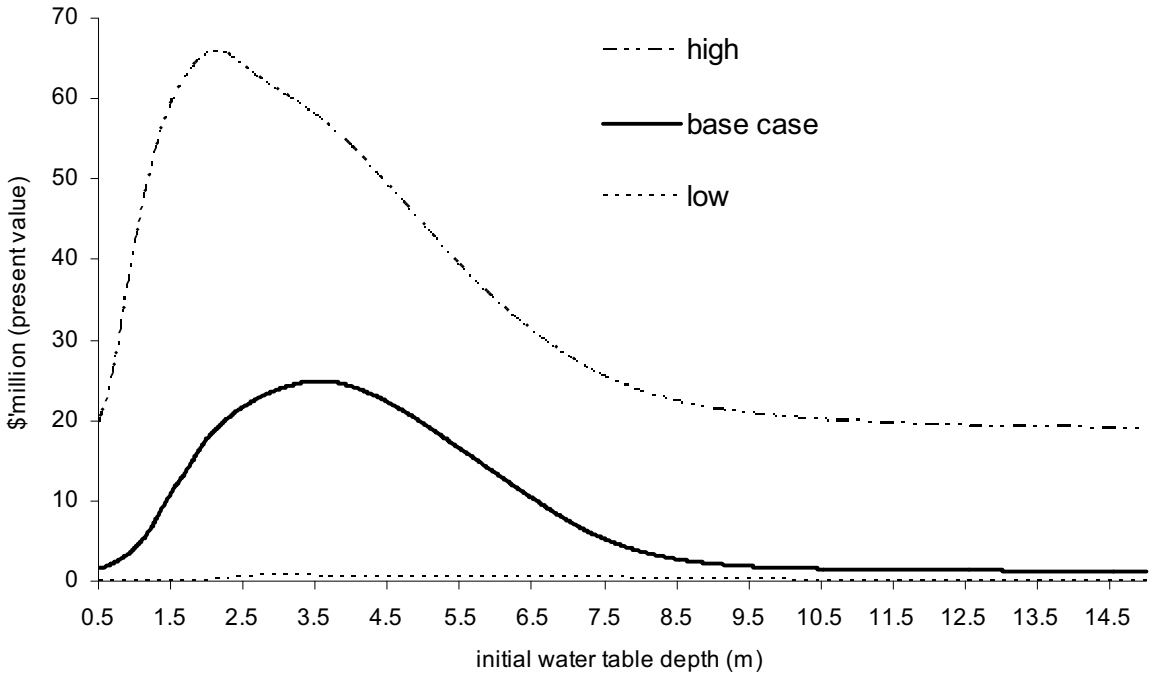


Figure 8.1 Benefits of birdsfoot trefoil introduction over a range of initial watertable depths –*range of possible outcomes*

8.1.1 Low potential benefits

Following the same procedure explained in Chapter 7, but now using parameters for the lower end of the range, the benefits from the introduction of BFT to a single subcatchment have been found to be potentially quite low. These are illustrated in Figure 8.1. In particular:

- the maximum benefit is estimated to be a present value of $0.7 million at an initial watertable depth of 3 m throughout a subcatchment;
- the minimum benefit is a present value of $0 for initial depths less than 1.8 m, and less than $1 for all initial depths greater than 10 m; and
- Kings Plain Subcatchment benefits could be expected to be in the order of $0.25 million based on estimated initial watertable depths in the subcatchment.

8.1.2 High potential benefits

The benefits from the introduction of BFT to a single subcatchment using values for the higher end of the range have been found to be potentially quite high. These are shown in Figure 8.1. In particular:

- the maximum benefit is estimated to be a present value of $66 million at an initial watertable depth of 2.4 m;
- the minimum benefit exceeds a present value of $19 million for all initial watertable depths examined; and
- Kings Plains Subcatchment benefits could be expected to be in the order of $33 million based on estimated initial watertable depths in the subcatchment.

8.1.3 Range of benefits

This analysis demonstrates the wide range of benefits that may be captured given the potential variance in the performance of BFT and the subcatchments to which it might be introduced. It shows that in some subcatchments there may be no benefits where the watertable is already very close to the surface (less than 1.8 m) or sufficiently

deep (greater than 10 m). On the basis of the watertable depths estimated using bore data, the benefits for a subcatchment such as the Kings Plain Subcatchment could potentially range from $0.25 million to $33 million as a present value over 50 years.

8.2 Stochastic analysis of the costs of invasion

The uncertain nature of plant populations is integrated within the analysis by considering three stochastic factors:

- germination rate (G);
- seedling survival rate (P_{J1}); and
- number of seeds produced per adult plant (F_A).

Monte Carlo simulations with 1,000 iterations were undertaken assuming annual seed dispersal events. The three factors were each assumed to conform to a beta distribution, which is used widely because of its flexibility (Cacho *et al.*, 2006; Gupta and Nadarajah, 2004). Stochastic values for the three factors were implemented by defining the random variables, as per Cacho *et al.* (2006):

$$\hat{G} = \tau_G G \qquad \textbf{(8.1)}$$

$$\hat{P}_{J1} = \tau_{PJ1} P_{J1} \qquad \textbf{(8.2)}$$

$$\hat{F}_A = \tau_{F_A} F_A \qquad \textbf{(8.3)}$$

where τ s represent random numbers drawn from a beta distribution and adjusted to fall within a range $(x_{\min}, x_{\max})$. The bounds $x_{\min}$ and $x_{\max}$ are expressed as proportions of the expected value. The random numbers are rescaled to fall within the range specified because those obtained from a beta distribution fall between 0 and 1. The following formula is applied to rescale the randomly selected numbers:

$$\tau_i = B_i(a,b)(x_{\min} - x_{\max}) + x_{\min} \qquad \textbf{(8.4)}$$

where $B_i(a,b)$ represents a random number drawn from the beta distribution with parameters *a* and *b*. The two parameters, *a* and *b*, determine the shape of the distribution and its position within the (0,1) interval. The random parameters τ_G, $\tau_{P_{J1}}$ and τ_{F_A} were obtained by setting (*a*,*b*) to (4,2) as this is a plausible representation of biological distributions (Cacho *et al.*, 2006). The bounds (x_{min}, x_{max}) were set to (0.01,1.35), (0.00,1.50) and (0.01, 1.50) for germination, seedling survival and number of seeds produced per adult plant respectively, to achieve plausible outcomes (pers comm., Ayres and Lane, 2007). In particular, the parameter values were selected to reflect the potential for very low germination, no seedling survival and very low seed production in unfavourable years and above average germination rates, seedling survival and seed production in highly favourable years.

A further element of uncertainty is introduced to the analysis by incorporating a random normal distribution around the number of seeds dispersed annually. For each iteration, the number of seeds dispersed is selected from a truncated normal distribution with coefficient of variation of 0.5 around the mean of 10,000 seeds.

As before, the analysis is based on a discount rate of 7 percent over 50 years. The results of this analysis for agricultural and natural areas are now presented.

8.2.1 Costs of plant invasiveness

The introduction of uncertainty demonstrates a considerable impact on the estimated cost of birdsfoot trefoil's potential invasion of the Border Rivers Catchment. The mean present value cost of the invasion is $0.45 million, with minimum and maximum values of $0.25 million and $0.75 million respectively. The distribution of costs in year 50 is found to be wide with a standard deviation of $0.10 million. The considerable variance in estimated costs is because of the considerable variance in the proportion of area invaded by Year 50. This is illustrated in Figure 8.2 which shows results of the Monte Carlo simulations over 50 years.

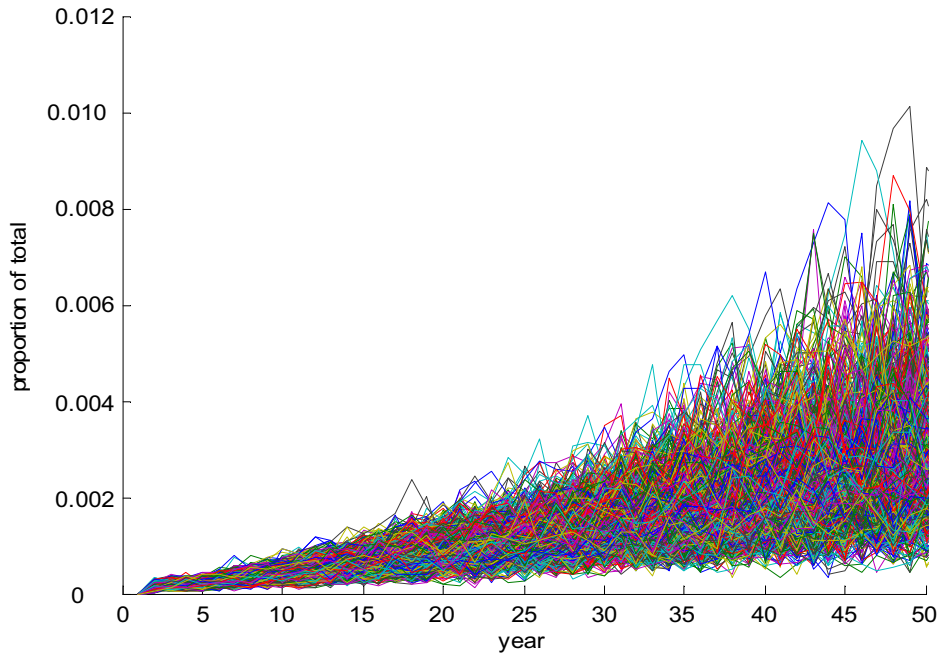


Figure 8.2 Agricultural area of Border Rivers Catchment invaded over 50 years (proportion of total) – *annual dispersals*

The cumulative probabilities of area invaded were estimated for 10, 20 and 50 years (Figure 8.3). Comparison of these three cumulative distribution functions (CDFs) shows that the costs of the invasion increase over time and that the range of potential area invaded increases. In particular:

- after 10 years, 90 percent of the time the area invaded will be less than 0.06 percent of the total potential area;
- after 20 years, 90 percent of the time the area invaded will be less than 0.14 percent of the total potential area; and
- after 50 years, 90 percent of the time the area invaded will be less than 0.47 percent of the total potential area.

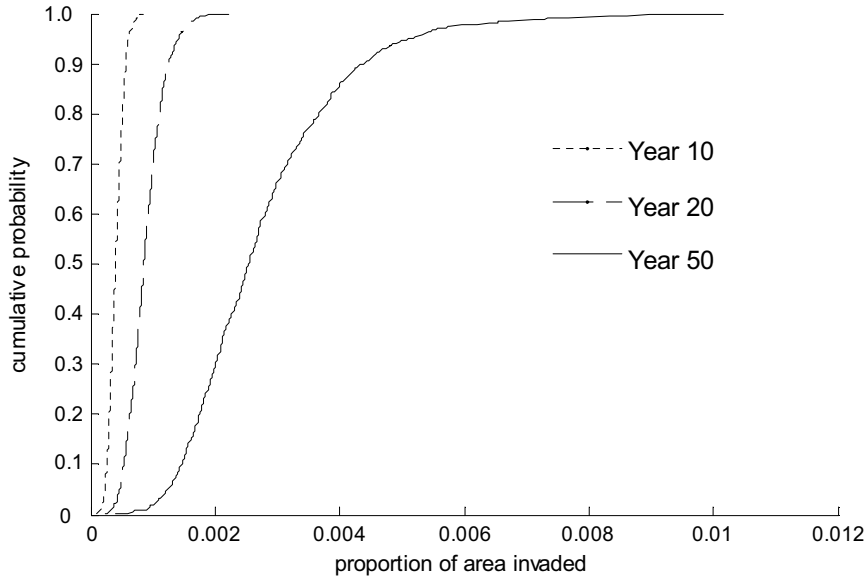


Figure 8.3 Cumulative probability distribution of area of plant invasion in agricultural areas of the Border Rivers Catchment – *Years 10, 20 & 50*

Analysis of the costs associated with the invasion after 50 years indicates a probability of 0.90 that costs will be below $0.6 million as a present value (Figure 8.4).

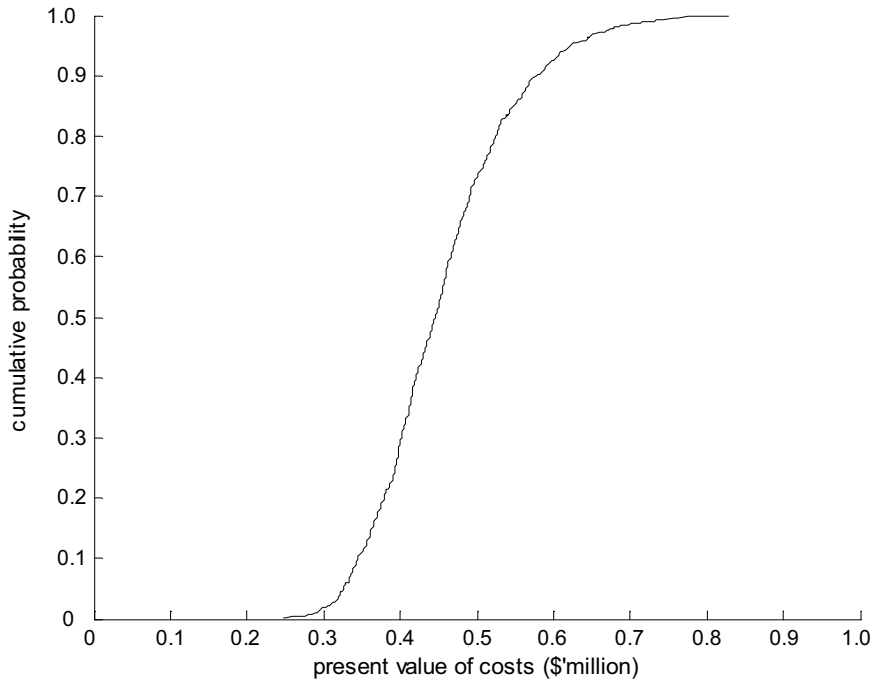


Figure 8.4 Cumulative probability distribution of costs of plant invasion in agricultural areas of the Border Rivers Catchment – *Year 50*

Equivalent stochastic simulation of the costs of an invasion in natural areas of the Border Rivers Catchment found that over 50 years the present value of costs:

- would have a mean of $8,590;
- would have a minimum and maximum of $5,450 and $15,990 respectively; and
- 90 percent of the time would be between $6,400 and $11,100.

8.2.2 Total Costs

Bringing the estimated costs in agricultural areas and natural areas of the Border Rivers Catchment together, the present value of total costs over 50 years are estimated to:

- have a mean of $0.45 million;
- have a minimum and maximum of $0.25 million and $0.81 million respectively; and
- be between $0.34 million and $0.55 million 90 percent of the time.

These total costs are the same as the estimated costs for just agricultural areas because the contribution of the costs in natural areas is too small to make a difference to the rounded estimates reported. Analysis for the remainder of this chapter therefore focuses on the costs in agricultural areas and the issue of costs in natural areas is revisited in Chapter 9 in the context of policy.

8.2.3 Stochastic dependency

When two or more stochastic elements are introduced to an analysis, there is the potential for there to be dependency between the elements (Hardaker *et al.*, 2004). That is, the uncertain elements may be influenced by the same environmental factors, such that sampling 'high' and 'low' for each stochastic element should be consistent. The analysis above has not incorporated potential dependencies.

Specifying a correlation matrix is one method for accounting for such dependency. Hardaker *et al.* (2004) point out that correlation matrices only describe dependency in terms of covariances and miss out on other aspects of stochastic dependency. However, because correlation matrices generally account for most of the dependencies, and because of difficulties with other ways of measuring and accounting for all aspects of dependency, they are commonly used to account for dependencies in stochastic simulation.

