An Economic Perspective on Land Degradation Issues

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AN ECONOMIC PERSPECTIVE ON LAND DEGRADATION ISSUES

Table of Contents

List of Figures Page

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Acronyms and Abbreviations Used in Report

ABARE	Australian Bureau of Agricultural and Resource Economics
ASS	Acid sulfate soils
CRC	Cooperative Research Centre
DP	Dynamic programming
ESD	Ecologically sustainable development
GDP	Gross Domestic Product
LP	Linear programming
LWRRDC	Land and Water Resources Research and Development Corporation
R&D	Research and development
RDC	Research and Development Corporation
SLA	Statistical Local Area
TFP	Total factor productivity
WTA	Willingness to accept
WTP	Willingness to pay

Executive Summary

The objective of this report has been to review the perspectives from economics on the debate about sustainability in general and land degradation in particular. Land degradation is a concern because of the potential costs it imposes on present and future generations in terms of lost production, poor water quality, reduced biodiversity and other poor environmental outcomes. It is one component of a broader debate about the sustainability of natural and environmental resource use by the present generation.

The focus of much economics research related to sustainability in agriculture has concerned questions of efficiency in resource use. The two main areas of contribution are first, in providing a framework within which the benefits and costs of alternative land management strategies can be evaluated and second, in providing an understanding of the incentives for efficient resource use that confront individual farmers and the community, particularly in the presence of externalities. Sections of the Report review techniques for assessing efficiency in resource use; studies of the cost of land degradation; and reasons for the slow adoption of landcare technology.

In common with similar reviews, the conclusion here is that with its focus on valuing land degradation in terms of foregone production, most empirical work from the past provides little insight into how land degradation is best managed from the viewpoint of either farmers or the community. The main deficiency of this approach of valuing the potential benefit from ameliorating degradation as the value of foregone production is that this estimate of benefit is not related to any feasible management strategy to achieve it. Hence this approach provides little guidance as to where resources are best used to combat degradation. Other difficulties with the approach are discussed in the Report.

An important source of market failure or inefficiency is the set of externalities associated with land degradation that arise through attenuated property rights. An important contribution of this Report has been to make clear the analogies between spatial and temporal externalities and, following Quiggin, to identify how the significance of externalities might be measured as the difference between wealth under a regime of open access to a resource as against a regime where the resource is treated as common property. The non-point nature of the externalities associated with many forms of land degradation means that precise farm level control mechanisms, whether they be of a market or regulatory nature, are not available under present technologies. Even if property rights could be made more secure, it is most likely that efficient resource use from society's viewpoint would result in some level of degradation, partly because the benefits and costs of changes in land use express themselves over very long investment periods.

The methodological approach of economists that is the basis of benefit/cost analysis is to identify the resource use strategy that maximises a measure of income or wealth. Resources are used to the point where marginal benefits equal marginal user cost where marginal user cost can be defined to include opportunity cost in terms of either using resources now rather than later (the temporal dimension) or in terms of off-site costs in a spatial dimension. When costs and benefits flow over several years, discounting techniques are used to reflect the opportunity costs (or time preference) of not being able to use future income now. Techniques to identify efficient

resource use are reviewed in the Report. The advent of powerful computers has enabled the solution of dynamic resource use problems over long time horizons but there is still great uncertainty about the biophysical relationships especially those associated with the hydrology of catchments.

However there is strong opposition to the use of benefit/cost analysis as the final, or even as an important, arbitrator in decisions about the use of natural resources. The most significant objections seem to be the focus of benefit/cost analysis on efficiency to the neglect of equity and related to this is its basis in a utilitarian philosophy which accords few rights to other species or to future generations.

The concern for sustainability seems as much a question of equity as efficiency and from social welfare theory (Bator 1968) we know that efficiency is a necessary but not sufficient condition to optimise social welfare. There is no unique way of using resources efficiently because there is an efficient solution to each of the many (infinite) ways in which resources could be distributed between generations. There are situations in which social welfare can be improved by moving to a less efficient solution but one which results in a more equitable distribution of resources. Hence efficient resource use by the present generation may not be sufficient to protect the rights of future generations, uncertain though they may be.

Much attention is presently being paid to the removal of externalities associated with land use and the choice of a discount rate as means of protecting the interests of future generations. It would seem that neither approach is sufficient. The removal of externalities has both efficiency implications, in that the wealth of an economy increases when resources are used in a way that accounts for all costs, and equity implications, in that property rights are less attenuated. However the concern is largely with present generations and it is likely that the interests of these generations are best served by a significant, though perhaps lower, level of resource degradation. Similarly any positive discount rate at some point discounts future benefits to zero. While much empirical work remains to be done, experience to date suggests that it is likely that even at low discount rates, the benefits from strategies (based on current technologies) to ameliorate land degradation, such as salinity, will be modest relative to costs. Both these issues are discussed in more detail later in this Report.

Concern about equity is often expressed as ‘fairness’ which is achieved if future generations have no cause to envy the resource endowments of the present generation. It seems highly unlikely that the present generation would be envious of any previous generations. If future generations are successively more wealthy, as has generally been the case to date, then the question arises as to whether they are likely to envy the present generation. To make this discussion less abstract, a pertinent question to consider with respect to Australian agriculture, is whether the present (or a future) generation placed in the position of the previous generation but with knowledge of the consequences for salinity of agricultural technologies then in use and also with knowledge of the benefits that have flowed from these technologies, would have preferred a different pattern of development (in the spirit of Rawls’ ‘veil of ignorance’).

As noted later in the Report, the rights of future generations are problematical as are the rights of other species. The individuals of these future generations are only ‘possible’ persons and which of them become actual persons depends on decisions made by preceding generations - about resource use among other things. On this basis some argue that the present generation is only weakly obligated to consider future generations. However others argue that there is a chain of obligation to future generations because of our obligation to our children, actual people, and their obligation to their children with this obligation continuing from generation to generation. Some writers have pointed out that this concern for future generations contrasts starkly with treatment of the poor of the present generation.

Despite this uncertainty about the obligations of the present to future generations, there appears to be a growing acceptance in the community, at least in rich countries, that government takes action to preserve the access of future generations to natural and environmental resources. If present generations are concerned about diminished access of future generations to natural resources in general or to specific natural resources then, in addition to using resources efficiently now, some form of saving must be entered into that preserves resources. The level and form of saving is largely dependant on beliefs about the substitutability between natural and manmade resources, and the rates of technical change, economic growth and world population, issues discussed later in the Report. This concern about sustainability raises a number of questions which are discussed but not resolved later in the Report:

- Are resources being used at an unsustainable rate?;
- How should we treat future generations?;
- What type of resources should be protected?;
- How should resources be protected?;
- What conservation measures are presently being taken?.

These issues are often discussed in the context of weak versus strong notions of sustainability. At the weak end of the spectrum, writers such as Solow (1993) suggest while each generation should save/invest at least enough to compensate for the natural and environmental resources it has used up, the form of this savings is not necessarily in maintaining stocks of particular resources but in maintaining ‘a generalised capacity to produce economic well-being (p.168)’, so that every generation is at least as well off as its predecessors. While no attempt is made in this Report to review particular policy instruments, it can be said that in general, policy prescriptions at this end of the spectrum largely relate to more efficient resource use through better information about the long term consequences of alternative technologies and the amelioration of externalities. In general, policy instruments of this nature impose few restrictions on the use of resources by the present generation as a whole even though there may be some distributional impacts as property rights associated with externalities are changed. Those who argue for weak sustainability are optimistic about the degree of substitution possible between natural and man-made capital and the continuing possibilities for technical change to be saving of scarce resources.

Stronger versions of sustainability require the conservation of specific resources such as stocks of fossil fuels and old growth forest and the maintenance of air and water quality and biodiversity. In an agricultural context it may extend to maintaining dimensions of soil health such as soil depth, carbon levels, pH, and efficient use of rainfall by maintaining perennials (in a salinity context).

At the weak end of strong sustainability is the concept of a safe minimum standard – the stock of a resource that is just high enough to prevent irreversible losses. The safe minimum standard is imposed as a constraint in the traditional economic model seeking to maximising wealth by using resources efficiently. The change in wealth (shadow value) from relaxing this constraint provides an estimate of its cost to the present generation.

Setting a safe minimum standard is an explicit act of saving for future generations but the risk is that future generations may not value the resource that is conserved as much as poor sections of the present generation. Hence many advocates of safe minimum standards apply a caveat that the costs of the standard are not ‘too high’. While ecologists and economists might provide some insight into the consequences of imposing a safe minimum standard, other unknown considerations will influence how future generations value these resources. Hence there is great uncertainty about which resources should be conserved. Those favouring conservation appeal to the ‘precautionary’ principle.

While there is a vocal section of the present generation who appear committed to strong measures to conserve resources for the future, governments have generally been slow to follow. For both individuals and governments, sustainability has the characteristics of a public good. Just, Hueth, and Schmitz (1982) pointed out that while many in the present generation may have concerns for future generations, the alleviation of these concerns by the sacrifices of a few are non-rival and non-exclusive and can be enjoyed by many who make no provision. Hence the real willingness of the present generation to save for future generations is difficult to ascertain.

One indicator of this willingness to save for the future is provided by the actions of governments. It seems to me that few policies have been adopted which require significant savings by present generations. Some writers have argued that practical concern for the future could be demonstrated by a redistribution of resources from rich economies well endowed with built capital to capital poor economies where the opportunity costs of conserving resources by the present generation is likely to be high (Toman, Pezzey and Krautkraemer (1995)).

In Australia resource conservation issues that may impose significant costs on the present generation include greenhouse gas abatement, land use change to control the rate of salinisation of land and water resources, and the protection of native vegetation including old growth forests. Some are disappointed at the lack of action by governments to protect these resources. However while we have some idea of the costs to the present generation of conserving these resources, there is a great deal of uncertainty as to whether some future generation would be envious of us were we not to conserve these resources in the same way as we are not envious in general, of the resource use decisions and endowments of our predecessors.

In an agricultural context, the concern of government until recently has been to increase productivity by more efficient resource use by farmers through the provision of research and extension services. In the Report it is noted that research and extension solutions are inadequate in dealing with resource use issues related to externalities and sustainability because generally they expect farmers to act against their own interests. While the amelioration of externalities may lead to more efficient resource use and address some inequities, it is not a sufficient condition for sustainability. The dimension of sustainability concerning the endowment of resources across generations requires explicit saving and the community has to judge whether it is equitable to impose the cost on a section of the community, such as farmers who have remnant native vegetation for example, or whether the cost should be borne more widely by the community.

1. Introduction

Agriculture uses natural resources such as land and water and other inputs such as machinery and labour to produce livestock, grains and fibres which after processing become final consumer goods in the form of food and clothing. One way to enhance the welfare of society is to develop agricultural technologies which use resources more efficiently. Productivity in Australian broadacre agriculture has grown at an average rate of 2.5 percent between 1953 and 1994 (Mullen and Cox, 1996) and much of this gain in efficiency can be attributed to public investment in R&D (Mullen and Cox, 1995).

While this rate of productivity growth within Australian agriculture has been impressive relative both to other sectors of the economy and to the agricultural sectors of other economies, there is concern as to whether the way in which agriculture and other sectors of the economy use resources is sustainable. At this time in history some of our greatest concerns seem to be with the use of the environment for waste disposal. Land degradation in various forms and the loss of biodiversity are also prominent issues. In earlier times, questions of sustainability were raised in the guise of concerns about whether future generations would be able to feed themselves or whether important non-renewable resources such as oil, would be used up. Adding to the concern is an appreciation that in many instances, through externalities, the interests of society, both present and future generations, may not coincide with the interests of individuals.

Because natural resources have the characteristics of public goods to some degree, there is a danger that they will be exploited by present generations. Hence there is strong support for the view that governments should intervene more directly and implement policies to achieve what is referred to as sustainable development.

It is tempting to think that there is general agreement about what sustainability is and what the objectives of sustainable development should be. The most widely used definition of sustainable development is that put forward in the Brundtland Report (‘Our Common Future’, World Commission on Environment and Development, 1987) - ‘….development that meets the needs of the present without compromising the ability of future generations to meet their own needs’. In an Australian context, the National Strategy for Ecologically Sustainable Development defines ecologically sustainable development (ESD) as:

> ‘…using, conserving and enhancing the community’s resources so that ecological processes, on which life depends, are maintained, and the total quality of life, now and in the future, can be increased’ (Commonwealth of Australia, 1992, p.6).

The three core objectives to Australia's strategy are:

- 'to enhance individual and community well-being and welfare by following a path of economic development that safeguards the welfare of future generations;

- to provide for equity within and between generations; and

- to protect biological diversity and maintain essential ecological processes and life support systems' (Commonwealth of Australia 1992, p. 8).

The way in which these concepts apply to Australian agriculture is described in the reports of the Ecologically Sustainable Development Working Group, 1991 and the Commonwealth of Australia, 1991.

However whilst the concept of ESD has been widely embraced there is little agreement as to how sustainability is measured and what changes in resource use are implied by it. It is even unclear what policy prescriptions are required. Randall (1994) noted that:

> 'Diagnoses range from simple market failure to modern lifestyles incompatible with the carrying capacity of the planet. Policy prescriptions run the gamut from correction of market failures to elimination of discounting, intergenerational reassignment of entitlements, optimal re-investment rules for natural resource rents, and safe minimum standard of conservation; and that is just from relatively mainstream resource economists. Some of our ecological economist colleagues would extend the range of prescriptions to include "robust strategies" emphasising resiliency, and radical restructuring of the modern consumer economy and society'.

In trying to 'make sense of sustainability', he identified the following five broad classes of sustainability objectives which differ in what is assumed about the degree of substitution between natural and reproducible capital:

- maintaining welfare or aggregate output;
- maintaining the total stock of capital;
- maintaining natural resources;
- maintaining biotic resources;
- preserving particular natural resources.