The analysis above was undertaken assuming the correlation coefficient matrix shown in Equation 8.5

$$\rho = \begin{bmatrix} 1 & 0 & 0 \\ 0 & 1 & 0 \\ 0 & 0 & 1 \end{bmatrix} \qquad \textbf{(8.5)}$$

where the elements represent correlation between the stochastic factors (G, P_{J1} and F_A). To investigate the impact of stochastic dependency, alternative correlation matrices were incorporated into the model using the Copulas technique (Perkins and Lane, 2003). Monte Carlo simulations were undertaken for the base case (ρ given in Equation 8.5) and for cases where the zeros in the correlation matrix were replaced by 0.5 and then 0.9. Comparison of results for all three correlation matrices allows assessment of the potential impacts of stochastic dependency. The values of 0.5 and 0.9 represent increasing levels of positive correlation among the stochastic factors. No negative correlations were tested, as it is unlikely that the factors (germination, survival and fecundity) would be dissimilarly impacted by favourable or unfavourable environmental factors. These arbitrary values have been used in the absence of correlation data.

The results of the analysis in Table 8.2 show that the median estimated cost varies by 2 percent as the assumption of correlation increases. Accounting for stochastic dependency causes the tails of the distribution to stretch, therefore, at percentiles less than the median, costs will be *less* than estimated under the base assumptions (e.g. at the 5th percentile costs are estimated to be 16 percent lower when $\rho = 0.9$) and at

percentiles greater than the median, costs will be *higher* than estimated under the base assumptions (e.g. at the 95th percentile the costs are 19 percent higher under the assumption $\rho = 0.9$). In this case, the assumption of no correlation is shown to potentially overestimate costs at low percentiles and potentially underestimate costs at high percentiles of the distribution.

Table 8.2 Impact of stochastic dependency on agricultural costs of birdsfoot trefoil invasion –*three assumptions of correlation*

	$\rho = 0.0$	$\rho = 0.5$	$\rho = 0.9$
Percentile		$'million PV	
0.05	0.32	0.29	0.27
0.10	0.35	0.32	0.30
0.50	0.45	0.44	0.44
0.90	0.58	0.62	0.65
0.95	0.63	0.68	0.75

Even when data are not available to specify the relevant correlation coefficients, assessment for the range of potential correlations as shown here may help decision makers assess whether their decision is robust under a wide range of conditions. This would be especially important at higher required levels of confidence, where there is the potential to underestimate costs if dependencies are ignored. On the basis of the assumptions used in this analysis the potential error introduced by assuming no dependency is minor. In particular, the assumption would not alter the estimates of costs enough to change the decision to introduce the plant on the basis of net benefits.

8.3 Net benefits under uncertainty

The benefits and costs under uncertainty are now brought together to consider the net benefits of the plant's introduction. Firstly, the net benefits are considered with respect to the original problem of just the Kings Plain Subcatchment and the Border Rivers Catchment. The scope is then expanded to consider the plant's wider introduction in a number of subcatchments.

8.3.1 Kings Plain Subcatchment and the Border River Catchment

Under the base assumptions, the net benefits of the plant's introduction are positive at all levels of confidence with respect to the plant's invasion (Table 8.3). Under the assumptions for the high end of the range, the benefits are positive at all levels of confidence. As such, under the base case and the high end of the range for expectations, the plant would be approved for introduction on the basis of the decision criterion developed in this study.

As shown in Table 8.3 however, under expectations for the lower end of potential benefits, the net benefits have a mean of - $0.20 million, and range from - $0.10 million to -$0.33 million for the 10th – 90th percentiles of estimated cost. In this case, the plant's entry would be denied at all levels of confidence.

Table 8.3 Net present value of benefits of introduction under uncertainty ($'million) - *Kings Plain Subcatchment only*

		Benefits of Reduced Salinity		
		Low	**Base**	**High**
Costs of Invasiveness		$0.25	$8.80	$33.00
percentile				
0.05	$0.32	- $0.07	$8.48	$32.68
0.10	$0.35	-$0.10	$8.45	$32.65
0.50	$0.45	-$0.20	$8.35	$32.55
0.90	$0.58	-$0.33	$8.22	$32.42
0.95	$0.63	-$0.38	$8.17	$32.37

Shaded results highlight positive net benefits and therefore that entry should be approved in these cases.

8.3.2 Multiple Subcatchments and the Border River Catchment

The net benefits of introduction are now considered with respect to introduction of the plant to multiple subcatchments in the Border Rivers Catchment (BRC). The Kings Plain Subcatchment (KPS) is just one of some 18 subcatchments in the BRC. Three

of these, located to the west of the BRC, have an average annual rainfall of less than 600 mm per annum across at least a large proportion of each (New South Wales Government, 2007), and as such are unsuitable for introduction of BFT pasture. Of the remaining 15, six others are similar to the KPS in terms of land use and size (DNR, 2006; New South Wales Government, 2007), and have average annual rainfall that is suitable for BFT establishment. Given that birdsfoot trefoil might be introduced to these six similar subcatchments and assuming that the benefits estimated for the KPS are typical of those that might be achieved in other similar subcatchments, the net benefits from the plant's introduction might increase from $0.25 million to $1.75 million at the lower end of the range of salinity benefits; from $8.8 million to $61.6 million on base assumptions; and from $33 million to $231 million at the high end of potential salinity benefits (Table 8.4). The potential area of invasion is still limited to the Border Rivers Catchment, but the potential area of invasion falls by half because of the expanded area where the plant is introduced intentionally (and recalling that we assume the plant is not considered invasive in subcatchments where it is introduced intentionally). As a result, the estimated costs of the invasion of external areas in the BRC also fall.

Table 8.4 Net benefits of introduction under uncertainty ($'million NPV) - *Multiple subcatchments in the Border Rivers Catchment*

		Benefits of Reduced Salinity (in 7 Subcatchments with assumed similar size and land use)		
Costs of Invasiveness		**Low**	**Base**	**High**
percentile		$1.75	$61.60	$231.00
0.05	$0.22	$1.53	$61.38	$230.78
0.10	$0.24	$1.51	$61.36	$230.76
0.50	$0.32	$1.44	$61.29	$230.69
0.90	$0.42	$1.33	$61.18	$230.58
0.95	$0.45	$1.30	$61.15	$230.55

Shaded results highlight positive net benefits and therefore that entry should be approved in these cases.

In this case, the plant's introduction would be approved in all instances and at all levels of confidence with respect to potential costs of plant invasion. As shown in Table 8.4, even at the lower end of the range of potential benefits and with 95 percent confidence of estimated costs, there are expected net benefits of $1.3 million. This analysis illustrates the extent to which BFT offers net benefits as the area where it may contribute to salinity reduction, increases.

In the above analysis, the potential for an increased number of dispersals or larger dispersals annually, because of the increased area from which seeds might be dispersed, is not considered. With a greater number of hectares planted to BFT throughout the Border Rivers Catchment, increased seed dispersal is probable. The relationship between the number of hectares of birdsfoot trefoil planted to pasture and the volume and number of dispersals is now investigated.

8.3.3 Multiple Dispersals and the Border River Catchment

The potential for the costs of birdsfoot trefoil's invasion of the BRC to exceed the benefits of its introduction to the KPS will be determined by the number of BFT seeds that find their way to natural areas and agricultural areas for which birdsfoot trefoil is not intended. So far in the analysis this has been largely ignored, primarily because of the difficulty in identifying values to describe this relationship. An attempt is now made to demonstrate this.

To be conservative with an introduction decision this analysis has been undertaken using the estimate of cost for the 90th percentile of the distribution: 90 percent of the time, costs could be expected to be less than or equal to the estimates used. Table 8.5 presents the estimated benefits for the KPS and a total of seven subcatchments (i.e. KPS and six others) in the BRC with the estimated cost of invasiveness shown as the percentage of seeds dispersed increases. The number of seeds produced annually was estimated on the basis of birdsfoot trefoil plants producing 2,875 seeds annually and with 8 adult plants/m^2 of pasture. On the basis of 36,000 hectares of birdsfoot trefoil, the number of seeds produced for the subcatchment is estimated as 2,875 x 8 x 10,000m^2 x 36,000.

Table 8.5 Net benefits of introduction under uncertainty ($'million NPV) – *by percentage of seed dispersed from (a) Kings Plain Subcatchment and (b) seven subcatchments*[a]

	(a) Kings Plain Subcatchment				(b) seven subcatchments			
Percentage of seeds dispersed (%)	Costs	Benefits			Costs	Benefits		
	90th percentile	Low $0.25	Base $8.80	High $33.00	90th percentile	Low $1.75	Base $61.60	High $231.00
0.50	$1.34	-$1.09	$7.46	$31.66	$19.39	-$17.64	$42.21	$211.61
0.75	$2.75	-$2.50	$6.05	$30.25	$29.70	-$27.95	$31.90	$201.30
1.00	$4.08	-$3.83	$4.72	$28.92	$39.60	-$37.85	$22.00	$191.40
1.25	$5.43	-$5.18	$3.37	$27.57	$49.20	-$47.45	$12.40	$181.80
1.50	$8.17	-$7.92	$0.63	$24.83	$60.23	-$58.48	$1.37	$170.77
2.00	$10.7	-$10.45	-$1.90	$22.30	$80.34	-$78.59	-$18.74	$150.66
3.00	$16.6	-$16.35	-$7.80	$16.40	$122.02	-$120.27	-$60.42	$108.98
4.00	$22.14	-$21.89	-$13.34	$10.86	$159.92	-$158.17	-$98.32	$71.08
5.00	$27.39	-$27.14	-$18.59	$5.61	$194.75	-$193.00	-$133.15	$36.25
6.00	$34.05	-$33.80	-$25.25	-$1.05	$225.76	-$224.01	-$164.16	$5.24
7.00	$38.10	-$37.85	-$29.30	-$5.10	$257.78	-$256.03	-$196.18	-$26.78
8.00	$45.70	-$45.45	-$36.90	-$12.70	$284.08	-$282.33	-$222.48	-$53.08
9.00	$50.29	-$50.04	-$41.49	-$17.29	$308.50	-$306.75	-$246.90	-$77.50
10.00	$55.28	-$55.03	-$46.48	-$22.28	$327.58	-$325.83	-$265.98	-$96.58

Shaded results highlight positive net benefits and therefore that entry should be approved in these cases

Note Dispersal of 10,000 seeds per hectare has been used in previous analysis. On a basis equivalent to that shown in this table, 10,000 seeds per hectare is equivalent to dispersal of 0.15% of seeds produced in the KPS and to dispersal of 0.01% of the seeds produced in seven subcatchments. These percentages are outside of the range of results shown here and so the cases where BFT would be introduced even under low expectations for salinity benefits (Table 8.4) are also not shown.

[a] At the 90th percentile for invasion costs.

The results of this analysis illustrate that:

- If expectations are for fewer than 2 percent of the seeds produced in the Kings Plain Subcatchment to be dispersed annually, then under the Base Case, the plant should be introduced; and
- If expectations are for fewer than 6 percent of the seeds produced in the Kings Plain Subcatchment to be dispersed annually, then under expectations for benefits at the high end of the range, the plant should be introduced.

Similarly,

- If expectations are for fewer than 2 percent of the seeds produced in the seven subcatchments to be dispersed annually, then under the base assumptions, the plant should be introduced; and
- If expectations are for fewer than 7 percent of the seeds produced in seven subcatchments to be dispersed annually, then under expectations for benefits at the high end of the range, the plant should be introduced.

These breakeven points are shown in Figure 8.5, and illustrate that as expectations of salinity benefits increase, the breakeven percentage of seeds dispersed increases. If the decision maker is less conservative and adopts a 70 percentile estimate of costs, then these curves would also shift up, as would the corresponding breakeven percentages of seeds dispersed.

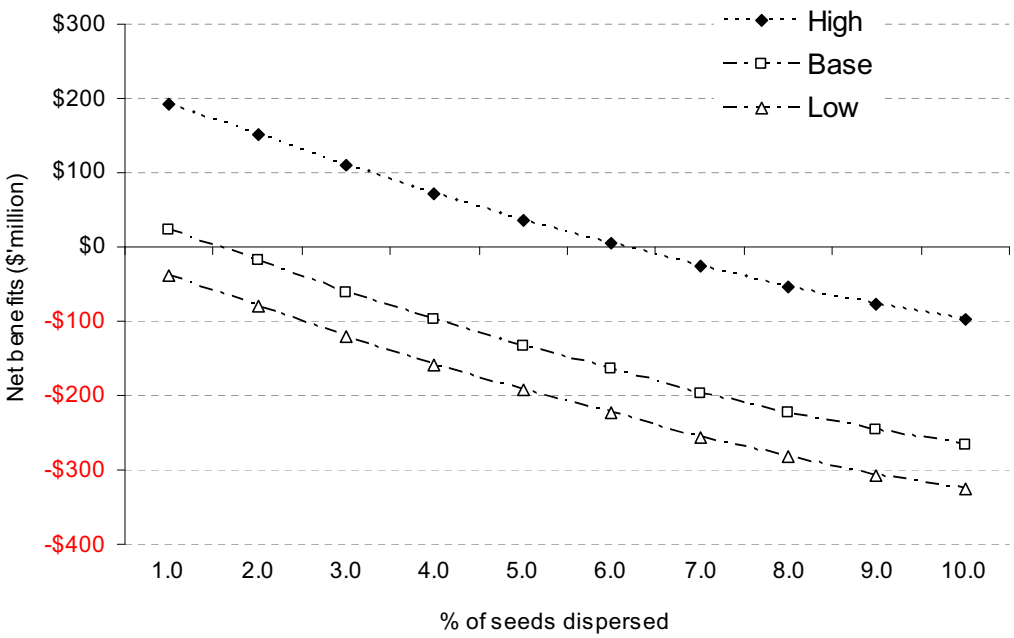


Figure 8.5 Net benefits of introduction to seven subcatchments in the Border Rivers Catchment – *90th percentile cost estimate and low, base and high expectations for benefits*

Given that in the order of 230 million seeds might be produced annually per hectare (8 plants per m^2, 2875 seeds per mature plant) even small percentages of the total number of seeds represent large dispersals. For example just 0.05 percent of the total number of seeds is 115,000 seeds, and this would need to be dispersed from each and

every one of the 36,000 hectares of planted pasture every year, for the estimated costs of $1.34 million to be realized at the 90th percent confidence level (Table 8.5).

To further put this in context, under the base assumptions each year some 869 billion birdsfoot trefoil seeds (i.e. more than 1.5 percent of all produced)[19] would need to be dispersed from the seven subcatchments before we could expect no net benefits. That is the equivalent to an average of 1.38 million[20] birdsfoot trefoil seeds arriving in each hectare of the surrounding catchment every year. The analysis previous to this section was undertaken on the basis of 10,000 BFT seeds arriving in each hectare of the surrounding catchment annually: this is equivalent to 0.15 percent of seeds produced in the Kings Plain Subcatchment and just 0.01 percent of seeds produced in the case of seven subcatchments.

8.3.4 National net benefits

A decision to introduce birdsfoot trefoil to the Border Rivers Catchment is a decision to introduce it to Australia. From the analysis in this study, it is possible to make a generalized judgement with regard to the net benefits on a national scale. This requires the assumption that the Border Rivers Catchment is a representative catchment across the entire area of potential introduction.

The estimated base-case net benefits where BFT is introduced to seven subcatchments was extrapolated to a national level as shown in Table 8.6. The extrapolation results show that net benefits could exceed $0.5 billion over 50 years where just 0.50 percent of seeds produced are dispersed annually. On an annualised basis this is $10 million per year. As shown in Table 8.6, where more than 1.5 percent of seeds are dispersed annually, there are estimated net costs.

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[19] 1.5 percent of the total (where the total is 230 million seeds per hectare multiplied by 36,000 hectares per subcatchment and multiplied by 7 subcatchments) is 869 billion seeds.

[20] 869 billion seeds divided by the 650,000 hectares of agricultural land considered external to the seven subcatchments where birdsfoot trefoil is planted.

Table 8.6 Net present value of benefits of introduction under uncertainty ($'million present value over 50 years) – *extrapolation to national estimates using the 90th percentile cost estimate*

Percentage of seeds dispersed (%)	Net Benefits of Introduction to the Border Rivers Catchment *(7 subcatchments)* [a] ($'million)	Net benefits [b] ($/ha)	National net benefits [c] ($'million)
0.50	$42.2	$167.50	$509.2
0.75	$31.9	$126.59	$384.8
1.00	$22.0	$87.30	$265.4
1.25	$12.4	$49.21	$149.6
1.50	$1.4	$5.44	$16.5
2.00	-$18.7	-$74.37	-$226.1

Shaded results highlight positive net benefits and therefore entry should be approved in these cases.

[a] Net benefits estimated for the base assumptions with seven subcatchments in the Border Rivers Catchment. [b] Per hectare benefits estimated as net benefits divided by the total number of hectares planted to birdsfoot trefoil in the target area as per the optimal solution for 4 metres (i.e, 36,000 ha per subcatchment). [c] Total benefits estimated on the basis of a total target area of 8 million ha (Ayres *et al*. (2007) multiplied by the proportion of this area that might be sown to the pasture (using the optimal proportion estimated in the DP model at the steady state (38 percent)) and multiplied by the net benefit per hectare estimated for the Catchment when the plant is introduced to seven subcatchments.

This extrapolation takes the decision criterion to the national level and provides an idea of the magnitude of potential net benefits assuming that the case study area is representative of all potential areas of introduction and that seed dispersal is consistent throughout all areas.

8.4 Other approaches to the Decision to Introduce

On the basis of the criterion developed in this study, the new varieties of birdsfoot trefoil would be introduced depending on the states of nature that prevail with respect to salinity emergence, plant invasiveness and seeds dispersed into external areas. The analysis provides a range of costs and benefits, but not the expected costs and expected benefits, because there is no information to specify the probabilities for the range of states of nature.

Without information on these probabilities, the analytical framework presented provides a basis on which decision makers can select an action based on their own attitude to risk and their tolerance of associated outcomes. If there is no knowledge at all, or decision makers are not confident that they have sufficient knowledge to define the probabilities of alternative outcomes, some simple decision criteria can be used to assess the situation (Lawrence and Pasternack, 2002) including the Maximin Criterion, which reflects a precautionary risk attitude, and the Maximax Criterion, which reflects an optimistic risk attitude. These are now applied to the case of introducing birdsfoot trefoil to the Kings Plains Subcatchment and the analysis continues to be for costs at the 90th percentile.

In the first instance these criterion are applied to the introduction decision where the number of seeds is the variable state of nature. The summarised results are shown in Table 8.7.

Table 8.7 Introduction decision based on Maximin and Maximax Criteria for low, base and high expectations of benefits – *where the % of seeds dispersed is the variable state of nature*

	Pessimistic Criterion	Optimistic Criteria
Performance of Plant in Subcatchment (Benefits)	Maximin	Maximax
Low	D	D
Base	D	I
High	I	I

D= do not introduce, I= Introduce. See Appendix H for accompanying payoff tables.

The decision to introduce under these alternative criteria is shown to be consistent with the findings of this study in that:

- When benefits in the subcatchment are low ($0.25 million), the plant *should not* be introduced from both an optimistic and a pessimistic perspective.
- When benefits in the subcatchment are high ($33.00 million), the plant *should* be introduced from both an optimistic and a pessimistic perspective.

- When there are base expectations for the benefits of the plant in the subcatchment ($8.80 million), the introduction decision will depend on whether a decision maker is optimistic or pessimistic regarding seed dispersal. Specifically this means that it will depend on the decision maker's beliefs about the proportion of seeds that will be dispersed.

In the second instance, the performance of the plant in the Kings Plain Subcatchment is considered the variable state of nature. The Maximin and Maximax criteria have been applied and the results are shown in Table 8.8. The decision to introduce is shown to be the same regardless of the optimism of a decision maker and dependent on the proportion of seeds that are dispersed. Under both criteria, where there is a high proportion of seeds dispersed the optimal decision is to not introduce and under a small proportion of seeds dispersed it is optimal to approve the introduction.

Table 8.8 Introduction decision based on Maximin and Maximax Criteria where 0.5, 2 or 10% of seeds are dispersed– *where the performance of plant in subcatchment is the variable state of nature*

	Pessimistic Criterion	Optimistic Criteria
Seeds Dispersed (% of those produced)	Maximin	Maximax
0.5%	I	I
2%	I	I
10%	D	D

D= do not introduce, I= Introduce. See Appendix G for accompanying payoff tables.

This analysis:

a) further iterates the importance of knowledge of the number of seeds dispersed for the decision to introduce a plant such as birdsfoot trefoil;
b) shows there will be cases under which it is economically rational to introduce the plant even when applying a precautionary / pessimistic criterion; and
c) illustrates that decision criteria that are optimistic or pessimistic may still present conflicts in decision making when there is more than one uncertain state of nature.

8.5 Conclusions

From an analytical perspective the results presented in this chapter illustrate three key observations. Firstly, the importance of incorporating uncertainty into the consideration of whether a plant should be classified as a weed, and as such whether its introduction should be allowed. The wide range of potential benefits and costs illustrates that there are difficulties making introduction decisions on the basis of deterministic analysis alone.

Secondly, the analysis shows the significance of case by case consideration of different plants and potential areas of introduction. The wide range of benefits, in particular, will be largely driven by differences in subcatchment characteristics and the performance of the plant in different subcatchments.

Thirdly, the analytical approach provides the opportunity to consider policy that accommodates uncertainty.

Particular results include:

- The expected net present value of benefits is small. The present value of benefits in the KPS is estimated to be $8.8 million over 50 years at a discount rate of 7 percent. On an annual basis this is equivalent to just under $2 per ha per year for each hectare of agricultural land in the Subcatchment. However, benefits will depend on the performance of birdsfoot trefoil and hydrological characteristics of the subcatchments, such that benefits in other subcatchments may range from $0.05 per ha to $7.10 per ha on an annual basis.
- The net present value of costs is small. The present value of costs in the Border Rivers Catchment when there are an average of 10,000 seeds dispersed to the average hectare of the subcatchment per annum is estimated to be less than $0.58 million over 50 years at 7 percent discount, 90 percent of the time. Annualised, this cost is equivalent to less than $0.01 per ha per year for each hectare in the Catchment.
- The costs of plant invasiveness are small relative to the salinity benefits.

- On the basis of base assumptions for benefits in the KPS and the 90th percentile of costs, net benefits will be positive when less than 2 percent of seeds produced in the KPS are dispersed into external areas of the surrounding Catchment.
- Assuming dispersals are constant, the net benefits of the introduction for a Catchment will be higher where there are a greater number of Subcatchments, within the Catchment, where the plant can be introduced to reduce recharge.
- Agricultural landholders will bear the majority of any costs if the plant is invasive in this case. On the assumptions used in this analysis, the costs in natural areas are small because of a) the relatively smaller area of natural areas at risk, b) the lower growth rate of the invasive plant population in natural areas, and c) the lower value placed on the outputs of natural areas. However, there are conditions under which the burden of costs may be borne in natural areas.
- The number of seeds dispersed is the key uncertainty on which the overall decision to introduce rests. No net benefits could be expected when relatively small percentages of all seeds produced are dispersed (i.e. more than 2 percent of seeds produced). However, these small percentages correspond to a large number of seeds that are known to have limited transfer vectors.

Together with the results of Chapter 7, the results presented in this chapter are now used to consider how decision makers might use the framework developed here to assess plant introductions and consider policies to accompany the decisions.

9 Policy Analysis

A range of results was reported in Chapters 7 and 8, and conclusions for approaches to the decision to introduce new plants and the nature of potential market failures drawn from them. In this chapter, these implications are brought together to address policy questions concerning the introduction of plants such as birdsfoot trefoil to address the emergence of dryland salinity in Australia.

9.1 Issues that arise in relation to the introduction of plant-based solutions to salinity

The trade-off between a higher weed burden and increasing costs of salinity directly concerns the problem of external costs. A number of questions arise from the presence of these externalities. Broadly, these questions are: should the plant be introduced and how could the introduction be approached to minimise externalities?

As argued in this study and in the context of benefit-cost framework, a change such as the introduction of an exotic plant to reduce salinity, should be sanctioned where the net benefit of doing so (the net change in economic surplus, including positive and negative externalities) is positive. The introduction is then an efficient change in the economy. The change may however not be a Pareto efficient change. That is, it may make some persons better off while making others worse off (Randall, 1987). This is because the salinity-plant invasion problem is one of a trade-off between two public *bads* (higher weed burden *v* increasing costs of salinity). In this case there is added complexity because the realisation of both is uncertain.

The compensation test approach indicates that the change should still go ahead if the gain to the beneficiaries is so large that they could compensate the losers (Chapman, 2000). Coase argued that this would happen and produce an efficient outcome when there is a market characterised by (Coase, 1960; Kolstad, 2000): perfect information; price-taking consumers and producers; a costless court system available to resolve disputes; utility-maximising producers and consumers; no wealth or income effects which would impinge on an exchange; and no transaction costs associated with an

exchange. These conditions are unlikely to hold due to the spatially and temporally disparate costs and benefits associated with invasive plants, salinity and the control of both. This is primarily because of the lack of perfect information, likely wealth and income effects, and transaction costs associated with a large number of parties (i.e. many individual landholders).

Establishment of property rights (either implicitly or explicitly), assistance to minimise the transaction costs associated with exchanges, or imposition of taxes in the Pigouvian tradition (Kolstad, 2000; Pigou, 1962), are opportunities for the government to circumvent these conditions and achieve an efficient and equitable outcome (Chapman, 2000). Such approaches require the government to recognise a polluter-pays or beneficiary-pays approach, a decision which is largely a product of political factors in any case (Pannell, 2008), and is even more challenging when there is a tradeoff between two public *bads*.

The importance of policy to accompany an introduction decision will largely depend on the relative value of the external costs. That is, when external costs are small, political factors may result in a policy of 'no intervention' being preferred. When they are large, a considered policy will have greater importance. When external costs are uncertain, it will be important to have a well designed policy because the costs *may* be large.

A decision to introduce a new plant ***should*** therefore not only be based on the relative expected benefits and costs associated with the introduction but also on consideration of relevant policies which accommodate the uncertain externalities.

So, from the broad questions surrounding an introduction and associated externalities, a number of specific policy questions arise:

- Is there a role for government to encourage the introduction of plants such as birdsfoot trefoil?
- Should birdsfoot trefoil be introduced?
- How can a government decide to introduce such plants when there are uncertain externalities?

- Is there a policy that could accommodate the uncertain external costs resulting from a decision to introduce a plant?

Each of these questions is now addressed in the following sections.

9.2 Is there a role for government to encourage birdsfoot trefoil adoption?

There has been a range of contributions to the relevance of government in the dryland salinity management problem (Greiner and Cacho, 2001; Mullen, 2001; Pannell *et al.*, 2001; Stoneham, 2004; van Buren and Pannell, 1999). This section considers whether the government should be involved in encouraging the adoption of plant-based solutions to salinity.

If we consider just the subcatchment where the plant is introduced, the analysis shows net benefits results from the adoption of birdsfoot trefoil, and as such an increase in welfare from such an introduction. The benefits are, however, potentially insufficient as an incentive for landholders to undertake this action individually.

The annualised per-hectare benefit is under $2.00 in the Kings Plain Subcatchment. On average, this represents a benefit of less than $940 per annum for an average private landholding of 470 hectares in the subcatchment. The present value of these benefits has been estimated on the basis of the financial returns. *Private benefits* are more encompassing than just financial returns and are influenced by factors such as risk, complexity of adoption, social considerations and personal attitudes to the environment (Pannell, 2008). As such, the private benefit may in fact be higher or lower than $940 depending on these factors, and will most certainly be different for different landholders. It is a value that could be higher if a landholder gets utility from implementing a practice that may help the local community or environment at large. However, it is also a value that could very easily approach zero if a landholder has additional barriers to adoption such an entrenched approach to farming practices. In this case, a private landholder is unlikely to introduce the plant to their farming system.

As mentioned, risk will be one factor which will influence a landholder's perceived value for a particular change in practice. Time preference for financial returns earlier in a planning period and concern as to whether the benefits will be realised will further reduce the perceived value of the plant's introduction to a landholder. Employing a discount rate of 10 percent, which reflects a higher preference for financial returns earlier in the planning period, reduces the annual average farm benefit from $940 to $550. On this basis alone there is likely to be insufficient adoption to maximise welfare because society has a lower discount rate than individuals.

There is another issue which may further reduce the private benefits of the new plant from the view of individual landholders. If a landholder knows, by virtue of their location in the subcatchment, or is uncertain that the introduction will reduce salinisation on their farm, their assessment of the private benefit will be lower. Again, the landholder may have a reduced incentive to introduce the new plant to their farm.

The implication is that, while private benefits are expected to occur within the subcatchment, the financial benefits as well as the broader private benefits may be sufficiently low that individual private landholders will not introduce the plant even if it is available to them. The two factors that produce this outcome are the separation of costs and benefits over time and over space. That is, the fact that benefits may accrue far into the future, or may not be captured on the landholder's land, makes the benefits of adopting the new plant not sufficiently attractive, even though this action would increase total welfare for all private landholders as a group.

Mullen (2001) states that the divergence in interests of farmers and the community suggests a role for government in managing salinity. Such a divergence is evident where the benefits of an action accrue sufficiently into the future. On this basis, and if the benefits of salinity intervention exceed the benefits of intervention elsewhere, a role for government in encouraging the adoption of new plant-based solutions to salinity would be warranted. There is also a divergence spatially. If the benefits of adoption of the new plants are captured by the landholder who adopted the plant then, it is argued, there is no role for the government in encouraging adoption. If, however,

the benefits are captured by surrounding landholders instead of the landholder who instigated the change then, it is argued that, government action is warranted.

Groundwater systems are complex and highly uncertain systems. Uncertainty regarding the timing and location of benefits arising from specific actions is inherent. In the interest of increasing welfare then, there appears to be a role for government in encouraging the adoption of new plants to mitigate salinity. Intervention in this sense would need to reduce the costs of adoption (extension to reduce learning costs, subsidies to reduce establishment costs etc). There is some evidence that the degree to which salinity is an externality varies across Australia (Pannell *et al.*, 2001); on this basis the extent to which such intervention is warranted may also vary by location.

9.3 Should birdsfoot trefoil be introduced?

The government's decision to encourage an introduction cannot be made from the perspective of just the benefits in the Subcatchment. The choice of policy response depends on the relative value of private benefits and public (or external) benefits (Pannell, 2008). Recognising this, and on the principle that the policy response should maximise net benefits, Pannell has developed a map for selecting the most appropriate policy response to natural resource management issues. The map of suggested policy responses is shown in Figure 9.1 and is applied now for further consideration of whether the government should allow the introduction of plants such as birdsfoot trefoil.

The map shows public (or external) benefits, on the vertical axis, and private benefits, on the horizontal axis, with the axes crossing at zero: the point that represents the existing situation prior to government intervention. The axes produce four quadrants (I-IV) which represent different classes of relative public and private benefits. Outcomes that lie to the right of the diagonal line AB generate positive net benefits overall (public plus private), while those to the left generate negative net benefits overall.

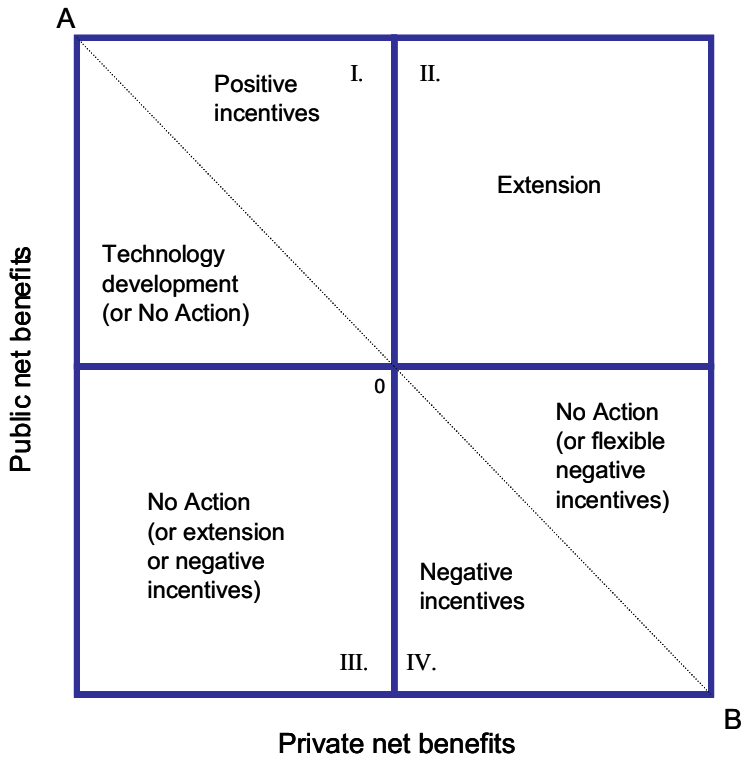

Figure 9.1 Map of suggested policy responses to different levels of public and private benefits (Pannell, 2008)

The recommendations for each quadrant are based on rules that are intuitively consistent with economic efficiency. For example, in quadrant III of the map, both public and private net benefits are negative; such that there is no incentive for private landholders to undertake actions that will produce outcomes in this region of the map *and* there is no need to encourage landholders because there is no public benefit to be gained. Potential changes in land management that fall into this quadrant require no policy response. The exception may be where private landholders are not fully informed such that extension or negative incentives may be required.