The last category comprises unique resources that most people wish to preserve whether or not they are essential for human survival. For the first two objectives to achieve sustainability there must be a high degree of substitutability between natural and reproducible assets and/or technical progress induced by the increasing scarcity of natural resources that is saving of these resources. It would seem that achieving these objectives would require little government intervention. This version of sustainability is referred to in the literature as a weak sustainability.

The third and fourth objectives reflect a belief that the restrictions on substitutability between natural and reproducible assets are severe and that technical progress will not be sufficiently saving of natural assets to ensure sustainability. The policy prescriptions associated with this strong version of sustainability generally involve restrictions on the use of specific natural resources to preserve stocks.

A major source of confusion and discord is that there is little agreement on which classes of natural resources require protecting to prevent exploitation by present generations. In well functioning markets as the prices of scarce resources rise there would also arise incentives to substitute other resources, reproducible resources in particular, for resources rising in price and to develop technologies that were saving of these scarce resources.

In more direct language raising the concerns of economists about the vagueness of the concept of ESD, Beckerman (1996) argued that:

'… the concept of 'sustainable development' is not only logically incoherent and hence incapable of providing any clear guidance to policy or measurement but if interpreted literally as is possible given its vagueness, could even prejudice the standards of living of future generations or add to inter-generational inequalities. Hence, it is concluded that, far from being an improvement on the time-honoured economist's objective of maximising social welfare over whatever time period is believed to be relevant, the objective of sustainable development is a step backwards in the formulation of guidelines for allocating resources in the socially optimal manner (p.153).'

1.1 'Economists' v 'Ecologists'

We can construct caricatures, referred to as 'economists' and 'ecologists', of the set of views held by those who subscribe to a 'weak' version of sustainability and of those subscribing to a 'strong' version of sustainability. 'Economists' are generally optimistic that efficient resource use by the present generation, encompassing the remedying, where possible, of externalities that impose significant costs, is sufficient to protect the interests of future generations. They are likely to see the obligations of the present to future generations in the form of 'passing on' a standard of living at least as high as it received and of a 'generalised capacity to produce economic well-being' (Sollow, 1993, p. 168) rather than the preservation of particular resource stocks. 'Economists' see scarcity as an economic concept in which the supply of a resource is related to its demand. As Randall (1987) pointed out a scarce resource has a positive price. As a resource becomes more scarce, its price rises calling for greater efforts in exploration, recycling and technical change to use it more sparingly or to allow greater rates of extraction. The concept of scarcity can be extended to include the ability of the environment to handle wastes. 'Economists' are more likely to be optimistic about continuing economic growth and technical progress and to allow some degree of substitution between natural and manmade resources. They are more likely to trust the market in allocating resources between generations, once property rights are more fully specified and they would be more comfortable with a positive discount rate than 'ecologists'.

On the other hand, 'ecologists' hold a strong view of sustainability and argue that the present generation has to directly protect a range of natural resources for future generations. 'Ecologists' are more likely to see scarcity in terms of known reserves and are much less comfortable with the efficacy of price signalling changes in technology and substitution between resources especially over the very long planning horizons that they deal with. 'Ecologists' are more likely to favour direct government restrictions on the use of resources on the basis of what they refer to as the precautionary principle. They are most likely to argue that substitution possibilities between natural and man made inputs are extremely small which implies a much diminished role for technical change and reduced economic growth associated with increasing environmental damage and finite resource stocks. ‘Ecologists’ argue for discount rates close to zero. Among the ‘ecologists’ there is a group of economists, ecological economists, who are concerned that traditional neoclassical theory is inadequate in dealing with this issue of the intergenerational equity of resource use or sustainability.

While the bases for these divergences in views between ‘economists’ and ‘ecologists’ are discussed in following sections of this report, it should be remembered that while both groups can provide valuable insights neither would appear to have any special case for arguing that they are the final arbiter in any of these areas. Questions about how we treat future generations and whether the rights of species other than man are recognised, are well beyond the disciplines of economics and ecology. Rather choices in these areas should reflect the preferences of society as informed by philosophical and ethical considerations and an awareness of the consequences of alternatives as identified by economics and ecology.

1.2 The Scope of this Report

The focus of this report is on reviewing economic perspectives concerning land degradation in Australian agriculture. There is a concern that the traditional (neoclassical) approaches of economists are short sighted with respect to the value of resources to future generations and the finiteness of resources including the capacity of the environment to absorb wastes. Hence in Section 2 there is a discussion of many of the key issues concerning the value of natural resources including the rights of future generations and other species, the controversy over discounting and a brief explanation as to why environmental and natural resources might be poorly priced. The issue of externalities is discussed in Section 3. In particular the distinction between off-site effects and externalities is explained and analogies between temporal and spatial externalities are identified. An approach suggested by Quiggin (1986 and 1991) to measuring the significance of externalities as a guide to public intervention is described. Then in Section 4 questions of the scarcity of natural resources, the sustainability of Australian agriculture and the adoption of landcare technologies are discussed. Approaches to valuing changes in the flows through time of the services provided by natural resources are reviewed in Section 5. The methodologies reviewed range from estimating the cost of degradation as the value of production foregone relative to a benchmark, to benefit/cost or management approaches that allow decision makers to respond tactically through time to changes in the stock of natural resources and to uncertainty with respect to weather and price. The Report concludes with a review of the nature, extent and economic analyses of management options for major types of land degradation in Australia including salinity, acid soils, acid sulfate soils, erosion and soil structural decline. Land degradation comes in many forms including loss of amenity value and of biodiveristy. The term is used very generally in most of this report. However the specific examples used and the types of land degradation reviewed in Section 6 relate more narrowly to dimensions of soil degradation.

Because there are often externalities for present and future generations, the use of natural resources is an important public policy issue. No attempt is made in this review to treat policy in a systematic way although the characteristics of particular land degradation issues that make them of public policy concern are identified. An excellent treatment of the general policy analysis framework can be found in Godden (1997). Currently there is much interest in suasive and market based policy instruments. A discussion of suasive instruments can be found in the Productivity Commission (1997) report on ‘Ecologically Sustainable Land Management’ and in a Commonwealth of Australia (1991) report on landcare. Market based approaches are reviewed in an Industry Commission (1997) research paper. A more cautionary discussion of the place of markets can be found in Bromley (1997) and Quiggin (1986).

1.3 Some Definitions

The environment can be thought of as an asset that yields a flow of services to society through time. These services take a variety of forms but Freeman (1993) identified the following four categories of service flow:

- material inputs such as fuels, minerals, soil nutrients and water etc. most commonly referred to as natural resources;
- life support services in terms of air and water quality;
- amenity services (both use and nonuse) related to recreation and leisure activities;
- waste dispersal services for the by-products of economic activity.

The latter three categories might be defined as environmental inputs but the distinction between natural and environmental resources or inputs is somewhat artificial and of little practical importance in this context.

Clearly these environmental service flows are provided jointly and consequently when producers or consumers change the flow of one category of these environmental services there is likely to be an accompanying change in the flow of another type of environmental resource. Hence the use of land as an input in the production of food may cause a deterioration in some other resource flow such as water quality or visual amenity. The key question for society is how do we allocate the use of these assets through time in an equitable and efficient way.

Resources derive value partly from their *use* in production or consumption activities. We are familiar with the value of mineral resources in producing cars and timber in producing houses. Perhaps less familiar is the value of a waterway in disposing of waste, part of the production process, and its value for recreation and scenic uses. The value of a resource in these types of uses is referred to as its *use value*.

Natural and environmental resources may also have *non-use values* attached to them. This is the value members of society derive from knowing that resources, such as Kakadu, exist even though they may never get to use them.

There are variations to the simple use and non-use system. Non-use value is also referred to as *passive* or *existence value*. Sub classes have been defined. *Option value* is defined as the value associated with the potential future use of a resource. If use value is defined to encompass both present and future uses then option value falls in the use value category but if use value refers to current use only then it falls in the non-use category. *Bequest value* is the value members of society are prepared to pay to preserve a resource for future generations.

Natural resources can also be classified as to whether they are renewable, non-renewable or capital resources. The stock of *renewable resources,* such as a fishery or a forest, can increase through a natural regenerative process provided it does not fall below some critical level. The economic problem associated with renewable resources is to determine optimal rate of use through time taking account of the rate of regeneration.

The stock of some natural resources, such as minerals and fossil fuels, is destructible or *nonrenewable,* except at very slow natural rates. If the supply of these resources is limited so that consumption has to be rationed through time, they are classified as *depletable or exhaustible*. For some nonrenewable resources such as minerals and some dimensions of soil quality, while the existing stock cannot be increased, the service flows can be restored to some degree through recycling or fertilising. The economic problem here is the intertemporal allocation of this resource.

Graphical presentations of how renewable and nonrenewable resources are allocated between two time periods can be found in many resource economics texts (for example, Just, Hueth and Schmitz (1982) pps 315-326 and in McInerney (1976)). The key concept from this literature is that when a natural resource is used for production in the current period in such a way that less is available for production or consumption in future periods then the discounted value of this lost future production and consumption can be regarded as an opportunity or *user cost* of the use of the resource in the current period. This user cost must be offset against current benefits from the use of the resource. Unless resource users are conscious of this user cost, resources are likely to be used at exploitative rates. Clearly there is great uncertainty about user cost or the value of future production and consumption and there is great interest in economic and biological modelling to reduce this uncertainty.

The rest of the Section consists of a more detailed graphical presentation of optimal use of a non-renewable resource over two periods. It is taken almost directly from McInerney (1976). McInerney extended this simple model to other types of natural resources which, for example, were renewable or had recycling opportunities. In analysing resource use through time he took the perspective of society. Marginal user cost incorporated not only the costs to the resource user in terms of foregone future production but also the impact of this on society. It should be borne in mind that this concept of user cost is also relevant to resource use decisions within a firm although from this perspective marginal user cost does not include costs to society. As explained more fully below, optimal resource use decisions from the perspective of the firm and the community diverge when externalities are present.

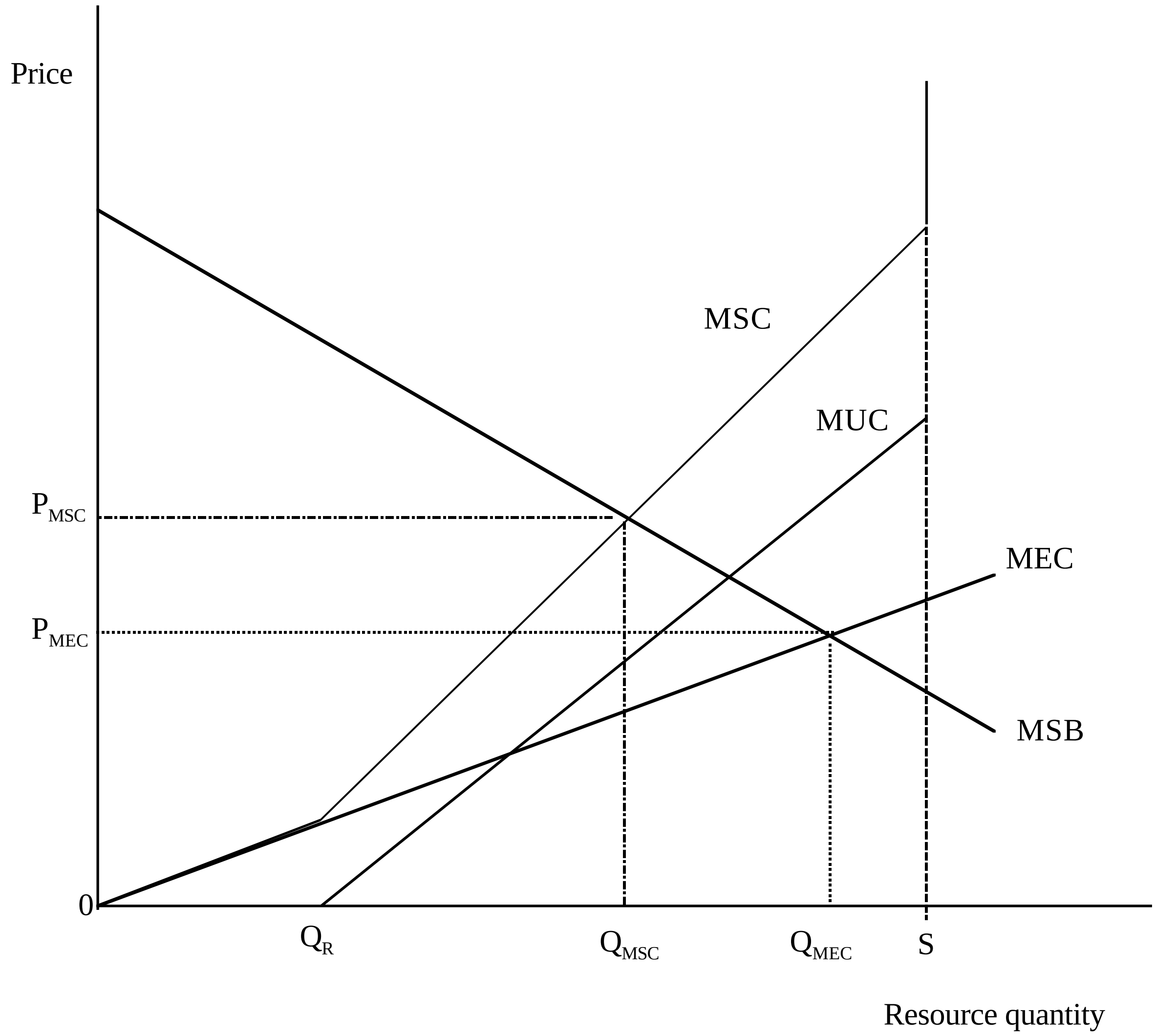
In Figure 1.1 the upward sloping marginal extraction cost curve (MEC) represents the value of other resources (labour and capital) needed to harvest the resource from its natural state. It is the ‘short sighted’ supply curve. Because in this case the natural resource stock is non-renewable or fixed in supply, any unit consumed now is unavailable to the future. Initially the resource stock exists in quantities that exceed future demand. Hence not every unit of current consumption incurs a user cost.