By further example, outcomes in quadrant I have positive public net benefits, but private net benefits are negative. Here there is no incentive for private landholders to undertake actions to achieve the outcomes, and policy intervention is warranted to achieve the public net benefits. In this case, when the public benefits of an action are greater than the private costs, then it is efficient to provide Positive Incentives (e.g. financial or regulatory instruments), but when the private net costs are less than the

public net benefits, then No Action or Technology development may be warranted. A full list of the rules and development of the map is provided by Pannell (2008).

The private net benefits and public net benefits as defined by Pannell can be considered analogously to the net benefits estimated for the Subcatchment (B) and the external costs estimated for the Catchment ($C_A + C_N$) in this analysis respectively. We can therefore consider whether and how government should intervene from a Catchment perspective using this framework.

The estimated Subcatchment private benefits and the Catchment external costs found in this study place the action to introduce a new plant such as birdsfoot trefoil somewhere in quadrant IV in Figure 9.1 because there are positive private benefits and negative public (external) benefits. As such, the efficient government policy would be either to introduce Negative Incentives (e.g. legislation to ban an action, an environmental tax) or undertake No Action[21]. The best action for introductions to the left of line AB in the quadrant, where external costs are greater than private benefits, is a Negative Incentive. This is consistent with the decision criterion in this study that such introductions should not be sanctioned (i.e. legislation against introduction is one (extreme) form of Negative Incentive) when costs exceed benefits. The best action for introductions to the right of line AB is No Action: that is, do not stop introduction. If other public benefits, not incorporated in the model in this study, were identified and found to be sufficiently large, the introduction may fall in the quadrant II. As prescribed in Figure 9.1 the appropriate policy response would then include extension.

Pannell recognises that the map is based on the premise that both public and private land managers are certain about the net public benefits and net private benefits they face as a result of the actions (i.e. they know where the option places them on the map). Pannell also goes further to show that incorporation of factors like adoption lags and learning costs can help refine the selection map further, and finds that:

[21] Pannell also includes Flexible Negative Incentives as a possible course of action in this segment of the map. Examples of flexible negative incentives provided by Pannell include tradable pollution permits or a conservation auction. These instruments imply the assignment of property rights so the market, rather than the government, can take action to identify the optimal level of an activity which has private benefits and public external costs.

1. the selection of cost-effective environmental projects is generally more sensitive to private net benefits than public net benefits, such that environmental managers should equally consider both the levels of public and private net benefits; and
2. as the required benefit-cost ratio for projects increases, so too does the number of projects for which no government intervention is most efficient.

Pannell's analysis and first conclusion supports the proposal that equal consideration should be given to private benefits and public benefits in relation to plant introduction decisions (Kalisch Gordon, 2004). It is also consistent with the framework developed and demonstrated in this study. The importance of the second finding is that by implication, the greater the required level of certainty (i.e. *expected* net benefits), the more likely that no government intervention to encourage a plant's introduction will be the most efficient course of action: a pertinent point but unlikely to always be obvious for decision makers.

The most efficient policy response with respect to plant introduction decisions will depend on the relative size of the costs and benefits, or on which side of line AB the action to introduce a new plant would fall. However what happens if the government is faced with outcomes which are inherently uncertain such that they cannot be sure which side of the AB line action will result?

9.4 How can a government decide in the presence of uncertainty?

The present value of benefits from an introduction such as birdsfoot trefoil to a subcatchment such as the Kings Plain Subcatchment of northern New South Wales is positive: the present value of benefits over 50 years at 7 percent is estimated to be $8.8 million over 93,145 hectares. From the perspective of the subcatchment treated as one single private landholder, such a plant should therefore be introduced because the present value of benefits is positive.

The government, however, needs to make a decision based on the impact on society as a whole, which in this case is represented by the wider Catchment; the net present

value of benefits from the introduction considered in this study is positive when the proportion of seeds dispersed into the wider Catchment is low. So the plant should be introduced when the number of seeds expected to be dispersed into external areas is low.

The mean net present value of benefits over 50 years at 7 percent is estimated to be $8.1 million if the plant were introduced to just one subcatchment and with a dispersal of 10,000 seeds into each external hectare annually. If the plant were introduced as a pasture to seven similar subcatchments the net present value of benefits, under base case benefits and costs at the 90th percentile, is estimated to range from $1.37 million to more than $42.21 million if less than 2 percent of seeds produced in the subcatchments are dispersed into the Catchment (Table 8.5). If more than 2 percent of seeds produced are dispersed into the surrounding Catchment, then the plant should not be introduced. Similarly, the estimated net benefits of introduction from a national perspective exceeds $500 million over 50 years at 7 percent when 0.5 percent of seeds produced are dispersed into external areas and falls to $16.5 milion when 1.5 percent of seeds are dispersed. When more than 2 percent of seeds are dispersed into external areas, there are no net benefits from the plant's introduction (Table 8.6).

Given the uncertainty about the number of seeds that will be dispersed and the benefits that will be realised where the plant is introduced as a pasture, and based on the analysis in this study, there is the opportunity to examine the level of certainty associated with the realisation of the external costs. The information provides a practical decision framework analogous to determining an optimal level of an externality (see Figure 4.6). This is illustrated in Figure 9.2, which shows the distribution of the present value of costs of a plant's introduction, where 10,000, 15,000 and 20,000 seeds (equivalent to 0.15 percent, 0.23 percent and 0.30 percent of seeds produced in the subcatchment respectively) are dispersed annually to external areas. Point ***a*** identifies where benefits are greater than or equal to the costs 90 percent of the time for the case where 10,000 seeds are dispersed into the external areas each year. Points ***b*** and ***c*** identify the same level of confidence when 15,000 and 20,000 seeds respectively are dispersed into external areas. Points ***d***, ***e*** and ***f*** identify the same respective decision points where a lower level, 70 percent, is the required level of confidence with respect to net benefits.

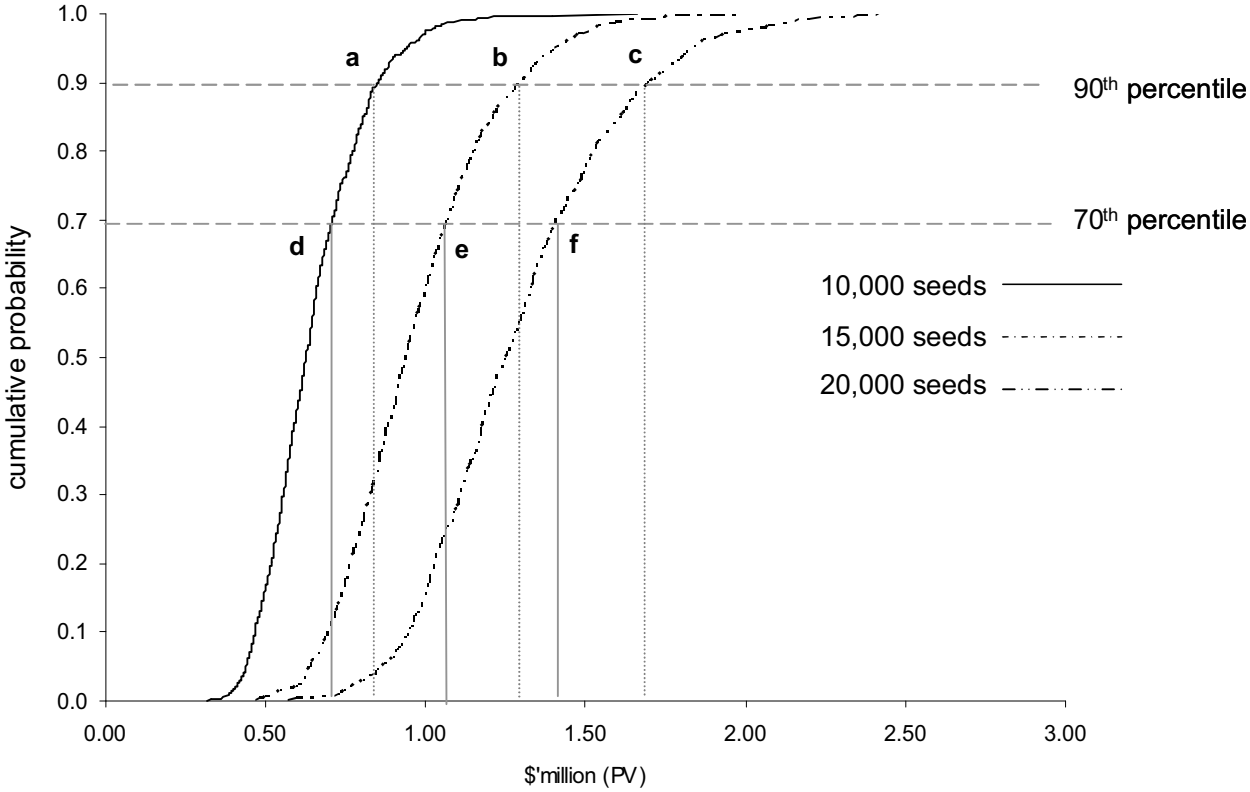

Figure 9.2 Decision Points based on the present value of benefits in the Kings Plain Subcatchment and costs in external areas of the Border Rivers Catchment – *90 and 70 percent confidence of net benefits*

With this decision framework, decision makers can identify a level of confidence that they are comfortable with and proceed with a decision that satisfies that desired confidence given their knowledge of seed dispersal. For example, the government could require a 90 percent confidence level. If 15,000 seeds are expected to be dispersed into external areas annually, then the present value of benefits arising from the introduction must be a minimum of $1.3 million (point ***b*** on the graph). By further example, if the requirement is for 70 percent confidence, the present value of benefits must be at least $1.05 million (point ***e***).

The need to set agreed threshold levels of risk, or uncertainty, with respect to plant introductions has been documented (Paynter *et al.*, 2003) and this provides a framework for doing so and also complements the decision criteria established in this study.

The approach provides decision makers with a way to accommodate uncertainty within their decision to introduce a new plant that offers potential benefits but also potential costs. The decision is based on there being confidence of net benefits, and as such confidence that there will be an increase in society's welfare. Such a decision is efficient, but by taking this approach the government accepts that there may be externalities resulting in some made worse off by the action to introduce a new plant. The following section presents policy approaches to accommodate those made worse off by a new plant introduction.

9.5 What policy measures could accommodate uncertain externalities?

To further consider the problem of uncertain externalities the problem is again considered within the constructs of Pannell's map. Then, possible approaches to policy to accommodate externalities are discussed.

Figure 9.3 is quadrant IV from Figure 9.1 adapted to show the benefits of the introduction of birdsfoot trefoil to the Kings Plain Subcatchments in the Border Rivers Catchment (B = \$8.80 million under the base case) and the 90th percentile costs of the invasion into external areas (C_A+C_N dependent on the number of seeds dispersed, from Table 8.5). The benefits increase across the horizontal axis from the origin 0. Costs are uncertain and increase with the proportion of seeds dispersed. These are shown to increase down the vertical axis consistent with Pannell's map where external costs are negative public benefits costs. The corresponding proportion of seeds dispersed is shown on the right vertical axis.

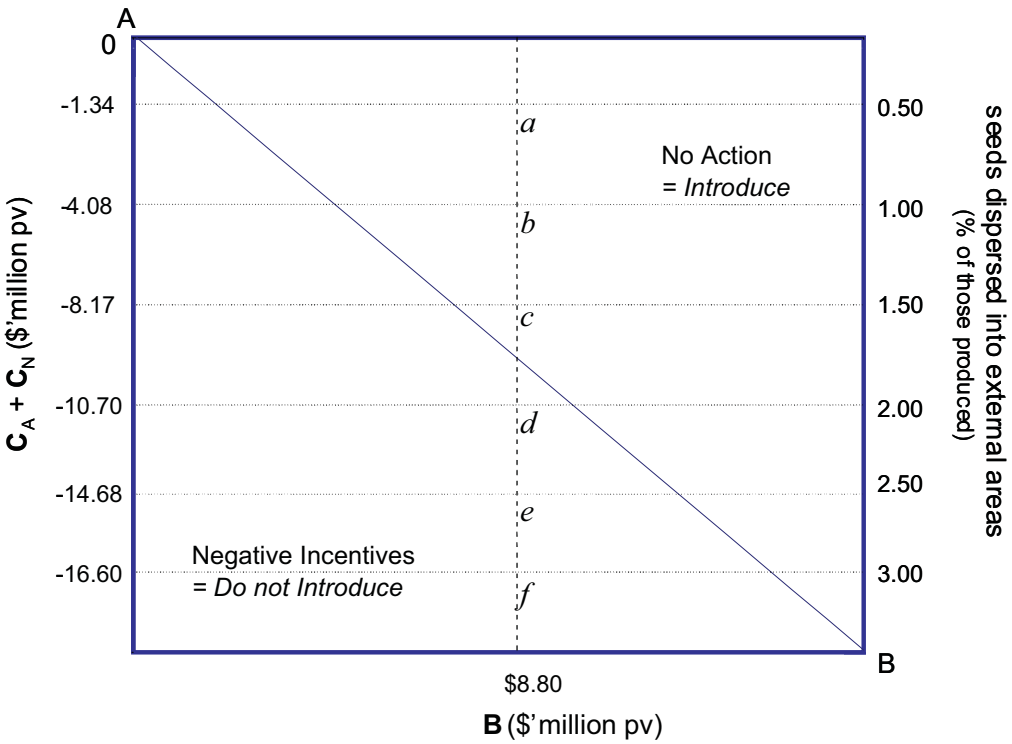

Figure 9.3 Incorporating uncertainty into selection of policy – *the case of introduction to the Kings Plain Subcatchment*

According to Pannell, No Action would be most appropriate option for introductions that yield the outcomes *a* through *c* shown in Figure 9.3, and Negative Incentives would be appropriate for outcomes *d-f* and beyond. In terms of the decision to introduce birdsfoot trefoil, this corresponds to introduce the plant for outcomes *a – c* and do not introduce the plant for outcomes *d-f* and beyond. The decision therefore depends on the factors the influence that proportion of seeds that are dispersed such as the dispersal mode of the plant, environmental effects *and* the level of precaution taken by landholders to limit seed spread.

In this framework, even though society's welfare is increased from an introduction that yields outcomes *a* through *c*, the fact remains that some people will be made worse off. Further, because the introduction has been undertaken on the basis of the 90th percentile estimate of costs, there is still the potential for costs to be higher than estimated.

Imposition of liability, direct regulation, insurance and performance bonds are four options which can promote precaution and accommodate externalities by assuring

against the losses of those in external areas. Here they are applied in the context of promoting precaution that ensures the number of seeds dispersed does not exceed expectations. These options are considered in the context that the property rights for freedom from weeds lie with landholders in the external areas.

9.5.1 Liability

Liability has the effect of providing an incentive for precaution (Kolstad, 2000). In the case of landholders in a subcatchment who introduce a new plant, the imposition of liability means they would be responsible for costs realised if the plant invades agricultural or natural external areas surrounding the subcatchment. In essence, it internalises potential external costs. Therefore landholders who introduce the plant have an incentive to undertake activities that reduce the risk of spreading the plant.

Examples of such actions might include: the inclusion of buffer zones around areas planted to the new pasture (i.e. areas of land set aside between the pasture and external areas on which the new plant is not planted); cleaning farm equipment after it has been used on or around the new pasture or not relocating it to areas without the new pasture at all; not selling hay cut from the new pasture into external areas; and emptying out stock that have grazed on new pasture before transferring them into external areas. Liability compels an individual to undertake actions according to their own risk profile. Where the risk of realised costs in external areas is assessed to be sufficiently high, the precaution introduced by liability would mean landholders would not introduce the new pasture at all.

In principle, the imposition of liability has the capacity to invoke efficient resource outcomes (Tietenberg, 1989) . However, for liability to invoke the level of precaution that corresponds to the optimal change in welfare there are several key requirements. These include the existence of legislation and access to a Court system that is not cost-prohibitive. If there is no prosecution there is no incentive to take precaution. A further crucial factor is the ability to ascertain causal effect. It is difficult to establish who is responsible, then liability will not invoke the required precaution. Causal effect could be difficult to establish for a new plant invasion in external areas due to

its dynamic and uncertain nature as well as the large number of landholders potentially involved. Another factor is the ability of those found liable to pay damages (Tietenberg, 1989).

If the government encourages the introduction of a new pasture such as birdsfoot trefoil on the grounds of intervening to rectify a market failure (e.g. subsidy for establishment costs to reduce salinity), then it might be argued that the government is liable for any external costs that might result. With a single entity, the problem of establishing causal effect can be reduced. This approach provides an avenue for those made worse off to be compensated for the external costs incurred. There are, however, two problems that undermine this approach. Firstly, governments are generally indemnified in legislation against damages in any case. Secondly, with the government liable, there is no incentive for the many landholders to individually undertake the optimal level of precaution. Coupling direct regulation with liability is one approach that may promote a level of precaution that limits externalities and assures compensation of external costs.

9.5.2 Direct Regulation & Liability

Direct regulation requires government to specify standards and actions that may result in an increased level of precaution (Kolstad, 2000). In the case of new pasture plants, examples of regulation might include specification of the size and location of buffer zones, prohibition of the transfer of stock and equipment into external areas or establishing a standard number of days that stock must not graze on the new pasture before they can be transferred into external areas.

Direct regulation is underpinned by liability and as such still requires a court system and the ability to establish causal effect. Further, however, it also requires that the government has information to be able to specify standards and actions that will result in the required level of precaution. This approach is therefore costly to establish in terms of both setting and monitoring the regulations. Further, no single standard will effectively reflect the appropriate level of precaution for each different landholder, location and set of environmental conditions (Kolstad, 2000). As such, standards and

regulations will over prescribe precaution for some while under prescribing it for others. An approach that incorporates market forces such as insurance can, under some conditions, provide for the optimal level of precaution for each (Mehr, 1986).

9.5.3 Insurance

Risk insurance is widely applied in the agricultural industry (Hardaker *et al.*, 2004) and has been advocated for use in conjunction with liability with respect to environmental hazards (Kolstad, 2000). If applied in the case of plant-based solutions to salinity, those undertaking to introduce the new pasture plant would be able to transfer the risky prospect of being liable for potential costs in external areas into a sure payment, by taking out an insurance policy. This approach requires the conditions of liability (a court system and ability to establish causal effect) to be present but would provide assurance that loss in external agricultural and natural areas could be reimbursed, with the optimum level of precaution established through the market for insurance policies. As with direct regulation, insurance also has a significant information requirement (Mehr, 1986), but in the case of insurance it is left to the market via the insurance companies that calculate premiums. From this perspective it would be a superior option to promote precaution.

However, in addition to the conditions of liability and the information requirements of direct regulation, an effective insurance market also requires that the risk is sufficiently high to attract insurers, and that the premiums are attractive to those at risk of being liable. This implies a number of conditions for operation: losses must not be amenable to risk pooling, be clear and defined, occur in a well defined time period, occur frequently enough to allow premium calculation and not be prone to moral hazard or adverse selection (Kolstad, 2000). Each of these conditions presents an issue for the insurance of potential losses from a new introduced plant:

- Invasions may or may not be correlated and therefore may or may not be risk pooled. This will depend on whether the risk of spread results primarily from regional environmental factors or primarily from the operations of individual landholders (and as such is an extension of the problem of causal effect).

- Losses may not be clear. Losses of biodiversity, for example, may result following an invasion of the new plant but initial stress may have been the result of other earlier invasions or other environmental stresses (e.g. salinity, another plant or pest invasion, acidity).
- Losses may occur over many years. This might especially be the case where a plant is a 'sleeper weed' such that it shows invasive tendencies only after many years.
- There is unlikely to be enough loss events to allow calculation of a premium. Precedents for losses resulting from plant introductions are many, but discrete cases where losses have been estimated and are sufficiently similar to those caused by the new plants introduced are unlikely to be adequate for the required calculations.
- Moral hazard would be likely, because monitoring the precaution to limit the spread of a new plant taken by the potentially large number of landholders could be cost prohibitive.
- Adverse selection may undermine any insurance market for invasive plants; given that it is likely to be cost prohibitive to determine the risk profile of each potential applicant landholder and there being an insufficient history on which to base estimates.

Aside from the difficulties establishing causal effect which is going to be problematic for the introduction of new plants under any policy approach, the characteristics of the market (large number of landholders, few precedents) are likely to compound difficulties for the application of risk insurance to accommodate uncertain externalities.

9.5.4 Performance bonds

Performance bonds are a related form of insurance where guarantees or undertakings are provided by a financial institution in relation to a contract struck between a contractor and a government agency (NSW Treasury, 2008). The government is guaranteed sufficient funds, in the form of a bond or security, to cover costs in the event of a firm's failure to perform. Performance bonds are widely required by

government in contracts for goods and services provided and in the finance industry, to cover potential losses made as a result of financial services provided. Performance bonds have also been used in a number of environmental-protection applications in Australia. These applications include the mining industry to encourage land rehabilitation, pollution reduction programs and effluent control, and potential applications with respect to forestry activities and the lease of national parks and other public lands (James, 1997).

In the case of new plant introductions, those who introduce the plant would be required to post a performance bond and failure to perform, meaning that the plant invaded external areas, would result in forfeiture of the bond. The bond in this case would be available to undertake plant eradication or control programs in the external areas. Performance bonds have the potential to be more feasible as a policy to achieve optimal precaution and assure losses because:

- the number potentially made worse off is represented by a single government body who legislates the requirement and manages the eradication or control programs; and
- the market (via those institutions issuing performance bonds) determines the appropriate level of precaution for those introducing the plant and is responsible for obtaining information to determine premiums.

There remains the problem of causal effect when there are many different landholders who could be responsible for a particular invasion of external areas. In the event that landholders acted collectively as a regional industry body or within a Catchment Management Authority arrangement, the issue of causal effect could be limited. As such, there are conditions under which performance bonds may offer a policy to accommodate uncertain externalities. Of course, these conditions include that there be sufficient information for the market-determined price of a performance bond to appropriately reflect the risks, that the government be prepared to limit the performance period and that losses are clear. Whether these conditions can be met will vary with the different plants proposed for introduction, the location of introduction and the government's attitude to risk.

9.5.5 Selecting an approach

The difficulty with the application of policy to accommodate uncertain externalities is borne primarily out of the potentially large number of landholders involved (such that the externalities are not borne from a point source), the low frequency of precedents to observe risks and the difficulty establishing causal effect. Given these difficulties, policy approaches will be problematic in identifying the optimal level of precaution and, potentially, not cost effective even when the introduction is otherwise efficient.

However, because the optimal level of precaution cannot be guaranteed and not all externalities will be accommodated, that it is not to say that no action should be taken. Any policy position must be considered in terms of the net benefits of the action with respect to the net benefits of alternative actions (with respect to this or other competing issues). Further, there are precedents in other areas which demonstrate the use of these policies.

With respect to the production of genetically-modified crop varieties, for example, a system of direct regulation including licensing contingent on operating standards and a system of monitoring and compliance is operational (OFTR, 2008). This application recognises that there are externalities which warrant precaution and, through the licensing requirement, internalises to some degree the potential externalities through added costs of operation. The preparedness of the government to fund the development of standards and undertake monitoring recognises the potential public benefit from imposed precaution. In the case of new plant-based solutions to salinity, government contributions may have further justification where there are public as well as private benefits from the introduction of a new plant (in this case where salinity is an externality within the subcatchment).

Direct regulation of genetically-modified crop production means the level of precaution established in standards will in some cases be too high, in others too low and overall potentially not optimal. Application of this approach to new plant-based solutions to salinity would necessarily be under the same caveat. If an insurance or performance bond approach could be applied, this would not only promote a level of precaution closer to the optimal level and thus enhanced welfare outcomes in itself,

but also allow the government to reduce its involvement in the issue and channel funds into alternative welfare enhancing activities.

The conditions required for the application of a performance bond or insurance approach could be enhanced with increasing attention to the problem of externalities. This may come with greater coordination of natural resource management (e.g. expansion of the role of Catchment Management Authorities or regional coordination of landholder/production groups), increased values being placed on natural and agricultural areas (and conversely the greater perceived threat of new invasive plants) and with increased demand for solutions to natural resource issues that may also pose potential costs. Where collective action is likely, development of a performance bond approach would appear to be a better option to develop promote precaution and assure against losses. Where collective-action is not feasible or anticipated, then direct regulation, accepting that the standard level of precaution may not be optimal, would appear to be the superior option.

9.5.6 Research & Development

Government funding of agricultural research and development in Australia has a long history with a more recent emphasis on joint government and industry arrangements. Funding research as a policy option might accompany any of the above mentioned policy approaches. Research would be useful in three ways.

Firstly, research might include further breeding to develop superior varieties of the new plants with reduced likelihood of dispersal of viable seed into external areas. A drawback of this strategy is that a variety which produces fewer seeds or has a reduced seed bank life is unlikely to be useful as it would reduce the persistence, and thus benefits, of the plant as a pasture. Breeding which ensures that new varieties are particularly specific to one rhizobium may be one option appropriate to leguminous pastures such as birdsfoot trefoil. In the case of other plants in general, research and development may be directed to minimising the seeds amenability to dispersal.

Secondly, research funding could be directed to obtaining data that inform the policy-development process. This would include information that helps set the

standards for direct regulation, for example identifying the nature and size of buffer zones or the best practice for transfer of stock and machinery between areas with and without the new plant. This could also include data on risks to help establish an insurance or performance bond approach to policy.

The third approach would be research to identify alternative, cost effective ways to reduce recharge of watertables. This might include other plant-based solutions which do not present an invasion risk or completely different strategies such as engineering options.

9.6 Summary

The decision criterion developed in this study will help the government make introduction decisions which increase welfare. The decision to introduce birdsfoot trefoil in this case would increase welfare, however in some cases may reduce the welfare of others in the process.

Whether or not this occurs will be dependent on the dispersal of seeds and the existence of appropriate vectors. The frameworks developed provide the opportunity to make a decision with a level of certainty regarding costs, benefits and externalities. Further investigation of the probability of seed dispersal could be undertaken to inform the decision process with respect to birdsfoot trefoil specifically.

Such information, with respect to plants generally, could aid the development of policy based on promoting the optimal level of precaution. This might include direct regulation, or potentially an insurance or performance bond system, to accompany uncertain decisions. Difficulties foreseen with these approaches to accommodating externalities resulting from new plant introductions, means any policy approach will be enhanced by research and development. Such research and development effort could be applied to inform the policy development process, limit the realisation of costs and/or ensure that benefits realised are greater than costs realised, and ensure that external costs resulting from a plant invasion are not a burden to the public or external stakeholders.

10 Conclusion

A criterion, based on economic principles, has been developed to guide the decision whether to permit new plants to Australia. The criterion states that a plant is a weed when there are no net benefits from its introduction, and in particular when

> "…the expected costs associated with the plant's introduction, including expected negative externalities, EXCEED the expected benefits from the plant, including expected positive externalities."

When a plant is defined as a weed, it should not be introduced to Australia.

This criterion has been applied to the potential introduction of the recently developed new cultivars of birdsfoot trefoil (*Lotus corniculatus* L.): 'Phoenix', 'Venture' and 'Matador' (Ayres *et al.*, 2008). This has been implemented through the application of a bioeconomic model with two sub-models. The first was a dynamic programming sub-model to estimate the benefits of salinity reduction and the second a plant simulation sub-model to estimate the costs of plant invasiveness. The analysis illustrates the range of considerations required for such decisions and the results show the range of conditions under which introduction of birdsfoot trefoil, and new pastures in general, would offer net benefits and therefore should be permitted. The study offers a balanced and improved means of determining if new plants should be introduced, and policy approaches to this decision when costs and benefits, including externalities, result from dynamic and uncertain natural processes.

10.1 Summary of Results

The sub-models were applied firstly through a deterministic analysis. They described the benefits of reduced salinity in the Kings Plain Subcatchment of NSW and the costs of the plant's potential invasion of the surrounding Border Rivers Catchment. The results were brought together in a guiding benefit-cost framework consistent with the decision criterion that was developed.

The present value of benefits was estimated as the difference in the profit for the optimal solution with and without birdsfoot trefoil as a land-use option. The key state variable of the model was the watertable depth and dynamic programming was used to solve for the optimal state path over the analysis period. The benefits were found to vary with initial depth of the watertable. The most benefits were obtained for an initial watertable depth of 3.5 m. The benefits for the Kings Plain Subcatchment were estimated at $8.8 million over 50 years discounted at 7 percent. This represents an annual present value of under $2.00 for every hectare in the subcatchment.

The present value of costs was estimated as the reduced value of output when birdsfoot trefoil invades agricultural and natural areas of the Border Rivers Catchment. The present value of these costs in agricultural areas was found to be $44,000 over 50 years discounted at 7 percent for a single initial dispersal of 10,000 seeds per hectare in the surrounding Catchment, and $0.6 million for annual dispersals of 10,000 seeds per hectare in the surrounding Catchment. The present value of these costs in natural areas was found to be less than $1,000 for a single initial dispersal of 10,000 seeds per hectare in the surrounding Catchment, and less than $12,000 for annual dispersals of 10,000 seeds per hectare in the surrounding Catchment. The total cost in both agricultural and natural areas was found to be equivalent to less than $0.01 annually for every hectare in the surrounding Catchment.

The net present value of benefits was estimated as the difference between the benefits of reduced salinity in the subcatchment and costs of birdsfoot trefoil invasion in the surrounding Catchment. The net present value of benefits was found to be positive under the base assumptions such that the new varieties of birdsfoot trefoil should be sanctioned for introduction. The analysis revealed not only positive net benefits, but benefit-cost ratios of 200 and 15 for the case of a single initial dispersal and annual ongoing dispersals, respectively.

Analysis of the range of potential benefits in the subcatchment indicated the present value of benefits to possibly range from $0.25 million under unfavourable conditions to $33 million under favourable conditions. Incorporation of stochastic elements in the plant population simulation sub-model was then undertaken. Application of probability distributions of the present value of costs in agricultural areas, for the case

where 10,000 seeds are dispersed annually, resulted in a mean of $0.50 million and a range from $0.25 million to $0.80 million. The distribution of the present value of costs in natural areas was found to have a mean of $8,590 and a range from $5,450 to $15,990.

The net present value of benefits depends on the seeds transferred to the surrounding Catchment. In the case of annual dispersals, as might be expected where pastures are grown on an ongoing basis, benefits were found to be positive where the number of seeds dispersed into the surrounding catchment is less than 2 – 3 percent. The exception is under the very worst conditions for salinity reduction in the Subcatchment when net benefits were estimated to be negative unless no seeds were dispersed into the catchment. This analysis assumed that the new varieties of birdsfoot trefoil are twice as persistent as existing varieties and so represented a conservative assessment of the costs.

On the basis of the analysis undertaken the new varieties of birdsfoot trefoil should be permitted for introduction to the Kings Plain Subcatchment when the proportion of seeds dispersed is low. This holds for high confidence (90th percentile) of the value of expected costs.