This is illustrated in Figure 1.1 where user costs are not incurred until current depletion of resources reaches OQ_R, the residual stock required to satisfy maximum future demand. Beyond this point, (Q_R), each unit of current consumption robs future generations of the benefits had those resources been retained for use in the future. Each successive unit of current consumption incurs a progressively higher loss of future benefits, yielding an upward sloping marginal user cost curve (MUC). The total real marginal cost to society of current resource use is therefore the sum of both the marginal extraction cost and the marginal user cost as shown in Figure 1.1, giving the discontinuous marginal social cost curve (MSC).

Society will gain the maximum net benefit from its resource stock when the marginal benefits and cost of consumption are equal. As shown in Fig. 1.1, equating MSB with MSC implies consuming (depleting) OQ_{MSC} of the resource stock in the current period, thereby leaving ($S\text{-}OQ_{MSC}$) for the future. The appropriate (real) resource price that would bring this about on the market is OP_{MSC}.

This is in contrast to the levels of resource extraction determined on purely static grounds. Under static conditions the marginal benefits need only be equated to the costs that have to be met in the current period ie, MEC. This then leads to extravagant current consumption of the resource (OQ_{MEC}), encouraged by its underpricing in real terms (OP_{MEC}), with marginal social benefits from consumption being less than the true marginal cost to society.

Figure 1.1: Resource utilisation and pricing



2. Some Issues in Valuing Changes in Resource Use

Economists commonly view decisions about the use of natural resources as requiring a weighing up of the benefits and costs of the alternative resource use patterns. There is a wide literature about the principles and practice of benefit cost analysis applied to natural resource issues (Randall 1987) and some of the issues that have to be confronted in valuing streams of costs and benefits are discussed below. However before doing so it should be noted that this use of a benefit-cost framework to assist in resource allocation decisions is based on a utilitarian philosophy that actions are good if they result in the satisfaction of preferences. Randall (1999b) pointed out that there is another branch of Western moral philosophy which argues that consequences and preference satisfaction ought to play a subordinate role to some ‘universal moral imperatives’ and hence some natural entities ought to be protected by constraint. In the face of this pluralism it would seem that while benefit-cost analysis may be acceptable as a decision rule in many situations, there may well arise decisions about the ultimate use of natural resources that the community expects to be made with nonutilitarian concerns in mind. The point remains that every time a resource use decision is made by individuals or society a benefit-cost judgement has been made either implicitly or explicitly.

In assessing the benefits and costs of an alternative pattern of resource use, economists attempt to elicit the changes in welfare of all those individuals affected by the change in resource use. As already noted, some of the environmental service flows may be poorly priced and means of eliciting their true value are the subject of an extensive literature. The concepts of opportunity cost and substitutability are central to how individuals perceive changes in their welfare. Perhaps these concepts can be illustrated by the case of water use in the Macquarie Valley. Society faces a choice as to whether to allocate all the river flow to agricultural and domestic water supply purposes or whether to set aside a proportion of this flow for environmental purposes such as the conservation of flora and fauna in the river marshes and associated recreational uses.

The opportunity cost of one pattern of resource use is the benefits to individuals were the resource to be used in the next most valuable way. The opportunity cost of using water solely for agricultural and domestic purposes is the loss of welfare occasioned by the decline in water flow and quality downstream, the loss of wildlife and the reduced recreational opportunities. Note that that the rate at which people choose between these alternatives is unlikely to be constant. As a higher proportion of the river flow is diverted to agricultural uses, the environmental benefits forgone increase so that people become less willing to trade off these environmental benefits for further increases in agricultural production. Some of these losses affect not only the present generation but also future generations and may involve the loss of animal or plant species and hence questions arise as to what are the rights of future generations and other species and how they are to be accounted for.

In the following section an economic perspective is presented on some of the difficult issues that arise in valuing changes in the flows of natural resources alluded to in the example above. An attempt is made to highlight where the views of ‘economists’ and ‘ecologists’ diverge.

2.1 Value Based on Willingness to Accept Compensation

The value of a resource or its opportunity cost should be measured as the willingness to accept compensation, WTA, to give up the use of a resource. The obverse of willingness to accept compensation is the willingness to pay of individuals to be able to use resources in a certain way. When a change in the pattern of resource use is being proposed, net benefits are assessed as the sum of the benefits to those who wish to acquire use of the resource, expressed in WTP terms less the costs to those who give up their rights to the resource, expressed in WTA compensation terms. In empirical work it has been found that WTA values are difficult to elicit and instead of asking respondents their WTA compensation for a loss in resource use, they are asked their WTP for a similar gain in resource use. This approach is based on the assumption that WTP and WTA should be equal apart from a small income effect.

However, as summarised in Knetsch (1994, 1995), experimental work to test this equality between WTA and WTP has consistently found that the WTA compensation of respondents for the loss of a resource exceeds their WTP for the gain of a resource by a large order of magnitude. The implication of this research is that empirical studies typically undervalue the loss of environmental resources because they are based on the WTP concept (Randall 1999b). As yet techniques to more carefully elicit WTA values have not been developed. Perhaps the pragmatic approach until such techniques have been developed is to examine the sensitivity of benefit/cost estimates to scaling the loss of environmental resources by a factor of two or three.

Money is the convenient measuring stick used to account for changes in costs and benefits but at a more fundamental level it is a proxy for the rate at which society would trade water used in agriculture for its alternative environmental uses, for example.

2.2 The Role of Price in Signalling Scarcity

Economists see scarcity as an economic concept where the supply of a resource is related to its demand, rather than as the physical size of a reserve of resources. Hence price is an important indicator of scarcity[1]. As a resource becomes more scarce, its price rises calling for greater efforts in exploration, recycling and for technical change to use it more sparingly or to allow greater rates of extraction. Price rises induce technical change that is saving of the scarce resource.

Higher prices also provide an incentive to substitute away from the use of the scarce resource in production and consumption activities. If the price of a natural resource increases reflecting greater scarcity, then the cost to society of this increased scarcity is much greater if producers have to continue using the same relative amount of the scarce resource - if inputs have to be used in fixed proportions in production in other words. The cost to society is less if producers can switch to other technologies that use relatively less of the scarce and more expensive resource[2].

[1] Smith (1980, p.277-78) warned against judging resource scarcity on the basis of a single index such as relative price.

[2] This discussion has been in terms of changing relative input prices but a change in relative demand for outputs is also likely to cause a change in relative input prices.

The extent to which it is possible to substitute away from natural resources that become scarce towards less scarce natural resources or man-made resources in production and consumption activities is a source of disagreement between ‘economists’ and ‘ecologists’. Perhaps ‘ecologists’ are saying that because manufactured inputs are derived from natural resources, input substitution is limited because it ultimately involves substitution between natural resources some of which are non-renewable.

This debate about input substitution might seem a bit arcane but it underlies the distinction between ‘weak’ and strong’ versions of sustainability and debate about appropriate policies for sustainable development. This debate is reviewed in Randall (1994). In brief, if there is a high degree of substitution between natural resources then the role of government can be restricted to ensuring markets are operating in an efficient manner. If however substitution possibilities are limited then the case for regulating the use of natural resources to prevent irreversibilities through ‘safe minimum standards’, for example, is stronger.

Similarly ‘economists’ and ‘ecologists’ disagree about the efficacy of prices in signally scarcity and inducing technical changes that is saving of scarce resources. Some of the limited evidence for these positions is reviewed in Section 4.

2.3 Unpriced Environmental Flows

A major impediment to prices signalling scarcity is that some resource flows are not priced in a way that reflects their value to society. Some of the benefits and costs of changes in resource use are relatively easy to value. They are reflected in changes in production which are generally priced in well functioning markets such as the market for wool or cattle where property rights are clearly specified. However changes in natural resource use may also cause changes in environmental flows that are not traded in markets. There is a concern that the value society places on many environmental resources is often not reflected in market transactions and hence there are no clear signals as to how neighbours and society value the changes in environmental flows associated with the use of natural resources in agricultural production. For example, the price of land may not clearly reflect the value of component attributes such as pH or fertility. In addition, the contribution of a farm to degradation problems elsewhere in a catchment through dryland salinity or acid sulphate soils, for example, is not normally reflected in the price of the land. Broader environmental flows such as biodiversity and aesthetic values are usually not priced.

The reason why environmental resources are likely to be poorly priced is that it is not possible to establish the property rights of those who use these services to the same degree as for more tangible inputs such as a tonne of wheat or a unit of labour. Property rights are often difficult to establish because environmental services have the characteristics of public goods, at least to some degree. Perhaps the most important of these characteristics is that it is difficult to exclude those who are unwilling to pay from use of the environmental service. The other characteristic is that the use of some environmental services, particularly nonuse services such as the knowledge that flora and fauna are protected, is non-rival, meaning that the use of these services by some members of society does not reduce their availability to other members of society.

Externalities[3] reflect the problem of attenuated property rights. Externalities generally take the form of costs imposed on others in the form of reduced water or air quality for example but sometimes they may take the form of a benefit. Hodge (1995)defined an externality as arising ‘wherever some agent, A (which may be either an individual or a firm), takes an action which has an impact on some other agent, B, that B has not chosen to accept (p.29)’. While externalities may be rival in use, it is difficult for B to claim compensation (establish price excludability). Another characteristic of externalities is that they are jointly produced with marketed commodities. These characteristics mean that externalities are poorly priced, that is, the value placed on them by society is not fully revealed in market transactions. Hence the appropriate incentives are not in place to encourage Agent A to produce only that level of externality that Agent B or society is willing to accept.

Externalities are also referred to as the off-site costs associated with resource use. Whether particular patterns of resource use have off-site impacts or externalities has implications for the extent of government involvement. The grounds for government involvement are less strong if the environmental degradation associated with a particular agricultural technology, for example, is contained on the farm employing the technology and is reflected in the value of the particular piece of land[4]. Hence in assessing environmental degradation it is important to identify whether the degradation arises largely through externalities or is largely caused by on-site ‘private’ use of resources.

The extent to which the services and commodities have the characteristics that result in them being poorly priced is not complete and can change over time. Property rights allowing price excludability, that is ensuring that those who use environmental resources bear the full cost of their use including the externalities they create, are dependent on the institutional arrangements society has put in place. Godden (1997) gave examples of how technology can change the extent to which price excludability and rivalry can be enforced.

Throughout this report the emphasis is on the externalities agriculture imposes on the rest of the community. However, it should be remembered that other sectors of the economy impose externalities on agriculture, in the form of losses in air and water quality for example, and that some of the externalities associated with agriculture maybe benefits in the form of enhanced amenity values, the control of pests, or enhanced public health through cheap food.

Agricultural technologies use inputs such as seed and fertiliser to produce outputs such as wheat. Wheat and seed and fertiliser are easily measured and valued because the property rights associated with them are clearly defined and they can be traded in a market. Hence for such a simple system it is easy to value the impact of either a change in the technology of combining seed and fertiliser or a change in demand for wheat which may arise from a change in the technology of producing or processing some other farm product.

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[3] Godden (1997) has a good discussion of excludability, rivalry and externalities and their implications for property rights and the existence of markets where the value to society of environmental services can be discovered. He pointed out that the extent to which commodities are excludable and rival is at least partly a function of the institutions society puts in place to establish property rights and partly a function of technology.

[4] As discussed further below, Just, Hueth and Schmitz (1982) noted that concern by present for future generations is in some degree a public good and hence excessive use of resources by the present generation imposes an intertemporal externality on future generations.

Clearly the technology associated with wheat production is far more complex than the simple representation we have used so far. In addition to market inputs the production of wheat requires the use of resource or environmental inputs such as soil and water. These inputs are not homogeneous but have a bundle of quality attributes associated with them. Soil, for example, can be characterised by its acidity, depth, structure, nutrient status etc. The value of a block of land in part reflects the value of these attributes but the value of any one attribute can only be inferred indirectly. The buyer of land may not have complete knowledge of these attributes and hence the price of land may not fully reflect their value.

Moreover the property rights to a block of land are not complete in that the actions of neighbours can change some attributes without enforceable compensation. Dryland salinity, erosion and pest infestations are examples of this problem. As a joint product of wheat production, there may well be a number of externalities associated with a deterioration in soil or water quality, for example run-off into waterways of excess nutrients supplied as fertiliser imposes costs on users downstream which they cannot avoid or claim compensation for under current technologies and institutional arrangements. Were there mechanisms by which wheat producers were charged the value to society of their use of environmental inputs then they would use different technologies and perhaps less wheat would be grown.

There are two points to make about the issue of non-priced resource flows. First decisions about resource use or the ranking of projects reflect implicit valuations of these unpriced elements. Second, economists would like to see these changes treated explicitly and techniques such as contingent valuation have been developed to make this possible, although such techniques remain controversial and difficult to apply. These techniques are reviewed in detail in many resource economics texts.

2.4 The Rights of Future Generations

Some wonder why we should be concerned about imposing costs on ourselves to leave resources for future generations whom we do not know and whom are likely to be far wealthier than ourselves if the past is a reliable guide to the future. The issue of the rights of future generations, like the rights of other species, is not one that scientific disciplines like economics and ecology are specially fitted to resolve. Other disciplines such as ethics and philosophy would seem to have larger roles in informing society in respect to these choices. So far I have referred to the wealth or welfare of society as being assessed over future generations as well as the present one. But what are the rights of future generations? To quote from Grey (1996, p. 161):

> ‘The moral status of future persons is problematic. It is often claimed that we should take the interests of the indefinite unborn very seriously because they have a right to a decent life. It is also claimed (often by the same people) that we should allow unrestricted access to abortion, because the indefinite unborn have no rights’.

He pointed out that to date, moral thinking has largely been synchronic or static whereas the consideration of the rights of future generations is a diachronic or dynamic issue[5]. He attempted to resolve the contradiction between two main schools of thought - impersonal

[5] This sounds like a familiar story to resource economists.

theories that may impose unacceptable costs on the present generations and person-affecting theories which may impose too few obligations on the present generation. He argued that concern for future generations can be justified by applying ‘impersonal principles subject to retroactive person-affecting constraints (p. 161)’ but did not seek to explain how this concern might be expressed in practice. Howarth (1992, 1995) argued that the rationale for concern with future generations lies with recognising that the present generation has obligations to its children who in turn have obligations to their children creating a ‘chain of obligation’ into the future.

The next question that arises is, how do we judge whether generations are fair to each other. Woodward (2000) invokes Foley’s principle which suggests that a generation acts fairly if it uses resources in such a way that future generations are not envious.

Part of this debate also concerns the nature of the assets which we should leave our descendants. One view is that we should leave physical stocks of natural resources. This is the strong sustainability position. The risk is that some of these resources will be of no value to future generations - as has been the case with peat and may become the case for coal if the technology of solar power continues to advance.

The position of Robert Solow (1986, p.142), Nobel Laureate in economics has been that:

> ‘The current generation does not especially owe its successors a share in this or that resource. If it owes anything, it owes generalised productive capacity or, even more generally, access to certain standards of living or level of consumption. Whether productive capacity should be transmitted across generations in the form of mineral deposits or capital equipment or technical knowledge is more a matter of efficiency that equity’.

The debate about the rights of future generations is reflected in the debate about the discount rate that should be applied to the flow of services from natural resources far into the future. Those who assign significant rights to future generations argue for low discount rates. However underlining this issue of inter- and intra- generational equity, Tulloch (1964) queried whether in redistributing wealth to future generations via a reduced social discount rate we should not first consider whether a more equitable wealth distribution could not be achieved by looking after the marginalised of the present generation. Perhaps the best way to take care of future generations is to distribute resources to the poor in the present generation.

Before discussing discounting procedure, assigning rights to future generations raises doubts that a sole focus on efficient solutions to land degradation issues will lead to outcomes that accord with these rights. Any review of the neo-classical theory of welfare maximisation (Bator, 1968; Randall, 1987; Howarth and Norgaard, 1990) stresses that there is an infinity of efficient solutions corresponding to an infinity of ways in which resources are distributed between members of society where society can be extended to future generations. At least conceptually situations can be identified where welfare within or over generations may be improved by a less efficient but more equitable use of resources. Simply removing externalities in land use is not a sufficient condition to fully account for the rights of future generations (Woodward, 2000).

2.5 Discounting Future Benefits and Costs

As has already been noted, natural resources yield a flow of services through time. Hence a discounting procedure has to be used to allow this flow of services to be aggregated to a net present value allowing alternative patterns of resource use to be compared on a like basis. The choice of the discount rate is a highly contentious issue because as the discount rate increases, resource use systems in which benefits accrue in the distant future are penalised relative to those patterns of resource use which yield benefits more immediately.

The view common in 'green' literature is that the discount rate should be zero (Young 1992). On the other hand, Randall (1994, p.6) argued that 'It is perhaps the most enduring of myths that a society which discounts future production and costs, *ipso facto*, sacrifices future welfare, and therefore violates reasonable requirements for intergenerational equity'. Eminent economists line up on either side of this argument and hence the issue will not be resolved here.

The effect of the choice of discount rate on the time period after which resource flows have no value can be seen in Figure 2.1. A discount rate of seven percent means that the present value of a $1 is less than half a cent after 79 years. For a discount rate of three percent the time period is still only 180 years, a very short time horizon by ecological standards.

As Weitzman (1998) and others before him have pointed out, we are uncomfortable with the view that a catastrophe a century or two in the future does not matter much. Conversely not to discount means that projects expensive now but with a long but small stream of benefits are always preferred to projects where the stream of benefits if limited.

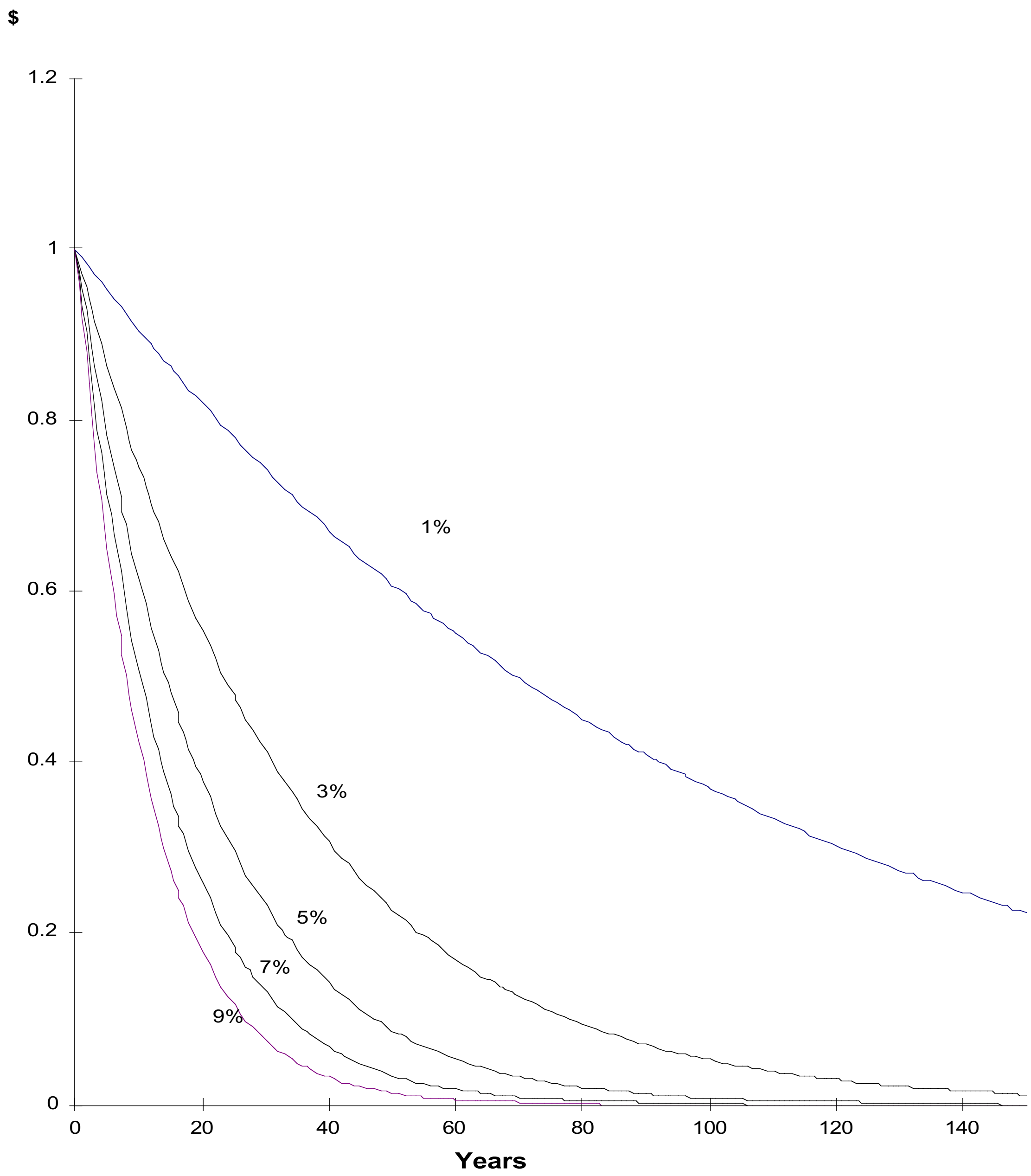
The discount rate reflects the tradeoff between the rate of time preference of consumers and the production possibilities they face. The rate of time preference is the rate at which consumers prefer current to future consumption. The rate of time preference is likely to be influenced by expectations of future levels of wealth (a positive relationship); by life cycle factors such as current dependant family size (positive) and concern for future generations (negative). The production possibilities faced by consumers determines the rate of return they can earn by diverting resources from current consumption into productive activities allowing consumption in the future.

There are several reasons why the social rate of discount may be lower that that used by individuals. To start with society is likely to be less risk averse, at least to short term sources of uncertainty associated with markets and weather, and does not pay tax.

More importantly while individuals and families have the incentive to optimally allocate their consumption through time and there are financial instruments to help them, this is not the case for consumption across generations. This issue of the social discount rate is discussed well in Just, Hueth and Schmitz (1982, 297-306). They argued that concern for the consumption opportunities of future generations has the characteristics of a public good requiring some form of government intervention. While many in the present generation may have concerns for future generations, the alleviation of these concerns by the sacrifices of a few are non-rival and non-exclusive and can be enjoyed by many who make no provision. Hence the discount rate preferred by the present generation may be too high because there is no means to ensure that it reflects the time preference over all generations. This problem of identifying a utility function reflecting time preference across generations is independent of issues related to the production possibilities frontier such as whether the markets for natural assets such as land accurately reflected quality attributes such as soil structure and acidity.

The discount rate is influenced by the rate of growth in wealth through time. As can be seen from Figure 2.2, adapted from Just, Hueth and Schmitz (1982), the shape of the production possibilities frontier influences the social discount rate and the shape of the frontier is influenced by whether future production possibilities are larger or smaller than present production possibilities. This relationship was pointed out by Ramsey and is discussed in Just, Hueth and Schmitz (1982).

Figure 2.1: The Impact of the discount Rate on the Future Value of a Dollar



The no-growth, declining and increasing production possibilities scenarios are represented by the P_1, P_3 and P_2 frontiers which are tangent to the social indifference curve across generations at points a, c and b giving social discount rates of r_1, r_3 and r_2 (the slopes of the tangents at these three points). If production possibilities were expected to expand then the social discount rate, r_2, is higher than for the no growth situation, favouring projects which deliver benefits earlier, but present consumption is lower than future consumption. The reverse holds for the scenario in which production possibilities decline in the future.

Just, Hueth and Schmitz (1982) went on to argue that if consumption by future generations was equally as important as consumption by the present generation, the social discount rate for declining, no-growth and increasing production possibilities would be negative, zero and positive respectively.

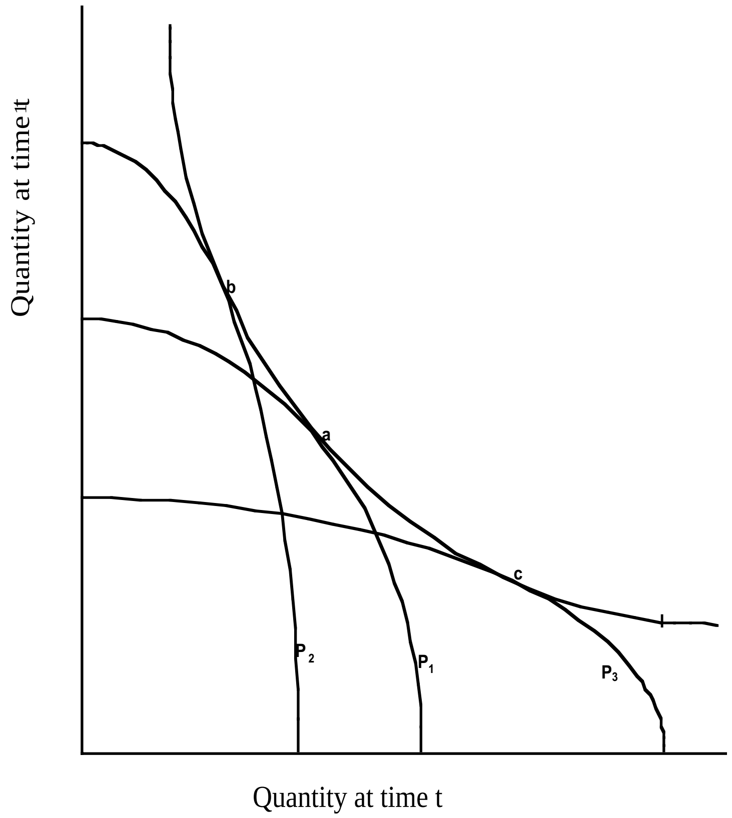
Another issue is whether the discount rate is invariant over time and between projects as normally assumed. Weitzman (2001) suggested that it maybe appropriate to lower discount rates as the planning horizon lengthens. Knetsch (1995) also reviewed this issue. He pointed to some anomalies which suggest that people do not apply the same discount to all types of investment decisions and concluded that ‘It may not be inconsistent for individuals to demand a high rate of return for their private investments while choosing to have public funds devoted to demonstrably low return reforestation efforts’ (p. 70). Knetsch pointed to the parallel that whether individuals accept a particular level of risk depends on the nature of the event as well as its expected value. It is not clear whether some of the anomalies in time preference referred to by Knetsch reflect different perceptions of risk nor have we discussed here the extent to which the discount rate should reflect risk.

It is beyond the scope of this paper to pursue in detail the policy issues associated with the choice of a social discount rate. Just, Hueth and Schmitz (1982, p. 303) pointed out that the issue is similar to the treatment of intragenerational equity reflected in the weights attached to the benefits and costs accruing to different income classes. They also pointed out that governments could use other instruments to reflect concerns about intergenerational equity other than the manipulation of the discount rate. They discussed problems concerning capital availability and rationing that may arise if there was a marked divergence between social and market rates of interest. Baumol (1968, p.789) referred to an ‘unavoidable indeterminancy’ in the choice of discount rate.