10.2 Key Findings

The key findings of this study are as follows:

- The decision criterion provides a practical means of considering the introduction of new plants to Australia[22]. The framework can assist decision makers with the choice to allow entry even if there are unknowns with respect to plant population dynamics and dynamic groundwater systems.
- The impact of the new varieties of birdsfoot trefoil as new pasture options is to delay rising watertables and to allow landholders to more profitably operate when watertables are closer to the surface (Chapter 7). Benefits accrue from the combined impact of these effects.

[22] Discussion of the use of this approach in the context of the existing WRA is provided in Appendix I.

- In particular, birdsfoot trefoil increases the value of a change in watertable depth (shadow price) where the initial watertable depth is less than 4 metres and decreases the value of a change for initial depths between 4 and 10 metres. This results because of the change in productive value at the surface when the plant is introduced. For depths greater than 10 metres there is no change (Figure 7.3).
- The costs of the potential invasion of the plant in external areas have been found to be small (Chapter 7).
- The new varieties of birdsfoot trefoil should be introduced when the proportion of seeds transferred into external areas is low (Chapter 7 and Chapter 8).
- Incorporation of uncertainty within the decision criterion is important. Net benefits, and thus the appropriate decision to maximise welfare, can change when stochasticity is introduced (Chapter 8). The plant should be introduced when less than 3 percent of seeds are transferred into external areas on the basis of the deterministic analysis (Chapter 7). On the basis of the stochastic analysis the plant should be introduced with a 90 percent confidence when less than 2 percent of seeds are dispersed (Chapter 8).
- The number of seeds dispersed from the subcatchment is the key uncertainty on which the overall decision to introduce rests (Chapter 8) and can provide the basis for policy decisions and research focus (Chapter 9).
- Agricultural landholders will bear the majority of any costs if a plant such as birdsfoot trefoil is invasive in areas similar to the Border Rivers Catchment (Chapters 7 and 8).
- The benefits of the new varieties are small when considered on a per-hectare or farm basis and the private value of benefits in the Subcatchment may not be sufficient to encourage adoption (Chapter 9).
- Even if a decision to introduce is made on the basis of an expected increase in welfare some groups may be made worse off by the decision so there is a role for government to minimise the external costs, either through research or the imposition of liability on those who introduce the plant. There is potentially a role for performance bonds for introductions where collective action is possible or direct regulation where it is not. However, research and development to minimise uncertainty of plant dispersal and inform policy is

considered likely to accompany any decision to introduce such new plants. (Chapter 9).

- In the case of a problem of salinity and invasive plants, there is potentially a role for government to both encourage the adoption of plants such as birdsfoot trefoil and also minimise the external impacts of invasiveness (Chapter 9).

10.3 Contributions of this Research

This study provides some particular contributions to the management of plant introductions, salinity and to associated policy. These are accompanied more broad contributions to the field of natural resource management.

Firstly, this study contributes a decision criterion and associated definition of what constitutes a weed that can help resolve conflict over plant introductions. The basis for considering a weed has been redefined in a way which will accommodate all manner of plants and plant purposes. This provides a balanced and improved basis on which to decide whether new plants should be introduced to Australia, including those with the potential to contribute positive increases in welfare to some but also decrease welfare to others. The incorporation of damage functions in the estimation of plant invasion costs in agricultural and natural areas provides a practical application of the recent work of Hester *et al.* (2006) and a contribution to the way marginal losses are estimated with respect to plant populations.

To the management of dryland salinity, this study provides an examination of a non-salinity external impact of management. To date the primary focus of studies of the external impacts of salinity have been focussed on off-site salinity. This study examines non-salinity impacts that may accompany salinity management (i.e. invasive plants) and provides the basis for further investigation of the trade-off between two public goods with respect to salinity management. The analysis has also revealed key factors that can guide researchers when selecting future pasture plants for introduction. These are provided in section 10.4.

With respect to natural resource management and policy, this study has developed and applied a policy framework which allows decision makers to introduce a plant under uncertainty. Decision makers can use the framework to establish a level of confidence with respect to net benefits and allow new plants whose distribution of costs and benefits meet this level of confidence. Similarly, decision makers can set research objectives for the discovery, or development, of new plants for introduction that meet this level of confidence. Further, the study identifies policy options that might accommodate the uncertain externalities and the conditions under which they might be appropriate. Finally, the study has identified that while the criterion provides a way to determine if a plant should be permitted, there is still the potential for the government to be faced with a conflict with respect to policy.

More broadly, this study makes a contribution to the field of natural resource management. The analysis of two uncertain and dynamic natural resource issues within a single framework is presented and is done so in the context of two concurrent supply shifts within an economic surplus framework. Also, but more specifically, the problem of the emergence of an invasive plant population and salinity appears to have not been considered together before analytically.

10.4 Key factors to address when selecting future new plants

The analysis has provided some information to assist researchers select and develop future new plants for introduction. The focus of selection or development of new plants should be on characteristics that influence net benefits and minimise externalities.

With respect to leguminous pasture plants developed to reduce salinity, these characteristics includes maximising pasture production, minimising the amount of recharge that passes the plant root zone and ensuring that plant seeds are not well adapted to dispersal. The following provides some guidance with respect to selection and introduction of new plants which will offer net benefits:

- The highest benefits stand to be gained when the new varieties of birdsfoot trefoil are introduced to areas where the depth of the watertable is initially between 2 and 7 metres (Chapter 7).
- When the ability of the subcatchment to drain is high (i.e. discharge is high) and the watertable deep, the benefits of the new varieties diminish. In particular, when the watertable is around 10 metres and deeper, subcatchment discharge is negatively related to benefits (Table 7.3);
- Increasing the pasture production capacity and reducing recharge increases benefits considerably. The combined effect of increasing the pasture capacity of the new varieties of birdsfoot trefoil by 10 percent and reducing recharge by 10 percent is to increase the benefits in the subcatchment by as much as 33 percent (Table 7.2).
- There are significantly greater benefits to be captured when net recharge approaches zero (Figure 7.9). At initial watertable depths less than 2 metres there is potential for benefits to be gained from reversing the upward trend in watertable depth (Figure 7.10).
- In the event of sustained higher agricultural prices, there would be a significant increase in the benefits presented where initial watertable depths are greater than 1.7 metres (Figure 7.12) because of the higher productive value at risk from a rising watertable.
- The greater the expected benefits in the subcatchment, the higher the breakeven proportion of seeds that can be dispersed (Figure 8.5). The greater the proportion of an area where the plant can be introduced beneficially, the higher the proportion of seeds that can be dispersed and there still be net benefits (Table 8.5).
- Broadly, the *private benefits* of the new varieties are likely to be low so that adoption would benefit from a program of extension (Chapter 9).

There is a range of features of plants such as birdsfoot trefoil that researchers could focus on to minimise the costs that 'escaped' populations impose in external areas. However, each of these brings with it the potential to reduce such a plant's capacity to be a useful pasture plant. For example, varieties could be selected for lower productive capacity (e.g. reduce the number of seeds produced) but this implies a reduction in the pasture's potential persistence. Another example might be to select

for lines of the species that are less competitive with other plants, however this would reduce the capacity of the plant to thrive in mixed sward pastures. As such the focus of breeding efforts should not be on factors which limit the potential costs, but instead on those factors which limit the realisation of those costs. That is, those factors that facilitate the transfer of seeds such that new populations begin in external areas.

Therefore research priorities should include research to verify that seeds of the new varieties of birdsfoot trefoil:

- are not well adapted to transfer;
- require the presence of specific rhizobia to develop into productive adult plants; and
- are produced at some times of the year are not viable and have a high proportion of hard seed such that scarification is required for germination (see Chapter 6).

These factors would limit the potential for birdsfoot trefoil populations to develop in external areas. To enhance confidence of this, researchers could assess the potential for the specific rhizobia, or even rhizobia of the same class, to be found in external areas. With respect to plant introductions in general, researchers should focus on ensuring that the seeds are not suited to transfer by common vectors.

Of course, all of these suggestions are prefaced by the need to consider the benefits of undertaking to improve plant options compared to the benefits that alternative technologies, for example, engineering options might offer. Further, funding of additional research to limit salinity needs to be considered with respect to the potential welfare gains from directing that funding to alternative uses such as education, health and infrastructure.

10.5 Limitations of the Research

This study has been limited by data. However, this is symptomatic of the problem decision makers will always face when considering the introduction of new plants into new areas: there is no history on which to base parameter estimates. In particular,

there are no data on the dispersal of birdsfoot trefoil and so a range of results and conditions are presented. As such, this study provides information for government to make an introduction decision based on their knowledge of dispersal and levels of confidence at which they are prepared to proceed.

Another restriction is the limited spatial approach of the analysis. The study does recognise the spatial dimensions of the problem but with the exception of the division of the Catchment into two areas, agricultural and natural areas, the spread of the plant is assumed to occur evenly throughout the areas. For the purpose of this analysis, this is sufficient to estimate the costs. However, for the purpose of developing more specific policy or understanding the dispersal of the invasion, the difference between a dense invasion at the periphery of a subcatchment, versus small invasions throughout surrounding areas, will be important.

Agricultural production has been assumed to be homogeneous throughout the KPS. The farm used in the Subcatchment is representative of the major enterprises in the region, but in practice there will be other enterprises. As such, the estimated benefits may vary from farm to farm and thus the likely adoption will vary from landholder to landholder. Similarly, the productive capacity of the agricultural areas and natural areas of the BRC has been assumed homogeneous throughout each area. The assumption of homogeneity in the KPS and the BRC has allowed estimation of benefits and costs respectively but further development of policy could be aided by relaxing this assumption.

Another limitation of the analysis is that management or control of potential invasions of the new plant in areas of the Catchment is not considered. This could form part of government policy accompanying an introduction and could alter the decision to introduce a plant. For example, if invasions in the Catchment were easily detected and cost-efficient eradication methods were available, then the government may be prepared to allow introduction of the plant to the subcatchment even if there was a high chance that the plant would spread to areas outside the subcatchment. Any approach to management at this level will however be largely a political decision, so any analysis would therefore be speculative with regard to actions that might be taken.

In this analysis, the invasion of birdsfoot trefoil into other agricultural areas and natural areas *within* the subcatchment is assumed not to occur because in these areas it is plausible to assume that the plant would be managed as part of the introduction. This has provided a simpler construct in which to consider the external cost issues of this problem. Along these lines, other potential costs such as nitrogen pollution of the soil in the Subcatchment and benefits such as increased honey production are also not included. Further, avoided loss of biodiversity from reduced salinity in the Subcatchment has also not been estimated. The importance of other such costs and benefits should be considered with respect to plant introduction decisions more broadly.

Finally, the dynamic programming model used to estimate the benefits of introducing the plant to the Subcatchment does not incorporate stochastic elements. Analysis of the range of potential benefits was instead estimated because of the numerical complexity of such a task, the data limitations and the time required to run the optimising model.

These limitations provide some opportunities for future research.

10.6 The Future

10.6.1 Issues for Future Research

Given the limitations discussed, there are opportunities to refine this application of the decision criterion.

Adopting a spatial approach to the plant invasion, in particular, would be a useful addition to this research. A spatial approach would require a significantly greater data set on a subject already lacking data and will likely pose challenges for the stochastic simulation of the plant invasion. However, spatial information on costs may help better define policies that would accompany a plant introduction and in particular facilitate the development of management approaches to minimise population transfer into new areas. Spatial analysis would be enhanced by data on probabilities of seed

dispersal, including different kinds of dispersal, which is likely to be difficult to obtain.

Similarly, the analysis presented in this study would be enhanced through the development of a functional link between the size of the planted population of pasture and the invasion in the external areas. This would allow development of the approach presented in this study, but again would need to be informed by data on probabilities of seed dispersal for the range of vectors.

Finally, given that this study provides an alternative to the existing border assessment process used by Australia, there is the opportunity to investigate the augmentation of the existing WRA. This might include investigation of how the new varieties of birdsfoot trefoil might be assessed under the WRA. Discussion of the use of the approach developed in this study in the context of the existing WRA is provided in Appendix I.

The model developed in this study has limitations, but these do not detract from the study's main purpose. Extensions to the model as discussed will enhance the basis on which decisions are made and inform the further development of related policy.

10.6.2 Other applications

There are several new plants being considered for introduction to farming systems to reduce salinity in addition to the new varieties of birdsfoot trefoil. Since the beginning of this study the CRC for Plant-Based Management of Salinity, and more recently the CRC for Future Farm Industries, has developed a Weed Risk Management Protocol for Environmental Weed Assessment (Stone *et al.*, 2008). The new varieties of birdsfoot trefoil and any other species proposed for development and introduction by the CRC for Future Farm Industries will be subject to the Protocol and removed from the CRC's program of development/introduction if found to have a high risk score. This process continues to take a precautionary approach to plant introductions by removing plants that may be invasive irrespective of the potential for the plant to provide benefits.

The criterion and framework developed and demonstrated in this study would be a beneficial addition to the decision-making process of the CRC for Future Farm Industries and other bodies involved in the introduction and development of new plants to combat environmental problems. In particular, this might also be applied to the introduction of genetically modified plant species. As the potential for genetically modified plants to deliver on human health and environmental outcomes (e.g. salt tolerance and reduced water use) is realised, the conflict between the potential for a plant to be invasive and to deliver benefits which increase society's welfare will be exacerbated.

The framework presented in this study provides the approach to help resolve such conflicts. As demonstrated, it can accommodate highly uncertain and dynamic problems which can be expected in relation to genetically modified plants. Further, this approach provides the basis, using similar bioeconomic methods, for applications to other *ex ante* decisions involving animals, genetically modified or not, including marine life and insects.

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Appendices

Appendix A Top 20 Weeds of National Significance (WONS)

No	Common name	Scientific name	Source of Introduction [a]	Invasion	
				hectares & location where available [b]	OR, % of Australia [c]
1	Parkinsonia	*Parkinsonia aculeata*	S[1]		12.4
2	mesquite	*Prosopis spp.*	O, S, F[2]	Core of 0.8 million	5.3
3	blackberry	*Rubus fruticosus* agg.	Fr[3]	8 million nationally	9.0
4	lantana	*Lantana camara*	O[4]	4 million nationally	5.1
5	rubber vine	*Cryptostegia grandiflora*	O[5]	0.7 million in Qld	7.7
6	bitou bush / boneseed	*Chrysanthemoides monilifera*	E[6]	Potential 6.5 million in Victoria alone	3.0
7	prickly acacia	*Acacia nilotica* spp. *Indica*	F, S[7]	6.6 million in Qld	2.3
8	hymenachne	*Hymenachne amplexicaulis*	P[8]		1.0
9	salvinia	*Salvinia molesta*	O[9]		5.0
10	mimosa	*Mimosa pigra*	O[10]	0.08 million in Northern Territory	1.0
11	cabomba	*Cabomba caroliniana*	O[11]		0.5
12	Chilean needle grass	*Nassella neesiana*	U[12]		0.2
13	athel pine	*Tamarix aphylla*	S[13]		1.0
14	willows	*Salix* spp.	E[14]		0.8
15	serrated tussock	*Nassella trichotoma*	U[15]		2.2
16	parthenium	*Parthenium hysterophorus*	U[16]	8.2 million hectares in central Queensland [d]	5.6
17	pond apple	*Annona glabra*	Fr[17]		0.4
18	gorse	*Ulex europaeus*	O[18]	0.03 million in Tasmania	3.0
19	bridal creeper	*Asparagus asparagoides*	O[19]		5.0
20	alligator weed	*Alternanthera philoxeroides*	U[20]		0.4

Key: F=fodder; S= Shade; O= ornamental/garden, U= unintentional / unknown; Fr = fruit; E= environmental management; P= pasture

Source: Adapted from Thorp and Lynch (2000), [a] various (see over page), [b] Martin (2003), [c] Sinden *et al.* (2004), [d] Agriculture & Resource Management Council of Australia and New Zealand (2001)

Appendix A *continued*

Item no.	Source
1	www.deh.gov.au/biodiversity/invasive/weeds/p-aculeata
2	www.deh.gov.au/biodiversity/invasive/weeds/proposis
3	assumed
4	www.deh.gov.au/biodiversity/invasive/weeds/l-camara
5	www.nrme.qld.gov.au/factsheets/pdf/pest/PP11.pdf
6	www.amonline.net.au/factsheets/bitou_bush
7	www.mountmorgan.com/acacia
8	www.deh.gov.au/biodiversity/invasive/weeds/h-amplexicaulis
9	www.ecoaction.net.au/ccserac/docs/weeds/salvinia
10	www.abc.net.au/catalyst/stories/s721816.htm
11	www.deh.gov.au/biodiversity/invasive/weeds/c-caroliniana
12	www.nationalparks.nsw.gov.au/npws.nsf/content/Invasion+of+native+plant+communities+by+exotic+perennial+grasses
13	ww.northwestweeds.nsw.gov.au/athel_pine
14	www.weeds.gov.au/publications/guidelines/wons/salix
15	www.deh.gov.au/biodiversity/invasive/weeds/n-trichotoma
16	www.agric.nsw.gov.au/reader/weed-pubs/p7615
17	www.weeds.crc.org.au/main/wom_pond_apple
18	www.deh.gov.au/biodiversity/invasive/weeds/u-europaeus
19	www.weeds.crc.org.au/main/wom_bridal_creeper
20	www.deh.gov.au/biodiversity/invasive/weeds/a-philoxeroides

Appendix B Plant Questionnaire

For the purpose of selecting a plant for consideration in the framework a questionnaire was developed, to identify with plant breeders:

1. plants they wish to introduce to Australian agricultural systems;
2. plant demography parameters that are significant;
3. plant demography parameters for which existing data parameters are available; and
4. where available, alternative sources for parameter estimates.