2.6 The treatment of risk and uncertainty

There is a large literature on the complex area of the treatment of risk and uncertainty and it is beyond the scope of this review to thoroughly cover this literature. Risk is reflected in benefit cost analyses in two ways. First, at least for private investment decisions, the discount rate incorporates a risk premium which reflects the relative riskiness of alternative investments.

Figure 2.2 The Social Discount Rate and Future Production Possibilities



There has been a debate as to whether public investments ought to similarly incorporate a risk premium (Just, Hueth and Schmitz (1982) p.310) but there now seems to be some agreement that the social discount rate be riskless especially for environmental investments (Arrow, 1996, p.31).

A conventional way to incorporate risk in decision analysis is by using expected utility or profit as the decision criteria. Firms and individuals are assumed to be able to assign probabilities which sum to one for uncertain events and discern which alternative investments yield the highest level of expected utility or profit. At a further level of sophistication, decision analysis allows tradeoffs between the expected profit and variance (of profit) of alternative investments (Anderson, Dillon and Hardaker, 1977) by either using a utility function reflecting risk aversion as the objective function or when that is unknown, applying tests of stochastic dominance (Meyer, 1977a&b, McCarl, 1990).

A range of techniques have been used to conduct investment analysis in situations of uncertainty and risk aversion. These are discussed in more detail below but they extend from linear programming to quadratic, dynamic and stochastic programming and often rely on extensive simulation based on sampling from hypothesised distributions of key stochastic variables such as rainfall and price. A review of these techniques, which are based on the expected utility hypothesis, can be found in Jones (1997).

There is mounting evidence that the simple-to-apply expected utility hypothesis is an inadequate representation of how individuals respond to risk. The literature in this area is reviewed in papers such as Kanneman and Tversky (1979) and Machina (1987) and describes not only evidence of consistent breaches of the expected utility hypothesis but also attempts to generalise this hypothesis into what is referred to as prospect theory. It is not clear how this research based on the study of the response of individuals to risk can be extended to public investments with long term environmental impacts.

In response to the difficulties of applying formal decision analysis to resource use decisions where there is an uncertain but likely low probability of an event which will cause large costs to a generation in the distant future, many argue for the application of the 'precautionary principle' via 'safe minimum standards'. Randall (1987) for example, pointed to the old distinction between risky events, whose probability of occurrence is known, and uncertain events, whose probability of occurrence is unknown. He argued that economics cannot adequately analyse uncertain events and proposed the use of safe minimum standards in resource use policy.

2.7 The Rights of Other Species

Randall (1987) pointed out that benefit-cost analysis generally applies what he refers to as 'a particular set of utilitarian ethical premises' (p. 414) including the premise that nonhuman components of the environment only have value to the extent that humans use or care about them. This anthropocentric basis of valuation is strongly opposed by some conservationists. Clearly it is an issue that has the potential to be highly emotive. If rights are granted to non-human species then the preservation of other species may have precedence over the welfare of very poor humans.

Randall (1987) noted two alternative bases for valuation. One system is based on a very strict interpretation of the concept that humans have a duty of stewardship over other elements of the biosphere. Another alternative is a rights based ethical system based on for example, the ability of animals to feel pain. This issue is also discussed in Kneese and Schulze (1985).

It should be noted that this choice of the basis for valuation is an ethical choice for society. It is well beyond the scope of either economics or ecology. It is also worth noting that the issue is far broader than whether resources are valued in terms of money or some other measuring rod.

Much of the discussion in this paper implicitly assumes an anthropocentric viewpoint in valuation.

2.8 The transdiscipline of ecological economics

In response to these concerns about the adequacy of neoclassical economics to address issues of sustainability, a new professional group who refer to themselves as 'ecological economists' has emerged. Costanza et al. (1996) describe ecological economics as **not**:

> 'analysing or expressing ecological, social and economic relationships in terms of concepts and principles of any one discipline. It is thus not merely ecology applied to economics nor is it merely economics applied to ecology. It is a transdisciplinary approach to the problem that addresses the relationships between ecosystems and economic systems in the broadest possible sense in order to develop a deep understanding of the entire system of humans and nature as the basis for effective policies for sustainability'.

There are a number of points to note about Costanza et al.'s description of ecological economics. First, it is clearly intended that the so called transdiscipline of ecological economics be more removed from economics than subdisciplines such as agricultural economics and health economics, although it is not clear how this is to be achieved. In some unspecified way the body of learning or the discipline of ecology contributes to economic theory and vice versa. When we speak of health economics, agricultural economics, transport economics etc, we are generally referring to the application of economic principles to issues of resource allocation in the health, agriculture and transport industries. There is neither a presumption that economics contributes to understanding diseases of humans, plants or machinery nor that physiology, agronomy or engineering contribute to economics principles.

Perhaps we can regard the emergence of a transdiscipline known as ecological economics as the emergence of what Randall (1993) would describe as a local provisional methodology. Using Randall's terminology it is unlikely that a demarcationist prescriptive methodology will emerge which will allow the conflicting hypotheses of 'economists' and 'ecologists' to be resolved - to demark and prescribe what is 'good science' in the analysis of natural resource issues. Instead knowledge in areas of interest to 'economists' and 'ecologists' might be most efficiently advanced if practitioners from these camps worked together to develop a local provisional methodology or a 'framework of reasoned and critical discourse … to enlighten and improve our discussion of ethical and value questions' and which 'draws upon all the tools of scholarship (including metamethodology), along with theories and procedures specific to the research program and its related disciplines' (Randall, 1993, p58).

Unfortunately the debate between 'economists' and 'ecologists' has often concentrated on limited representations of opposing viewpoints.

3. Externalities

3.1 Introduction

This Section begins by clarifying the distinction between externalities and off-site effects in both spatial and temporal dimensions. Then the implications of externalities for potential government involvement are examined and the relationship between the concepts of externality and sustainability is explored with a view to defining sustainability in a more objective way that allows an assessment of whether an agricultural system, for example, is becoming more or less sustainable. Towards the end of the section the analogies between efficient resource use in temporal and spatial dimensions, particularly with respect to the concept of marginal user cost, are highlighted and empirical approaches to measuring the divergence between individual and community interests as a guide to sustainability and the need for government intervention or collective action are discussed. Much of the material comes from Mullen, Helyar and Pagan (2000).

One of the key conclusions of the Chapter is that some of the techniques that have been developed to examine efficient resource use by firms through time can be used to measure the costs of externalities either spatially or temporally and hence indicate whether farming systems are sustainable or whether there is the potential for more efficient resource use through some form of government intervention or collective action.

Throughout this Section it should be borne in mind that, as already noted in the previous Section, the removal of externalities, while a condition for efficient resource use, is not a sufficient condition to optimise welfare over generations because it ignores equity considerations. Externalities may be removed to the extent considered efficient from the viewpoint of the present generation but a level of degradation may continue that future generations may not regard as being fair.

3.2 The distinction between off-site effects and externalities

As van Bueren and Pannell (1999) pointed out, the terms 'on- and off-site effects’ are often used ambiguously. This ambiguity extends to the term externalities and to the temporal and spatial dimensions of these terms.

In this paper on-site effects are the effects from resource use decisions on a particular unit of land at a point in time and off-site effects are the spillover effects on other units of land or on the same unit of land at different points in time. Hence off-site effects may have both spatial and temporal components, which is perhaps broader than the usual connotation. Temporal off-site effects have been referred to as carryover or feedback effects.

Externalities are a subset of off-site effects under this terminology, and arise if spatial and temporal off-site effects are not confined to those who cause them. Rather, they have an impact on neighbours and the community whose property rights are attenuated because they cannot choose the extent to which they are exposed to these off-site effects. Externalities may be thought of as off-farm effects where off-farm also includes future purchasers of the farm. The significance of the distinction is that in the presence of externalities there is a divergence in the interests of individual farmers and the community which leads to a rate of resource use by individuals that the community considers to be exploitative.

The classification of off-site effects and externalities also varies according to perspective (refer to Figure 3.1). From a paddock perspective, off-site effects are the effects from resource use decisions in the paddock at a point in time on other units of land (paddocks), or on the paddock itself at different points in time. From a paddock perspective some spatial and temporal off-site effects may be contained within the farm.

However externalities arise if spatial and temporal off-site effects are not confined to those who cause them (the current owners/ managers of the farm) but are experienced off-farm. From a farm perspective, some spatial off-site effects at a paddock level become externalities if they have an impact on neighbours. From a farm perspective, some of the temporal off-site effects may be borne by the current owners in the future and are therefore not externalities. Temporal offsite effects on future owners are externalities.

The interests of farmers are most strongly related to maximising the flow of wealth from the unit of land which they control. Their interest in externalities extends to mitigating the cost of externalities inflicted on them and this may include acting collectively with neighbours. The interest of the community is to maximise a measure of wealth aggregated over all individuals separated either spatially or temporally who are affected by the resource use issue under consideration. The interests of the community and the individual are the same in the absence of externalities.

Finally, from the perspective of the region or subcatchment, spatial off-site effects are externalities imposed on communities outside of the region. For example, downstream agricultural and other community members may be affected by upstream activities that affect water quality. Temporal off-site effects will be externalities where the effects are borne by people other than the current members of the region or subcatchment from which the effects originate.

3.3 Externalities and the Role of Government

Perhaps the most important reason for carefully distinguishing between off-site effects and externalities is the different implications they have for government. The consequences of off-site effects (as distinct from externalities) are confined to the resource user who initiates them and may involve inefficient resource use in either spatial or temporal dimensions. This inefficiency may arise because farmers are unaware of the off-site implications of their resource use decisions. The main potential source of market failure in this situation is that farmers would underinvest in research into off-site effects because of the public goods characteristics of the information generated by research. Traditionally government has provided support to agriculture in the form of research and extension services on this basis.

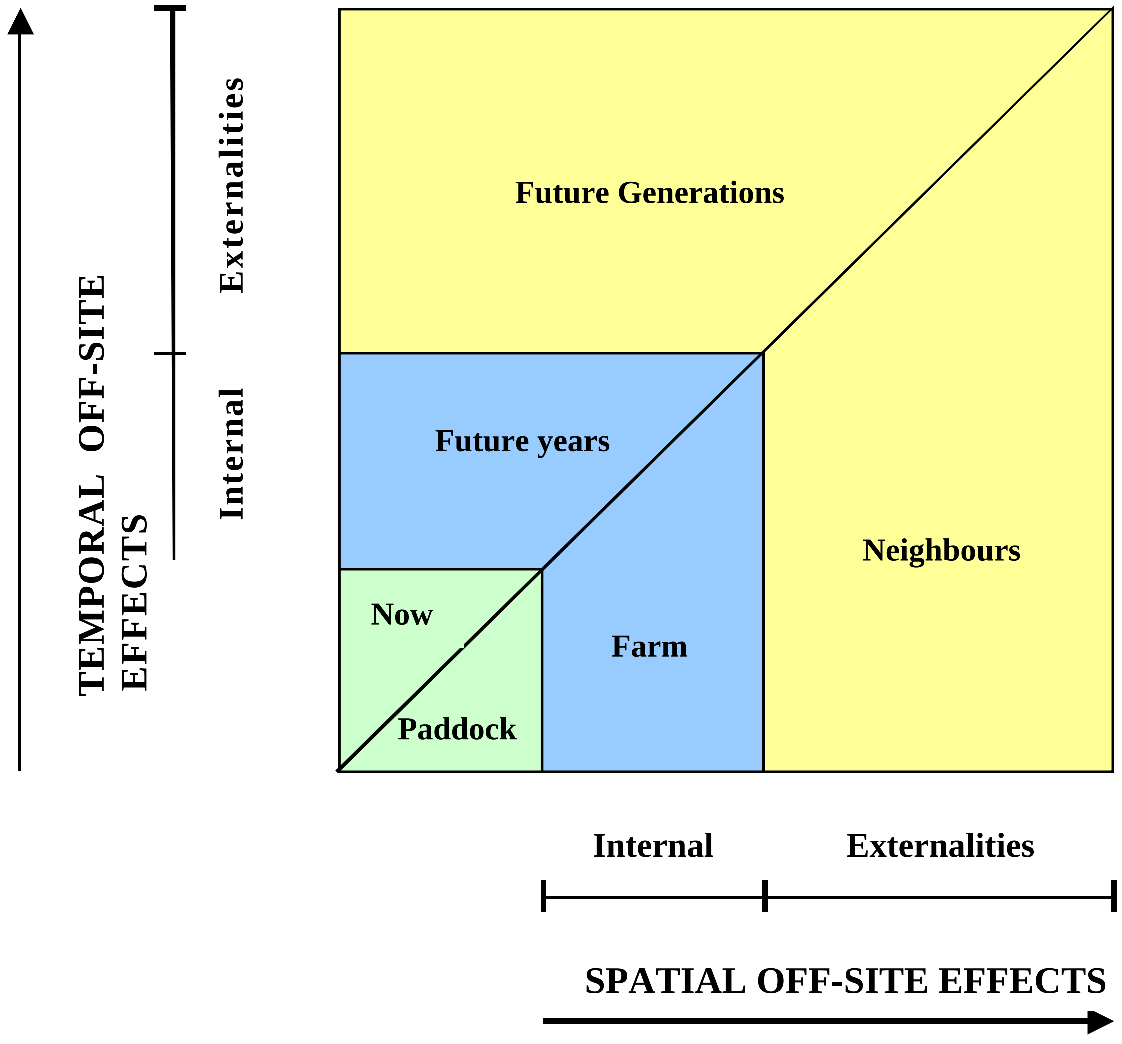
An important objective in developing the Research and Development Corporation model of research funding in Australia has been to increase industry funding and direction of research and extension. The ability to impose levies on industry participants is a mechanism by which industry can ‘solve’ market failure arising from the public good nature of research. In reviewing the evolving role of government in the provision of research and extension services, Mullen, Vernon and Fishpool (2000) noted that in response to slowly growing budgets and demands for new services, State Departments of Agriculture have responded by seeking more industry support for ‘industry good’ activities through ‘fees for service’ and RDC support. ‘The contribution of the RDCs may now be in the order of thirty per cent[6] (although more than half of this money comes from general taxation). It will be necessary and appropriate for this contribution to continue to grow if key research and extension services are to be retained that provide benefits almost exclusively to the producers, processors and consumers in particular agricultural industries (p. 643)’. There is also an increasing capacity on the part of the private sector to deliver these services.

Externalities are the consequence of market failure associated with attenuated property rights[7]. In the minds of some, ‘market failure’ is an automatic signal for some form of government intervention. Randall (1999a) used the term ‘isolation paradox’ to describe such situations where farmers acting alone have little incentive to consider their neighbours but where ‘everyone can enjoy a net benefit from coordinated action (p.30)’[8]. The attraction of expressing the problem in this way is that it points to a much broader range of responses that recognise the incentives facing farmers and their neighbours. ‘The isolation paradox concept, then, suggests an openness to solutions that invoke a variety of institutional forms: private enterprises, voluntary associations, and government from the most local to the national scale and beyond. Given the centrality of information and coordination, the array of feasible institutions is continually shifting as information, communication and exclusion technologies develop (Randall, 1999a, p.31)’.

It is not the purpose of this paper to review the range of market and non-market mechanisms which could be employed to counteract externalities (for this see Industry Commission (1997) or texts on environmental policy). However Mullen et al. (2000) pointed out that traditional research and extension programs which ignore the incentives facing farmers are an inadequate response to externalities. For example, there has been an increasing reliance on the suasive power of groups such as Landcare groups to achieve community goals. As Marsh and Pannell (2000) pointed out, suasive extension strategies to combat land degradation are likely to fail where they attempt to address community concerns by requiring farmers as individuals to take

[6] Including Commonwealth institutions such as ABARE and CSIRO who recover a high proportion of their salary expenses from the RDCs.
[7] Godden (1997) has a good discussion of externalities and the role of institutions and technology in establishing property rights.
[8] Marshall (1999) expressed similar views.

Figure 3.1: On-Site and Off-Site Effects from Spatial and Temporal Perspectives



actions that are not in their best interests. Marsh and Pannell (2000) queried whether such programs are ethical, particularly if they do not point out the costs farmers are expected to bear.

Market based mechanisms are in vogue as having the potential to provide more efficient solutions to externalities than traditional regulatory approaches. Mullen et al. (2000) pointed out that market based solutions to resource issues such as salinity, water quality, greenhouse gases and biodiversity have different research and extension requirements than traditional production issues. On the research side the scientific basis of property rights and trading instruments will have to be established for credible markets to develop. This will require information about the relationship between resource use strategies and consequent off-farm effects, not just at the farm boundary, but at the points in time and space where they express themselves.

Some change in the nature of extension programs is also anticipated. Extension resources will be required to encourage buyers and sellers to enter these new markets or collective activities to combat land degradation, in an informed manner. An important new role for extension in State Departments may be to explain to the broader community the interests of agriculture with respect to environmental issues. Other issues to which this may also apply include food safety and biotechnology. In this way the divergence between the interests of the community and farmers may be narrowed if the community through public participation and collective action, to use Marshall's (1999) terminology, becomes more aware of the costs of alternative land management strategies.

For all sources of market failure, the existence of *some* market failure is a *necessary* but not always *sufficient* condition for government intervention. The issue is the extent to which market failures constrain the capacity of farmers acting either individually or collectively from appropriating sufficient benefits from investments in the management of resources associated with externalities (in research and extension in the case of information deficiencies) such that there is significant underinvestment from society's point of view. Then there may be grounds for government intervention, subject to the usual concerns about the tractability of the issue and the cost of intervention relative to the cost of market failure. The aim should be to intervene to the minimum extent necessary to achieve community objectives.

Hussey pointed out 'public benefits frequently free ride private investments (1996, p.11)', hence implying that the costs of the externality have to be significant before government intervention is justified. However as the divergence between public and private interests widens, that is, as the opportunity cost to the farmer of adopting land use practices that impose fewer externalities becomes larger, we ought to expect that farmers are likely to be more guided by self interest and less inclined to adopt landcare technologies. Hussey noted that government need only become involved to the extent 'sufficient to secure the desired change in the behaviour of market participants'(1996, p.11) and this may be achieved by changing the incentives individuals face particularly through the specification of property rights.

Hence a key step in establishing *sufficient* conditions for government intervention in an externality situation is to estimate the costs associated with the externality. In the case of land degradation, the fallback position has been to value degradation in terms of production lost, sometimes referred to as the 'production equivalent of degradation' approach, by estimating the change in production at a farm or regional level and applying a value to this change in production. Problems with this approach are discussed in Gretton and Salma (Appendix E, 1996) and in Reeves et al. (1998) and are reviewed later in this paper. A key issue is establishing the benchmark from which production losses are measured. The benchmark is difficult to define but is generally taken to be an estimate of the production that could be achieved from the land in its most likely use were there no degradation. Rarely are the interests of individual landholders and the broader community distinguished and rarely are the benefits from reduced degradation related to the costs of associated resource management strategies.

At least conceptually a more rigorous approach to the empirical analysis of externalities can be found in the optimal control literature. However, while there are a number of empirical analyses of efficient resource use through time from the viewpoint of the firm, there seem to be few examples of the extension of this type of analysis to efficient resource use from the community's viewpoint, a step necessary to estimate the cost of an externality. As is explained more fully below, this requires comparing income flows to the community for a scenario in which farmers/firms pursue their own interests with one in which they take account of their impact on their neighbours. This difference in income is a measure of the extent of the externality which can be compared with the cost of intervention required to remove or ameliorate the externality.

3.4 Externalities and sustainability

As noted earlier, while the view that natural resources are being used unsustainably and that this ought to be a concern of government is widespread, there is little agreement on what constitutes sustainable resource use and the nature of government intervention required to achieve it. The broad intent of sustainable development has rarely been translated into specific management strategies for natural resources. The Kyoto agreement on greenhouse gas emissions was one of the first attempts at international cooperation in the management of a natural resource issue, yet it is unlikely that it will be ratified by all participants nor is it clear how the Australian government would intervene to ensure that Australia met its commitments.

Because of the vagueness of proposals for sustainable resource use, it generally has not been possible to identify and estimate benefits and costs to individuals or the community (for an exception see ABARE (1997) for an evaluation of climate change). Hence, as Beckerman (1996) suggested, without an attempt to weigh up benefits and costs, some policies for sustainable development may actually reduce the welfare of future generations as well as the present generation.

At the operational level of institutions such as Departments of Agriculture, sustainable development has been interpreted to mean the identification of production technologies and cropping rotations that conserve dimensions of soil and water quality that are at best, slowly renewable. A problem with these technologies is that often they have been unprofitable for farmers. The issue of why landcare technologies are not more widely adopted is discussed more fully in the next section and was raised in the discussion above about the role of government. However a key reason is that because of attenuated property rights, the interests of individual farmers and the community diverge.

This divergence in the interests of farmers and other resource users and the community suggests a more practical definition of sustainability, at least from the perspective of public institutions. A farming system or resource use regime is more sustainable if it does not impose externalities on the community (although fairness to future generations may remain an issue).

Defining sustainable resource use as resource use strategies that are consistent with the wishes of the community is in one sense empty because it is subsumed within an objective of maximising net social welfare over a number of generations. This definition would also be objectionable to those who do not accept the utilitarian approach to valuation that is the basis of the benefit/cost framework used by economists. These issues were discussed in Section 2.

However, defining sustainable resource use in this way does have a number of attractions. It narrows the focus of attention from all resource use to resource use that has an impact on the community. Hence it aligns sustainable resource use more closely with a recognised source of market failure, attenuated property rights, than has often been the case in the past.

Defining sustainability in terms of externalities has the added attraction of providing a practical yardstick for whether particular patterns of resource use are more sustainable than others. This measurement issue is discussed more fully below, however the significance of externalities can be estimated by comparing the flow of income to society when individual resource users take no account of their impact on others against a scenario in which resources are used in a way that maximises the flow of income to society. If the difference is small then the cost to the community is small and successful intervention to correct the externality unlikely. If the divergence is large then there is a concern that resources are not being used sustainably and the full range of intervention measures needs to be considered to find that which is most efficient.

Whilst conceptually it is possible to estimate the significance of externalities in this way, there remain the problems, already discussed, of valuing flows of environmental services over time which are often unpriced. In particular there are the related problems of the choice of a discount rate and the rights of future generations. Note also that as the community values environmental services differently through time, so patterns of resource use considered sustainable today may be considered unsustainable in the future. Hence the real difficulty is agreeing on the rights of future generations and anticipating how they will value particular environmental resources. Randall (1987) suggested a role for ‘safe minimum standards’.

Finally it should be pointed out that defining sustainability in terms of externalities does not mean that the solution to all externalities and sustainability issues lies solely in addressing property rights issues. As already noted some off-site effects arise in part because resource users are unaware of the impact of their actions on their own wealth let alone those of their neighbours. Another source of market failure is underinvestment in research by individuals because of the public goods characteristics of information. Overcoming these deficiencies in information may lead to a change in resource use that has benefits to both the individual and the community without having to resolve issues of attenuated property rights.

3.5 Resource allocation when there are temporal off-site effects

There has been much interest in modelling technologies that have temporal off-site effects (Kennedy, 1988). The process of soil acidification through time is a good example but other examples include the more general problem of nutrient carryover (Godden and Helyar, 1980) and the growth of seedbanks in a weeds context (Jones and Medd, 2000 and 1997). These resource management issues are dynamic in the sense that they deal with a resource stock, such as soil acidity, which influences the level of current production but which in turn is affected, at least in the next period, by current management practices including decisions about liming and pasture and crop choice. There are feedback effects in both directions between the state variable, soil acidity, and control variables such as stocking and liming rates. Hence profit in any year depends not just on decisions made in that year but also on resource use decisions made in previous years.

McInerney (1976) demonstrated that optimal resource use occurred at the point where marginal benefit equals marginal user cost where the latter includes a measure of the benefits lost from using a resource now rather than in some later period - the opportunity cost of resource use in other words. Equivalently, the optimal solution to problems of this nature requires the maximisation of profit (including the terminal value of the asset) over a long investment period for the particular unit of land under consideration. In Kennedy’s terms, resource management decisions of a dynamic nature are not separable from year to year. Simply maximising income in each year without regard to these temporal linkages leads to exploitative resource use and lower aggregate income over the full horizon for which temporal effects persist. This is because the marginal user cost of resource use is being ignored. This suggests that a measure of the significance of these temporal effects can be gained by comparing for a farm, measures of wealth when the resources are used as though each year was separable with the situation when resources are used in a way that recognises temporal off-site effects.

These intertemporal effects do not result in externalities if all costs are borne by the present owners. However, when these intertemporal effects last for generations then the issue of intertemporal externalities does arise. This is because there is concern that present generations are likely to exploit resources at the expense of future generations. The problem is that while individuals and families have the incentive to optimally allocate their use of resources through time and there are financial instruments to help them, this is not the case for resource use across generations, as discussed in the previous Section.

Another cause of intertemporal externalities may be associated with imperfections in the land market. Intertemporal externalities may occur if the nutrient capital is reduced in an exploitation phase (ie. a period when nutrient additions are less than losses plus fixation), but the land value does not decline as much as the capital value of the lost nutrient reserve. It has been argued that land values do not always reflect the degraded state of important dimensions of land quality and hence this failure of land markets encourages degradation and imposes an externality on future generations.

The empirical evidence to support this hypothesis is limited. A study by King and Sinden (1988) of land values in the Manilla Shire of NSW where soil erosion was a problem found that the land market appeared to be working satisfactorily but the hypothesis remains to be tested for less visible soil degradation problems such as structural decline and soil acidity. If this is a problem then one remedy is the provision to buyers of objective information about land quality. There seem to be few barriers to the emergence of such a method of selling.

Again, the question arises as to how to measure the significance of these off-site effects, now externalities. In a similar fashion to the measurement of off-site effects for individual landholders, a measure of the divergence between individual and community interests could be derived by comparing the net wealth of a farm managed to maximise wealth over several generations with the net wealth of a farm managed to maximise the wealth of each generation with no concern for other generations. An important source of wealth to each generation is the value of the land. This appears as a terminal value in calculations of wealth over time. If the land market is efficient the terminal value should reflect the net present value of the future stream of income from the farm (plus any real estate potential) and hence its quality with respect to nutrient status, acidity etc. Hence the two measures of wealth will diverge if the market value of land of interest to individuals is greater than the stream of income that flows from it to future generations.

3.6 Resource allocation when there are spatial off-site effects

The treatment of spatial effects or (contemporaneous) externalities would seem to be analogous to the treatment of intertemporal effects although the concepts of efficient resource use and empirical experience gained in analysing resource use through time do not appear to have been widely applied in a spatial dimension[9]. In McInerney's (1976) terms, resources are used to the point where marginal benefits equal marginal user cost, where the latter now has a spatial rather than temporal component and represents the losses on other units of agricultural land and other costs to the community. Note that these spatial effects may well be separated in time as well.

In the case of salt mobilisation by water flows for example, there are potential off-site effects associated with groundwater and surface water systems. Some off-site effects are contained within farm boundaries and the farmer has to assess the significance of off-site costs in discharge areas relative to increased production in recharge areas. Dryland salinity might arise because the landholder may judge that the higher production from removing trees in the recharge area may more than offset the production lost to dryland salinity in the discharge area.