The questionnaire was developed with the input from members of the CRC for Australian Weed Management and the CRC for Plant-Based Management of Dryland Salinity (CRC PBMDS). The final version of the questionnaire was reviewed by Associate Professor Mike Ewing, Program Leader, New & Improved Plant Species, CRC PBMDS.

The questionnaire was distributed to sixteen plant breeders. From the responses received, *Lotus corniculatus* was selected for assessment. Selection was based on availability of data and location of potential suitable case study sites.

The questionnaire is shown following.

A red tab on the right hand corner of a question cell indicates that further explanation can be found in the text box associated with that cell.

	New Perennial Pasture Plant	Response for plant sought for introduction	Information source (person or citation if available)	Additional information or comment based on your professional experience
i	Botanical name			
ii	Common name			
iii	Variety / Breeding Line			
iv	Origin of proposed accession (country)			
v	Centre of Origin			
vi	Countries of Significant Current Usage			
vii	Prominent closely related species already in Australia			
A	**Environmental Constraints**			
1a	Minimum 'ph' tolerance			
2a	A.A.R. limits (mm p.a.)			
2b	When is the plant's growing season (winter or summer active?)			
3	Land Class Preference *(please specify - see attached excel note)*			
4	Is the plant suited to arable or non-arable land?			
5	Soil Fertility Requirements *(low, medium or high)*			
6	Toxicity Tolerances or Intolerances - *please specify (Al, Mn, Salts, Other)*			
7	Is the plant more or less likely to populate disturbed or undisturbed environments?			
B	**Rhizobial requirements**			
1	Is the plant likely to be effectively nodulated by a commercial inoculant *(specify Group)*			
2	Is the plant likely to be nodulated by the naturalised rhizobia in Australian soils? *(if yes, pls answer Q3, otherwise go to Q4)*			
3	If nodulated by the naturalised rhizobia are the symbioses likely to be effective?			
4a	If exotic rhizobia needs to be imported or accessed from rhizobial germplasm collections, are they likely to form nodules with existing Australian crop and pasture species? *(if yes, pls answer Q4b)*			
4b	Is there any evidence to suggest this nodulation might be ineffective?			
C	**Life cycle**			
1	Is the dominant mechanism of perennation, **a)** seedling recruitment, **b)** clonal regeneration or **c)** other *(specify)*?			
2	Time *(weeks)* from seedling stage to reproductive stage (*ie, first ripe seed or plantlet formation(clonal regeneration)*)?			
3	No. of seeds (grams per plant) or plantlets (no per plant) produced by juvenille plants in their first growth cyle?			
4	No. of seeds (grams per plant) or plantlets (no. per plant) produced by mature plants in 2nd & subsequent growth cycles?			
5	Weight of seeds (grams)?			
	The remainder of the questions in this section relate to seed dominant perennators. Please skip to ***Section D)*** *Production, if your species is clonal regeneration dominant*			
6	At the beginning of **winter** following seed prod'n, what proportion of seeds will be…			
a	hard?	%		
b	soft?	%		
c	inert/unproductive?	%		

Cheryl Kalisch: insert data or your estimate in relation to the plant proposed for introduction.

Cheryl Kalisch: insert your data source, if known, or 'estimate' if it is an estimate based on your professional experience

Cheryl Kalisch: specify 3-4 species of importance (agricultural plant, garden plant or weed (agricultural or environmental)

Cheryl Kalisch: ph value below which plant vigour is reduced

Cheryl Kalisch: A.A.R. = Annual Average Rainfall

Cheryl Kalisch: coastal plain, coastal hinterland, coastal hilllands, tablelands -*steep hill country*, tablelands-*valley floor*, slopes, semi-arid plains. Please specify as many as applicable, or ALL if no particular land class

Cheryl Kalisch: consider with regard to phosphate fertility as this is the major limiter on the Australian landscape

Cheryl Kalisch: Please nominate the class of the inoculant required

Cheryl Kalisch: How many weeks from germination to first reproduction?

7	What is the dispersal distance (plant *in situ*) for seeds in the absence of spread vectors? (metres, radius)			
8	Describe the seed in terms of its dispersal mechanism *(eg. Awned, sticky, pod shatter)*			
9	Does the seed have characteristics *(size, colour)* that resemble prominent seed crops such that it could be a contaminant?			
10	What are the likely spread vectors for seed *(by wind, water, animals (excreta or in fur/wool etc), seed contaminant, forage)*?			
11	Germination rate *(%)* of new seeds *(First year following seed dispersal)*			
12	Germination rate *(%)* of 2-5 year old seeds in the seed bank			
13	Germination rate *(%)* of 5+ year old seeds in the seed bank			
14	What is the maximum age of seeds in the seed bank at which some will still be viable?			
15a	Seedling mortality *(%)*			
15b	What are the key factors to influence seedling mortality? *(moisture, temp,compet etc)*			
D	**Production**			
1	Size of mature plant *(height & spread of individual in meters)*?			
2	What is the livestock grazing suitability *(cattle type, sheep type)*?			
3	What is the feed value of the species *(with respect to nutritional value, digestibility etc)* - higher, same or lower as compared to lucerne?			
4	Antinutritional factors *(HCN, high levels of condensed tannins etc)*?			
5	Herbage production at maturity? *(kg/hectare/p.a.)*			
6a	In what season is the majority of herbage produced?			
6b	How long is the growing season (months)?			
7a	What is the ideal application of the plant in a farming system? *(e.g. permanent pasture, croppping system (undersow, alleys))*			
7b	Are there specific management requirements for this species and application?			
8	Is the plant a self pollinator or cross pollinator?			
9	Flowers - *describe in terms of colour, visibility, shape.*			
10	Does the plant produce burrs or thorns?			
11	Is the plant prone to disease or pest predators in Australia? *Please specify*.			
12	What is the total number of years the plant lives? *(i.e. longevity)*			
13	What is the perisistence (years) of a stand/population of the species?			
14	Are any special conditions required for germination? *(eg. fire, cultivation)*			
E	**Recharge characteristics**			
1a	described in Section A, what proportion *(%)* of recharge is likely to be prevented by the species....			
1b	If this estimate is dependent on the species being a particular density per ha or other factor, please specify that condition...			
2	What is the likely root depth (meters)?			
3	Are there other characteristics that I should be interested in when addressing this plant's potential to reduce recharge?			

Cheryl Kalisch:
e.g. the seeds are produced in pods with 10-15 seeds on average and the pod twists as it dries to release the seeds in an explosive action.

Cheryl Kalisch:
What proportion of seeds will germinate in an average year? If this is unlikely to vary by seed age, please write N/A for Questions 11 and 12.

Cheryl Kalisch:
What proportion of germinated seedlings will survive to become a young plant?

Page 2

Cheryl Kalisch:
eg. whole paddock, alleys, undersown with a cereal etc)

Cheryl Kalisch:
from germination to death

Cheryl Kalisch:
e.g. intense heat is required for some Australian natives to

Cheryl Kalisch:
Some researchers have published estimates regarding the ability of plant's to reduce recharge as compared to annual alternatives. Examples include Bennet, Ayres et al (2002) who report reductions of recharge to between 0 – 3 per cent under introduced species, down from 6 – 11 per cent under annual crops and pastures in trials in the 400 – 750 mm zone of NSW.

Cheryl Kalisch:
Provide range if

F	**Control Options**			
	If the species required removal from the following areas, are there feasible control options? *(describe for each)…*			
1a	- in permanent pastures?			
1b	- in cropping			
1c	- other areas *(including natural areas)*			
2a	How effective are these controls likely to be *(highly, mildly, uncertain)*?			
2b	How costly are these controls likely to be *(high, medium, low cost)*?			
3	Describe any adverse impacts/difficulties/dangers associated with the control methods.			
G	**Breeding Objectives**			
1a	Do you intend to alter the characteristics of the plant through breeding/selection/transgenic methods? *(Yes/No, specify which)*			
1b	In ranked order of importance, please indicate your top 3 explicit breeding objectives.			
2	Are there likely to be other characteristics of the plant which are modified as a result? *(specify)*			
H	***Other***			
1	***Other Characteristics:*** Are there other characteristics that I should be interested in when addressing this plant's potential to *be weedy*?			
2	***Import Assessment:*** Has this plant been subject to **W**eed **R**isk **A**ssessment *(process used by Biosecurity Australia to assess proposed plant introductions)*? On what basis was it rejected?			
3	***Other Contacts:*** Can you recommend others with whom that I should discuss this plant? *(please provide name, organisation and email address)*			
	Many thanks for your assistance.			
	Please return to C. Kalisch Gordon at cheryl.kalisch@une.edu.au or by post, c/o School of Economics, University of New England, Armidale, NSW, 2351			

Appendix C Pasture production – *birdsfoot trefoil*

		BFT in NP					BFT in Tall Fescue				
days		GR[1]	ME[2]	TME[3]	TME/ season		GR[1]	ME[2]	TME[3]	TME/ season	
31	*Jan*	20	8.5	5,270			20	8.7	5,394		
28	*Feb*	20	8.5	4,760	17,083	Su	20	8.7	4,872	24,216	Su
31	*Mar*	30	8.5	7,905			30	9.4	8,742		
30	*Apr*	30	8.5	7,650			30	9.4	8,460		
31	*May*	25	8.5	6,588	22,143	Au	25	9.4	7,285	24,487	Au
30	*Jun*	10	8.2	2,460			20	9	5,400		
31	*Jul*	5	6.5	1,008			10	8.2	2,542		
31	*Aug*	5	6.5	1,008	4,475	Wi	10	8.2	2,542	10,484	Wi
30	*Sep*	15	8.2	3,690			20	9	5,400		
31	*Oct*	40	9.1	11,284			50	9.6	14,880		
30	*Nov*	40	9.1	10,920	25,894	Sp	50	9.6	14,400	34,680	Sp
31	*Dec*	25	9.1	7,053			50	9	13,950		
				Annual Total	69,594				Annual Total	93,867	

1 Growth rate (kg DM/ha/day) of total pasture. Source, John Ayres, September 2006.

2 Metabolisable energy (MJ ME/kg). Source, John Ayres, September 2006.

3 Total Metabolisable Energy Available per month per hectare. TME = GR x ME x days.

Appendix D Gross margins – *birdsfoot trefoil*

Direct drill binary seed mix into herbicide treated 'run-down' sward - North East Slopes, Northern NSW

Notes:
- run down pasture is sprayed with herbicide three times to fallow
- pasture is direct drilled into sprayed pasture
- pasture mix includes BFT and fescue
- stand expected to last 5 years before re-establishment is required.
- expect stocking rate to lift from 4.5 DSE per ha to 9 DSE per ha.

		Machinery			Inputs			Total Cost
Operation	**Month**	hrs/ha	cost $/hr	**Total $/ha**	rate/ha	Cost $/unit	**Total $/ha**	**$/ha**
Broadleaf and grass weed control	Oct							
e.g. Glyphosate 450		0.05	41.54	**2.08**	1.20L	$5.00/L	**$6.00**	**$8.08**
Broadleaf and grass weed control	Dec							
e.g. Glyphosate 450		0.05	41.54	**2.08**	1.20L	$5.00/L	**$6.00**	**$8.08**
Broadleaf and grass weed control	Feb							
e.g. Glyphosate 450		0.05	41.54	**2.08**	1.20L	$5.00/L	**$6.00**	**$8.08**
Sowing	May	0.20	68.2	**13.64**				**$13.64**
-Birdsfoot trefoil					2kg	$15.00/kg	**$30.00**	**$30.00**
- Innoculant					100g/100kg seed	$8/100g	**$0.24**	**$0.24**
- Fescue					6kg	$7.50/kg	**$45.00**	**$45.00**
- PSN Starter Fertiliser, e.g. Granulock 15					100kg	0.60/kg	**$60.00**	**$60.00**
								$173.11

Developed with the assistance of Bob McGufficke, District Agronomist, NSW DPI, Inverell. July 2006.

Aerial Broadcast of Birdsfoot Trefoil into Native Pasture - North East Slopes, Northern NSW

Notes:
- aerial application into native pastures
- stand expected to last 5 years before re-establishment is required.
- expect stocking rate to lift from 3 DSE per ha to 7.5 DSE per ha.

		Machinery			Inputs			Total Cost
Operation	**Month**	hrs/ha	cost $/hr	**Total $/ha**	rate/ha	Cost $/unit	**Total $/ha**	**$/ha**
Aerial sowing	May	-	-	**77.00**				**$77.00**
-Birdsfoot trefoil					3kg	$15/kg	**$45.00**	**$45.00**
- Innoculant					100g/100kg seed	$8/100g	**$0.24**	**$0.24**
-Phosphorous Fertiliser e.g. Single Super					100kg	0.60/kg	**$60.00**	**$30.00**
								$152.24

Developed with the assistance of Bob McGufficke, District Agronomist, NSW DPI, Inverell. July 2006

Appendix E Land capability classification legend

Land Capability Map Legend – Source: Extracted from Rural Land Capability Mapping, KA Emery SCS and provided by DNR Information Sciences Branch of the Gunnedah Resource Centre (2006).