However some off-site effects are potential externalities because of their likely impact on neighbours. The groundwater effects are experienced on land within the catchment or sub catchment that is hydrologically related to the land where the initial resource use decisions are being made, through a shared water table. Hence the land subject to off-site effects may be quite extensive, it may be quite a distance from the original site and it may take many years for the effects to be experienced. In addition, groundwater effects eventually feed

[9] The literature of production economics contains similar problems which may provide useful insights to the issue of land degradation being addressed here. In particular the literature concerning the economics of horizontal and vertical integration of firms and transactions costs may be of interest.

through to the surface water systems in the form of increased salt in the stream flow. Because the resource user does not bear the full cost of how inputs are used on a particular unit of land (or cannot capture all the benefits of input use on a particular unit of land), there is a divergence between the interests of the person who owns the unit of land and those whom his resource use decisions affect.

For some externalities there is a relatively symmetric effect on all neighbours or users of the resource. Common examples here include the grazing of the ‘commons’, and the use of a fish stock or watertable. In these cases the resource user reduces that stock of the resource that is available in the future both to himself and to his neighbours and the impact on the stock is dependent only on the size of the stock at that time and not on the identity of the user. In this case it is in the self interest of all if they can act collectively to control the exploitation of the resource.

At the other extreme is the case of a clear demarcation between those who cause the externality and those who bear the cost. The obvious example here is dryland salinity. The effect on a watertable of removing trees clearly depends on the ‘identity’ or location of where this occurs. Removing trees in the recharge area may benefit farmers there but cause losses to those in discharge areas. In this case it is not in the self-interest of those in the areas where recharge of the water table occurs to act collectively with those in discharge areas to protect the resource stock, the watertable. Depending on how property rights are defined, those in recharge areas will have to be either taxed or compensated to control their land management (e.g. tree felling) activities.

Many land degradation issues fall between these two extremes of perfectly symmetric and perfectly asymmetric impacts, including acidity and erosion. In these cases the production possibilities through time of a farmer are influenced both by the way he uses a natural resource stock, land, and by decisions made by neighbours upstream of him. In turn he has an impact on the production possibilities of downstream neighbours. Hence, it may be in the self-interest of farmers to ameliorate to some degree the land degradation on their own unit of land caused by their own actions. It may also be in their self-interest to ameliorate the degradation on their block caused by the actions of upstream neighbours, and collective action may be an efficient way of doing this.

As for the case of temporal off-site effects, a measure of the importance of spatial off-site effects is provided by the difference in wealth were all units of land linked spatially by off-site effects managed as one unit, as compared to the ‘real world’ situation where many spatially linked units of land are managed independently. This principle can be applied both at a farm level where there are off-site effects within the farm boundaries and at a catchment level where there are externalities from a farm or regional perspective.

This approach was used by Quiggin (1991) in his study of salinity in the Murray[10]. Quiggin modelled the different stages of the river with six representative farm models and a seventh stage for urban water use in Adelaide. He first estimated the total profit to the seven regions in the river under an open access regime by solving a series of six linear programming models, where a constraint on the downstream farms was the quality of the water available after the unconstrained management decisions of the farms upstream. He then simulated a common property regime by formulating the problem as ‘a dynamic programming

[10] Barton (1992) used a similar approach.

problem in which the stages of the river take the place of successive time periods in a standard dynamic programming problem (p.57), and the objective is to maximise the profit from the farms operated as a group. He found that profit under the common property regime was higher than under the open access regime indicating the extent of the externality problem[11]. It is important to note that under Quiggin's (1991) approach, farms have a capacity to adjust enterprise mix in response to degradation, and that some level of degradation may be optimal even from the community's viewpoint (if not from the viewpoint of future generations). Hence this approach provides more conservative estimates of the cost of externalities.

In an application to dryland salinity on the Liverpool Plains, Greiner (1988) developed representative farm models for four areas within the catchment with significantly different biophysical and hydrological features. Each of these farms was linked hydrologically in such a way that resource use in the recharge area affected the watertable in the discharge area which in turn affected production in the discharge area. A difficulty with the representative farm model approach is that some consistent aggregation process is required. To spell this out more fully, while the farm models may represent their environment in being of average size for that environment for example, it is likely that their results will have to be scaled differently to reflect the relative sizes of the biophysical environment they represent. To fully reflect the interests of the community, the impact on downstream users (transmitted via the surface water systems) have also to be accounted for.

3.7 The need for monitoring

In addition to the need to measure the costs of off-site effects and externalities to assess whether intervention is warranted, there is also a need to be able to monitor changes in the resource flows of interest so as to be able to assess whether management interventions have been successful. There is a large literature related to monitoring and sustainability issues. Some of this literature is reviewed in a special forthcoming issue of the *Australian Journal of Experimental Agriculture* (2001) (Farquharson, Mullen and Schwenke, 2001). An economic perspective on the value of monitoring can be found in a paper by Glenn and Pannell (1998). While no attempt is made here to thoroughly review this literature, the type of information required in the context of externalities is discussed.

The nature of externalities means that they are most appropriately measured at the site of the affected party rather than at the site of the one who causes the externality. This situation creates a challenge, as it is also necessary to relate the cost of the externality to some action by those causing the externality. This requires knowledge of the economic cost borne by recipients of the externality, and a clear understanding of the technical processes through which actions by the originator cause the externality. An additional difficulty in developing indicators is that there may be a significant stochastic element to important hydrological and biological processes.

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[11] It is not clear to us that Quiggin's results hold generally as would appear to be the case for the temporal case where feedback effects are complete. For soil acidity for example, the production function for the unit of land in any period is a function of soil pH which is turn is a function of agricultural practices. In the case of dryland salinity however it appears the feedback effects are not complete. In discharge areas the production is a function of accessions in recharge areas but the reverse may not apply.

The possession of the level of technical and economic knowledge outlined above would allow the development of efficient, enforceable mechanisms which address externalities of concern by directly mitigating adverse effects in accordance with the severity of their impact. Obviously however, this level of technical and economic knowledge is rarely known and, second best solutions are the only alternatives at present.

Consequently, it should be recognised that even with more appropriate recognition of off-farm effects in biological and economic research of alternative land management strategies, as is advocated in this paper, there remain significant problems in the translation of this knowledge into appropriate policy responses that can be implemented efficiently.

3.8 Conclusions

The objective of this chapter has been to note and perhaps clarify some of the ambiguities surrounding off-site effects and externalities relating to land degradation issues such as soil acidification and dryland salinity and to note the importance of being able to measure the extent of externalities to guide efficient intervention by government. Sustainability is a notoriously vague concept. Here it is suggested that a clearer role for government can be identified by defining sustainability in terms of the presence of externalities, although this may not account fully for obligations to future generations. A sustainable farming system is one that does not impose significant externalities and hence the role of government in promoting sustainable development lies in developing mechanisms to alleviate attenuated property rights in situations where the costs of externalities are large.

The terms off-site effects and externalities are often used loosely and interchangeably in the literature. One common usage is that the terms off-site effects and externalities have a spatial dimension whereas the terms carryover effects and dynamic are associated with resource use through time. In this paper externalities have been defined as a subset of spatial and temporal off-site effects which are not confined to the person making the resource use decisions.

There are clear analogies between conditions for efficient resource use through time and across space. Farm managers need to account for temporal and spatial off-site effects within their farms as components of the marginal cost of using natural resources, if they are to maximise wealth from their farm over time. From a community viewpoint, the objective is to maximise wealth through time, from farms and households managed jointly as though they were common property.

An important distinction is that for externalities, the interests of individuals and the community are likely to diverge, creating grounds for potential government intervention. This is generally not the case when off-site effects are contained within a farm (temporally as well as spatially). For government to be involved in ameliorating externalities, the potential efficiency gains (net of the costs of intervention) have to be of a similar order of magnitude to other uses of public funds and there has to be a practical means of intervention.

A guide to the significance of potential efficiency gains may be provided by the divergence between community wealth when land is managed in a way that accounts for temporal and spatial externalities, as opposed to the wealth of individuals when these linkages are ignored. Quiggin (1991) used the terms open access and common property to define these regimes. The significance of off-site effects within a farm can be gauged using a similar rule.

4. Are Environmental Resources Becoming Scarce?

The objective of this section of the Report is less ambitious than its title might suggest. The objective is to briefly, and somewhat selectively review, some of the literature which raises questions such as:

- Are environmental resources adequate to support a growing world population?
- Is Australian agriculture sustainable despite evidence of degradation?
- What motivates farmers to adopt landcare technologies?

The Section starts with broad questions concerning factors that influence the demand for and supply of natural resources at a global level and then, in the light of these global influences considers the sustainability of Australian agriculture and the response by farmers to land degradation. These issues are closely related to the discussion in Section 2 concerning how the present generation meets any obligations it might have to future generations.

4.1 Can Living Standards be Maintained as World Population Grows?

Clearly this generation has higher levels of consumption than previous generations. A number of recent papers have pointed to the large improvement in living standards in most of the world over the past century (Easterlin, 2000). Whether succeeding generations will similarly have higher consumption levels is uncertain and the subject of a growing literature. This question is posed out at two levels - the prospects for the people of the world to continue to become wealthier, and the prospects for the population of the world being able to feed itself.

Concern about the ability of economies to grow being restricted by a finite supply of natural resources is not a new phenomenon. A well known example of such concern is the report by the Club of Rome in *The Limits to Growth* (Meadows et al., 1972) containing predictions about reserves of key natural resources. Such projections of future resource availability are generally based on current technologies and reserves. Economists argue that these types of projections are always likely to be unduly pessimistic because they ignore the responses of consumers and producers, discussed above, to changes in relative prices. Beckerman (1996) pointed out that known reserves in 1989 exceeded those identified by the Club of Rome in 1970 and in fact, the reserves at 1970 for some resources had already been consumed during the 19 years to 1989.

A less sanguine review of the adequacy of natural resources is that of Smith (1980) who warned that inadequate economic theory and methodology and knowledge of technology meant that drawing definitive conclusions was fraught with danger. Clearly we can’t know what is going to happen with certainty even a few decades out and hence there does not appear to be any easy way to reconcile these viewpoints.

Concern about the world’s ability to feed itself also has a long history and a wide literature. Well known proponents of the view that a food crisis is imminent include Malthus (1826), Paul Ehrlich (1968) and Lester Brown (1994). Some recent studies that have found no evidence of an impending food crisis include Beckerman (1996); Duncan (1997) and Johnson (1997). Duncan (1997) has pointed to evidence that since the 1950s, the growth in food production has exceeded the growth in population in most parts of the world except sub-Saharan Africa and that the real price of food commodities fell by 78 percent between 1950 and 1993 while real per capita GDP has been rising in developing countries. Duncan (1997, p.16) concluded that:

> ‘With population growth continuing to slow and the most rapid phase of growth in food consumption now past for most of the world’s population, the rate of growth needed to meet the expected effective growth in demand is much lower than it has been in forty years. ….. It is therefore highly likely that the world prices of grains will continue to fall in real terms….’

Review papers marking the bicentenary of the work of Thomas Malthus in a recent issue of Choices (Third Quarter 1998) warned that ‘world food demand-supply balance is likely to be tighter over the next three of four decades than in recent decades’ (Tweeten, 1998, p.11) and of ‘potentially larger fluctuations in food production and prices and higher associated risks of for insecurity for the world’s most vulnerable countries and people’ (Pinstrup-Anderson and Pandya-Lorch, 1998, p.7) but did not project a period of rising commodity prices. A threat to the rate of growth of food production is the concern by consumers in some developed countries about food quality and safety issues and concern for environmental outcomes associated with the use of pesticides and biotechnology.

The objective here is to review theoretical insights and limited empirical evidence concerning scarcity using as a framework the demand for and supply of natural resources.

4.1.1 Factors Influencing the Demand for Natural Resources

The two main factors influencing the demand for natural resources are population and income growth but the demand for resources is also influenced by whether users of resources can substitute between them as some become scarce relative to others. These three influences are discussed in turn.

Clearly as world population grows so does the pressure on natural resources. More resources are used in production and consumption activities and there is greater demand on the environment as a disposer of waste and a source of aesthetic pleasures.

This century has seen population grow at unprecedented rates. No doubt this has fuelled much of the concern about sustainability. Projections concerning world population are reviewed in Tweeten (1998). He noted estimates of world population and the year in which zero growth is achieved ranging from 11.3 billion in 2128 by the World Bank to 7 billion in 2030 by the Population Research Institute. A schematic (courtesy of Professor John Dillon) of the trend in world population based on these sources is presented in Figure 4.1. The possibility of a declining world population within the lifetime of some of us would appear to have profound implications for agriculture and the demand for natural resources more generally. Population growth has offset the long term pressure on prices from advancing technology. If population stabilises or declines then the pressures for rural adjustment in the face of an ongoing cost/price squeeze will intensify.

Another important demand side influence is the growth in per capita income. GDP per capita is presently growing in most of the world at an annual rate of about 2.5 percent (Easterlin, 2000), Sub-Saharan Africa being the main exception. At this rate of growth, Easterlin pointed out that parents have only a half of the resources of their children at the same stage in their life cycle[12]. He went on to conclude that ‘As a new century opens up, the prospect of a winding down in the advance of living standards – absent some terrible catastrophe – seems less probable today than when Mill wrote a century and a half ago (p. 23)’.

As income grows we would expect the demand for natural resources also to grow but will the rate of growth in demand exceed, keep pace with or be slower than the rate of growth in income? No attempt has been made to thoroughly review the empirical literature in this area. Rather the four types of services from natural resources identified in Section 1 are briefly reviewed for characteristics that are likely to influence the way in which demand for them is likely to respond to income changes.