Land Classification and Soil Conservation Practices			**Interpretations and implications**
	I	No special soil conservation works or practices.	Land suitable for a wide variety of uses. Where soils are fertile, this is land with the highest potential for agriculture, and may be cultivated for vegetation and fruit production, cereal and other grain crops, energy crops, fodder and forage, and sugar cane in specific areas. Includes "prime agricultural land".
Suitable for regular cultivation	II	Soil Conservation practices such as strip cropping, conservation tillage and adequate crop rotation	Usually gently sloping land suitable for a wide variety of agricultural uses. Has a high potential for production of crops on fertile soils similar to Class I, but increasing limitations to production due to site conditions. Includes "prime agricultural land".
	III	Structural soil conservation works such as graded banks, waterways and diversion banks, together with soil conservation practices such as conservation tillage and adequate crop rotation.	Sloping land suitable for cropping on a rotational basis. Generally used for the production of the same type of crops as listed for Class I, although productivity will vary depending upon soil fertility. Individual yields may be the same as for Classes I and II, but increasing restrictions due to the erosion hazard will reduce the total yield over time. Soil erosion problems are often severe. Generally fair to good agricultural land.
Suitable for grazing and occasional cultivation	IV	Soil conservation practices such as pasture improvement, stock control, application of fertiliser and minimal cultivation for the establishment or re-establishment of permanent pasture.	Land not suitable for cultivation on a regular basis owing to limitations of slope gradient, soil erosion, shallowness or rockiness, climate or a combination of these factors. Comprises the better classes of grazing land of the State and can be cultivated for an occasional crop, particularly a fodder crop, or for pasture renewal. Not suited to the range of agricultural uses listed for Classes I to III. If used for "hobby farms", adequate provision should be made for water supply, effluent disposal and selection of safe building sites and access roads.
	V	Structural soil conservation works such as absorption banks, diversion banks and contour ripping, together with the practices as in Class IV.	Land not suitable for cultivation on a regular basis owing to considerable limitations of slope gradient, soil erosion, shallowness or rockiness, climate or a combination of these factors. Soil erosion problems are often severe. Production is generally lower than for grazing lands in Class IV. Can be cultivated for

			an occasional crop, particularly a fodder crop or for pasture renewal. Not suited to the range of agricultural uses listed for Classes I to III. If used for "hobby farms" adequate provision should be made for water supply, effluent disposal, and selection of safe building sites and access roads.	
No cultivation	VI	Soil conservation practices including limitation of stock, broadcasting of seed and fertiliser, prevention of fire and destruction of vermin. May include some isolated structural works.	Productivity will vary due to the soil depth and the soil fertility. Comprises the less productive grazing lands. If used for "hobby farms", adequate provision should be made for water supply, effluent disposal, and selection of safe building sites and access roads.	
OTHER	VII	Land best protected by green timber	Generally comprises areas of steep slopes, shallow soils and/or rock outcrop. Adequate ground protection must be maintained by limiting grazing and minimising damage by fire. Destruction of trees is not generally recommended, but partial clearing for grazing purposes under strict management controls can be practised on small areas of low erosion hazard. Where clearing of these lands has occurred in the past, unstable soil and terrain sites should be returned to timber cover.	
	VIII	Cliffs, lakes or swamps and other lands unsuitable for agricultural and pastoral production	Land unusable for agricultural or pastoral uses. Recommended uses are those compatible with the preservation of the natural vegetation, namely: water supply Catchments, wildlife refuges, national and state parks, and scenic areas.	
	U	Urban areas	Class Subscripts	Special Uses
	M	Mining and quarrying areas	c	Terrain developed for a specific crop (capability class range IV to VII) as a result of the combination of particular soil, terrain, climatic and economic conditions. The class includes such crops as grapes, bananas, avocados and pineapples.
			d	Terrain developed for intensive agricultural production and associated with flood irrigation. The class includes land developed for cotton and rice production.

Appendix F LP Matrix

	Forage Oats	Lucerne	BFT Tall Fescue	BFT native pasture	Native Pasture	Wheat Rotation	Fallow	High Density Trees	Sheep	Cattle	Recharge		
Gross margin*	-170	-35	-35	-30	0	176	-11	-33	48	461	0		
Feed Spring	-14601	-11751	-13872	-10358	-7676	0	0	0	2332	15458	0	<=	0
Feed Summer	0	-9999	-9686	-6833	-11880	0	0	0	1930	15725	0	<=	0
Feed Autumn	-8952	-9087	-9795	-8857	-5910	0	0	0	1505	15004	0	<=	0
Feed Winter	-14463	-5370	-4194	-1790	-1914	0	0	0	1441	13674	0	<=	0
Lucerne constraint	0	1	0	0	0	0	0	0	0	0	0	<=	50
BFT constraint	0	0	1	1	0	0	0	0	0	0	0	<=	100
Tree constraint	0	0	0	0	0	0	0	1	0	0	0	<=	100
Cropping land	1	0	0	0	0	1	1	0	0	0	0	<=	30
Land	1	1	1	1	1	1	1	1	0	0	0	=	100
Recharge transfer	55	11	20	27	40	60	110	0	0	0	-1	=	0
Recharge constraint	0	0	0	0	0	0	0	0	0	0	1	=	2800**

* adjusted by the DP model based on yield for the given watertable depth (Equation 5.11).

** controlled by the DP model to represent each desired state transition.

Appendix G Pasture production estimates for the Kings Plain Subcatchment

		Forage Oats					Lucerne				
days		GR[1]	ME[2]	TME[3]	TME/ season		GR[1]	ME[2]	TME[3]	TME/ season	
31	Jan	0	0.00	0			24	10.48	7,797		
28	Feb	0	0.00	0	0	Su	24	10.84	7,281	24,997	Su
31	Mar	0	0.00	0			25	10.88	8,430		
30	Apr	15	14.01	6,306			24	11.23	8,086		
31	May	37	14.01	16,073	22,379	Au	17	11.77	6,202	22,718	Au
30	Jun	30	14.01	12,612			14	10.10	4,244		
31	Jul	26	14.14	11,400			13	10.52	4,241		
31	Aug	28	13.99	12,146	36,158	Wi	15	10.63	4,941	13,426	Wi
30	Sep	47	13.84	19,518			26	10.78	8,405		
31	Oct	35	13.00	14,105			40	9.67	11,987		
30	Nov	8	12.00	2,880	36,503	Sp	35	8.56	8,985	29,377	Sp
31	Dec	0	0.00	0			27	11.85	9,918		
				Annual Total	95,040				Annual Total	90,518	

1 Growth rate (kg DM/ha/day) of pasture. Source, McDonald (1999)
2 Metabolisable energy (MJ ME/kg). Source, Nutritive values database, NSW DPI, August 2006.
3 Total Metabolisable Energy Available per month per hectare. TME = GR x ME x days.

		Native Pasture				
days		GR[1]	ME[2]	TME[3]	TME/season	
31	Jan	40	8.25	10,230		
28	Feb	40	8.25	9,240	29,700	S
31	Mar	22	7.3	4,979		
30	Apr	22	7.3	4,818		
31	May	22	7.3	4,979	14,775	A
30	Jun	8	6.5	1,560		
31	Jul	8	6.5	1,612		
31	Aug	8	6.5	1,612	4,784	W
30	Sep	8	6.5	1,560		
31	Oct	34	8.5	8,959		
30	Nov	34	8.5	8,670	19,189	S
31	Dec	40	8.25	10,230		
				Annual total	68,448	

1 Growth rate (kg DM/ha/day) of total pasture. Source, John Ayres, September 2006.
2 Metabolisable energy (MJ ME/kg). Source, John Ayres, September 2006.
3 Total Metabolisable Energy Available per month

Appendix H Decision analysis criteria application

Payoff tables and introduction decision based on Maximin and Maximax Criteria for low, base and high expectations of benefits – *where the % of seeds dispersed is the variable state of nature and the range 0.5% - 10% is assumed exhaustive*

Benefits ($'million PV)	Decision	Payoff Table ($'million PV) states of nature (% of seeds dispersed) 0.5	1	2	3	4	5	6	7	8	9	10	Decision Criteria: Maximin Selects the maximum minimum	Maximax Selects the maximum maximum
Low ($0.25)	*Introduce*	-$1.09	-$3.83	-$10.45	-$16.35	-$21.89	-$27.14	-$33.80	-$37.85	-$45.45	-$50.04	-$55.03	-$55.03	-$1.09
	Don't	-$0.25	-$0.25	-$0.25	-$0.25	-$0.25	-$0.25	-$0.25	-$0.25	-$0.25	-$0.25	-$0.25	-$0.25	-0.25
													Don't	*Don't*
Base ($8.80)	*Introduce*	$7.46	$4.72	-$1.90	-$7.80	-$13.34	-$18.59	-$25.25	-$29.30	-$36.90	-$41.49	-$46.48	-$46.48	$7.46
	Don't	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80	-$8.80
													Don't	*Introduce*
High ($33.00)	*Introduce*	$31.66	$28.92	$22.30	$16.40	$10.86	$5.61	-$1.05	-$5.10	-$12.70	-$17.29	-$22.28	-$22.28	$31.66
	Don't	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00	-$33.00
													Introduce	*Introduce*

Payoff Table and decisions based on Maximin and Maximax Criteria where 0.5, 2 and 10 percent of seeds are dispersed– *where the performance of plant in subcatchment is the variable state of nature and the three levels of performance are assumed exhaustive*

		Payoff tables ($'million PV)			**Decision Criteria**	
		states of nature			Maximin	Maximax
% of Seeds Dispersed	*Decision*	Low benefits	Base benefits	High benefits	Selects maximum minimum	Selects maximum maximum
0.5%	*Introduce*	-$1.09	$7.47	$31.66	-$1.09	$31.66
	Don't	-$0.25	-$8.80	-$33.00	-$33.00	-$0.25
					Introduce	*Introduce*
2.0%	*Introduce*	-$10.45	-$1.90	$22.30	-$10.45	$22.30
	Don't	-$0.25	-$8.80	-$33.00	-$33.00	-$0.25
					Introduce	*Introduce*
10%	*Introduce*	-$55.03	-$46.48	-$22.28	-$55.03	-$22.28
	Don't	-$0.25	-$8.80	-$33.00	-$33.00	-$0.25
					Don't	*Don't*

Appendix I The Weed Risk Assessment (WRA) process & the approach developed in this study

New plant species must be assessed for their potential to become weeds before they are permitted for import to Australia. Biosecurity Australia (BA) is the agency responsible for the review and development of policies which allow for the safe importation of animals, their genetic material and other products. BA uses a Weed Risk Assessment (WRA) process for all new plant species proposed for introduction into Australia as seeds, tissue culture or any other material for propagation. WRAs are usually undertaken at the species level but sub-specific taxa or hybrids may also be assessed. The weed risk assessment (WRA) process is a three tier, science-based, quarantine risk analysis tool for determining the weed potential of proposed new plant imports. By using this WRA, Australia complies with its rights and obligations as a signatory to the World Trade Organisation, under the Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement), and the International Plant Protection Convention.

The WRA involves the proponent or importer, BA and the Australian Quarantine and Inspection Service (AQIS). The first tier requires determination of whether the plant is already present or permitted to enter Australia. This tier is completed in conjunction with AQIS. If the plant is not present in Australia, the species is then subject to the second tier, weed risk assessment, which is conducted by BA. The second tier includes the prediction of the possibility of a plant becoming a weed on the basis of a series of scored questions.

The WRA scored questions are as shown in the question sheet shown on the following page. Review of the questions reveals that there is in an emphasis on identifying the nature and potential extent of costs. This includes investigation of financial, environmental and to some degree, social, costs. Uncertainty is considered by way of incorporating knowledge of the plant's history of investigation of the species' history of repeated introductions outside of its natural range.

Biosecurity Australia's Weed Risk Assessment Questionnaire

Answer yes (y) or no (n), or don't know (leave blank or ?), unless otherwise indicated

	Botanical name: Common name: Family name		Outcome: Score: Your name:	
			History/Biogeography	
A	**1**	*Domestication/*	1.01 Is the species highly domesticated. If answer is 'no' got to question 2.01	
C		*cultivation*	1.02 Has the species become naturalised where grown	
C			1.03 Does the species have weedy races	
	2	*Climate and*	2.01 Species suited to Australian climates (0-low; 1-intermediate; 2-high)	2
		Distribution	2.02 Quality of climate match data (0-low; 1-intermediate; 2-high)	2
C			2.03 Broad climate suitability (environmental versatility)	
C			2.04 Native or naturalised in regions with extended dry periods	
			2.05 Does the species have a history of repeated introductions outside its natural range	
C	**3**	*Weed*	3.01 Naturalised beyond native range	
E		*elsewhere*	3.02 Garden/amenity/disturbance weed	
A			3.03 Weed of agriculture/horticulture/forestry	
E			3.04 Environmental weed	
			3.05 Congeneric weed	
			Biology/Ecology	
A	**4**	*Undesirable*	4.01 Produces spines, thorns or burrs	
C		*traits*	4.02 Allelopathic	
C			4.03 Parasitic	
A			4.04 Unpalatable to grazing animals	
C			4.05 Toxic to animals	
C			4.06 Host for recognised pests and pathogens	
C			4.07 Causes allergies or is otherwise toxic to humans	
E			4.08 Creates a fire hazard in natural ecosystems	
E			4.09 Is a shade tolerant plant at some stage of its life cycle	
E			4.10 Grows on infertile soils	
E			4.11 Climbing or smothering growth habit	
E			4.12 Forms dense thickets	
E	**5**	*Plant type*	5.01 Aquatic	
C			5.02 Grass	
E			5.03 Nitrogen fixing woody plant	
C			5.04 Geophyte	
C	**6**	*Reproduction*	6.01 Evidence of substantial reproductive failure in native habitat	
C			6.02 Produces viable seed	
C			6.03 Hybridises naturally	
C			6.04 Self-fertilisation	
C			6.05 Requires specialist pollinators	
C			6.06 Reproduction by vegetative propagation	
C			6.07 Minimum generative time (years)	1
A	**7**	*Dispersal*	7.01 Propagules likely to be dispersed unintentionally	
C		*mechanisms*	7.02 Propagules dispersed intentionally by people	
A			7.03 Propagules likely to disperse as a produce contaminant	
C			7.04 Propagules adapted to wind dispersal	
E			7.05 Propagules buoyant	
E			7.06 Propagules bird dispersed	
C			7.07 Propagules dispersed by other animals (externally)	
C			7.08 Propagules dispersed by other animals (internally)	
C	**8**	*Persistence*	8.01 Prolific seed production	
A		*attributes*	8.02 Evidence that a persistent propagule bank is formed (>1 yr)	
A			8.03 Well controlled by herbicides	
C			8.04 Tolerates or benefits from mutilation, cultivation or fire	
E			8.05 Effective natural enemies present in Australia	

A= agricultural, E = environmental, C= combined

Source: http://www.daff.gov.au/ba/reviews/weeds/system/weed_risk_assessment. Accessed 25 Nov, 2007.

The WRA system was reviewed after 7 years of use (NWRAS Review Group, 2005). The review found that the system was generally robust and effective. Keller *et al*. (2007) estimated that the net benefits to Australia of the WRA system would be $1.8bn over 50 years. Their assessment demonstrates the benefits of the use of the WRA for the ornamental plant industry in Australia using simulation of the aggregated market costs and benefits associated with ornamental plant imports. In a review of the WRA in the context of geographical variability, Gordon *et al*. (2008) found that the WRA system rejects an average of 90 percent of known invasive species and accepts an average of 70 percent known non-invasive species. On this basis, it is possible to extrapolate that the benefits that would be provided by 30 percent of non-invasive species (which will not be accepted by the WRA) and the benefits that accompany the 90 percent of invasive plants which will be rejected, will not be captured by Australia.

The WRA is a process recognised as robust and that allows for the avoidance of considerable agricultural and environmental costs. The studies by Keller *et al*. and Gordon *et al*. both demonstrate the aggregate value derived from the use of the WRA to exclude plants with considerable potential costs. The WRA will however exclude some plants with the potential for significant benefits, including external benefits. In terms of the analysis in this study, the present value of lost benefits from excluding a plant such as birdsfoot trefoil are estimated under the base case to be $8.8 million for the Kings Plain Subcatchment alone. The WRA process does not allow consideration of these benefits and neither does it assist in managing the situation where some agents may believe they should have access to a plant and others that wish to avoid exposure to the plant.

The addition of a small number of questions to the existing WRA would facilitate a parallel decision making process which considers the potential welfare gains from specific plant introductions. The approach developed in this study provides the basis for this because it includes benefits and costs, as well as externalities. The questions would relate to the 'Desirable Traits' of the plant proposed for introduction and include consideration of the value of the traits and the extent to which these values

could be captured. Questions relating to the value of costs would also be necessary to allow balanced consideration of changes in welfare. This approach might augment the existing WRA process or be used to supplement it in the context of post border plant management decisions (given considerations of Australia's WTO obligations).

The augmentation of the existing WRA with these minimal additional inputs and on the basis of the approach developed in this study would go some way to providing a more balanced approach to plant introduction decisions. Importantly it would provide the basis for accommodating the positive and negative externalities that can be ignored by processes that take a precautionary approach and focus entirely on the costs of a potential plant introduction. A process which includes balanced consideration of costs and benefits provides the platform for resource allocation decisions to be made by the market and in the interests of efficiency and social welfare (with accompanying policy as required). Undertaking a WRA on a plant such as birdsfoot trefoil, with and without questions and associated analysis relating to its Desirable Traits, would be a useful next step in the research.