With the exception of waste disposal services, these services are normal goods in that as income rises we would expect the demand for these services to also rise. However the demand for life support and amenity services is likely to rise more quickly than income per capita. The growing concern for the environment in developed countries is evidence of this.

Offsetting this is the continued growth in demand for natural resources as inputs in production of food and other consumption goods. We know that the demand for food at least in developed countries is rising less quickly than per capita incomes. Once people reach an adequate plane of nutrition, a one percent growth in income results in a less than one percent growth in the consumption of foodstuffs as this growth in income is diverted to other consumption activities.

Offsetting the slowly growing demand for food is a higher rate of growth in the demand for heating and cooling and transport but presumably, as for food, once an adequate level of consumption is reached the pressure on natural resources use becomes less.

Turning to the use of the environment to dispose of wastes, we would expect that this is an inferior good. As income rises and as waste disposal activities begin to have significant effects on the environment we would expect that people would search for ways to dispose of waste that are less harmful. There is evidence of this trend in developed countries.

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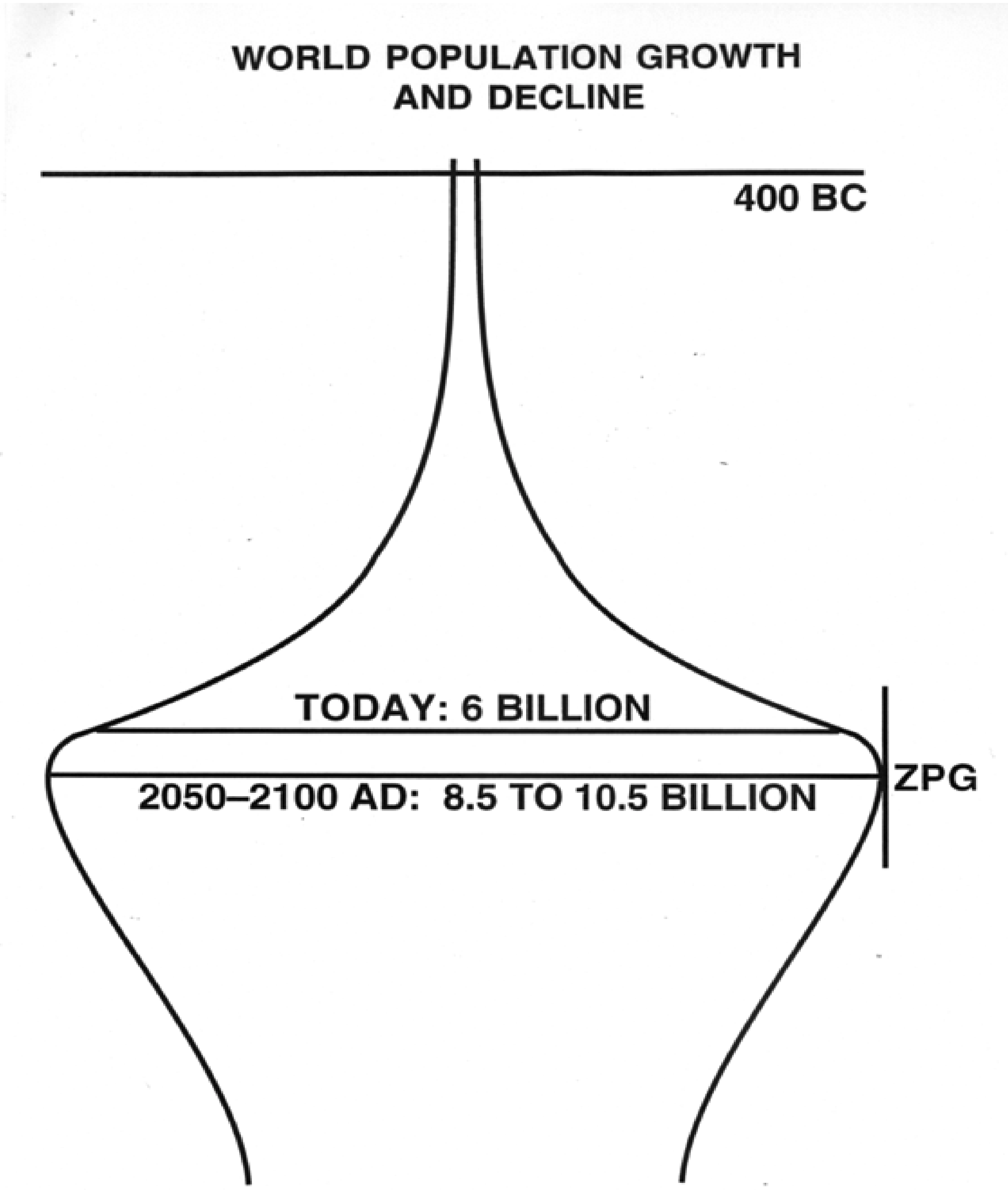
[12] Easterlin (2000) noted rapid growth in other measures of wellbeing including life expectancy and literacy.

Figure 4.1: World population growth and decline



Hence while we would expect rising incomes to increase the pressure on natural resources, the rate at which this pressure is increasing is perhaps slowing partly because the demand for natural resources is changing towards life support and amenity services.

Input substitution is the process by which producers switch between inputs in response to relative price changes. The literature of economics is somewhat ambivalent with respect to input substitution. Textbook accounts of production theory usually presume a high degree of substitution yet empirical studies of derived demand and price spread behaviour have often assumed that farm and non-farm inputs are used in fixed proportions as pointed out by Mullen, Wohlgenant and Farris (1988).

There is little empirical evidence about the extent of substitution between natural and manufactured resources. At the level of an individual firm or household, substitution possibilities might be quite limited. Smith (1980) noted that while micro-level studies of particular technologies suggested that substitution between natural and other resources was low, there were dangers in extrapolating these findings to an aggregate level. Diewert (1981) pointed out that at the industry level where firms can choose, at least in the long run, from a range of technologies that use inputs in (almost) fixed but different proportions, substitution possibilities may be larger as firms switch between technologies. In agriculture there are many technologies combining inputs such as soil and water and manufactured inputs such as fertiliser in different ways to produce a given level of output.

4.1.2 Factors Influencing the Supply of Natural Resources

Key supply side factors include the discovery of new sources of natural resources, and the development of new technologies that save on the use of natural resources in production activities or allow more efficient extraction or recycling of natural resources. Economists argue that, as for demand, the supply of resources responds to changes in relative prices so that as a resource becomes scarce and its price rises relative to prices of other resources, there are incentives to use relatively more of other resources in production or consumption; to develop technologies that economise on the use of scarce resources or allow them to be efficiently recycled; and to discover new sources of natural resources or allow the development of already known sources that previously were too expensive to exploit.

Historically technical change, exploration and discovery of new sources of resources and, more lately, recycling, have been important sources of productivity growth. The term technical change will be used to encompass these three sources of growth. Some view technical change as an exogenous event - ‘manna from heaven’ - but many ‘economists’ support the induced innovation hypothesis put forward by Hayami and Ruttan (1985) that the rate and direction of research and innovation is influenced by changes in factor prices and endowments as just described. Empirical testing of this induced innovation hypothesis has proved elusive and was critically reviewed in a paper by Olmstead (1999).

Smith (1980) concluded that empirical work to support the hypothesis that technical change has been saving of natural resources is difficult and evidence is sparse. This is especially true for short run although it is not clear what Smith regards as the short run. From a long run perspective he accepted Rosenberg’s (1973) conclusion that:

> ‘There is no obvious reason why the future growth of technological skills should not make it possible to continue the shift from dependence on scarce sources of materials to dependence upon more abundant sources’ (p. 117).

‘Ecologists’ are likely to question whether the historical rate of technical progress can be maintained. Another concern is that if natural resources are poorly priced, the incentive for innovation in their use is less strong than it should be from society’s viewpoint (Coxhead (1996)). It is interesting to observe the emphasis now given to environmental objectives in much rural research in public institutions reflecting in part a greater realisation of the role of the public sector in ameliorating externalities and, more generally, in meeting community expectations concerning the use of natural resources .

4.2 Is Australian Agriculture Sustainable?

Retreating from this global perspective, perhaps a more practical but no less difficult question is whether Australian agriculture is sustainable. The objective here is to briefly point to some of the literature that reviews this question.

Agriculture has made a major contribution to the economic development of Australia but not without some costs in the form of land and water degradation. From casual observation we are all aware of pockets of land degradation but the more widespread significance of these problems to production is unclear. At this more general level, a typical observation about the productivity of common agricultural cropping and pasture systems is that by White et al. (1999) which I would paraphrase as a decline through time in crop and/or animal production from a unit of land for an agricultural technology which is unchanging in terms of tillage and fertiliser practices. The decline in production can be attributed to induced processes such as soil acidification and salinisation. Much of this evidence about this perceived decline in productivity is anecdotal rather than empirical.

The nature and extent of specific types of land degradation are reviewed in the last Section. However Gretton and Salma (1996) pointed to the lack of consistency in approach and the lack of continuity of such studies. Reeves et al. (1998) argued that there was no clear evidence of a trend in the cost of land degradation. The hope is that the recently commenced audit of natural resources will provide clearer evidence and a baseline against which to track future degradation.

In contrast to this somewhat anecdotal and discontinuous evidence are the empirical measures of total factor productivity (TFP) which relate the growth in outputs to the growth in inputs[13]. Mullen and Cox (1996) using ABARE farm survey data, estimated that productivity in broadacre agriculture in Australia, graphed in Figure 4.2, grew at a rate of 2.5 percent per annum from 1953 to 1994[14] which is high relative to other industries in Australia and high relative to agriculture in other countries.

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[13] Note that TFP measures do not hold technologies constant and this may reconcile the anecdotal and TFP approaches.

[14] The qualifications to using TFP as an indicator of sustainability are that there is a presumption that land degradation is reflected fully in land prices and the failure to account for externalities imposed by agriculture on the broader community.

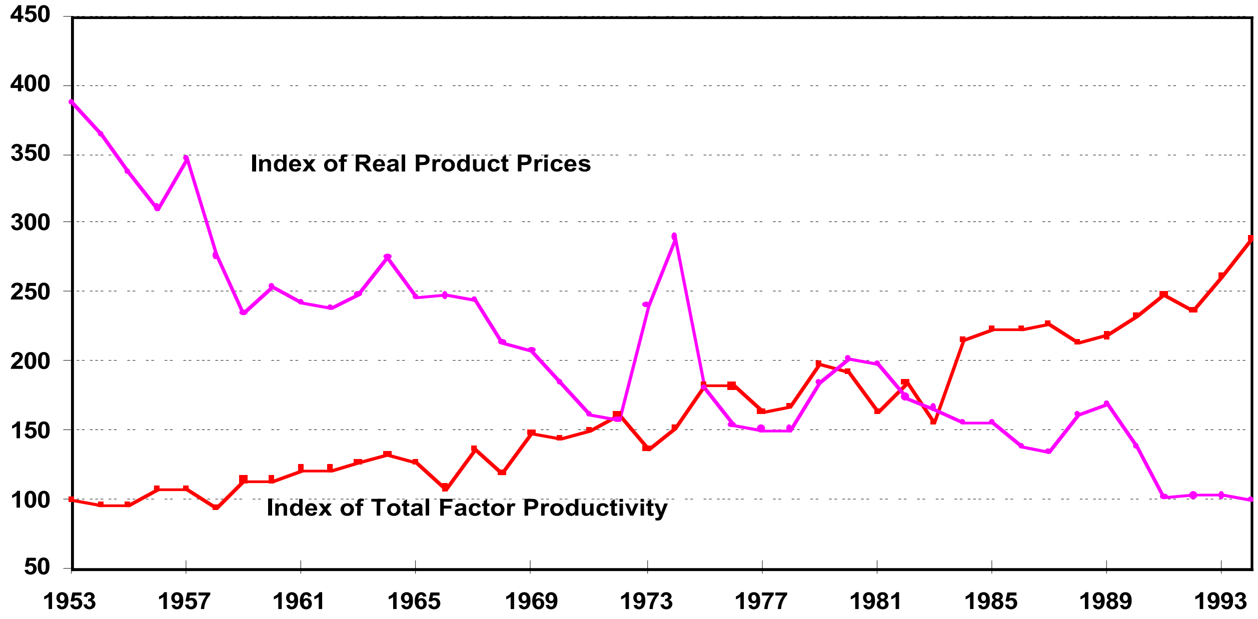
A common criticism of TFP estimates is that they do not reflect the cost of resource degradation. This criticism is only partly valid. TFP estimates based on farm survey data include both production losses associated with degradation and the expenses farmers incur in a range of conservation activities and arising from regulations to protect the environment. They inadequately reflect costs associated with environmental outcomes imposed on those in the broader community. There is much interest in developing measures of productivity (Repetto, 1996) and GDP (Solow, 1993) adjusted for resource use.

In this spirit of accounting for degradation, Chisholm (1992) pointed out that one widely used estimate of the cost of degradation in the form of lost agricultural production was $600m per year in 1988/89 dollars which he translated into a farm level cost of around $4,800 per year. Adjusting TFP by this estimate of the cost of degradation only reduced the growth in TFP by 0.1 percent although this does not account for the environmental costs to the broader community[15].

Chisholm's paper remains the broadest review of the sustainability of Australian agriculture. While he accepted the view that land degradation has seemingly had a very minor impact on the productivity of Australian agriculture and hence agriculture remains sustainable from this viewpoint, he did question the sustainability of agriculture from a socio-economic viewpoint, which he viewed as a more relevant measure of sustainability. The real value of agricultural production in Australia has declined and there are strong adjustment pressures in many agricultural industries raising concerns about the socio-economic sustainability of industries and regions in Australia. An index of the real price of aggregate output from the dataset used by Mullen and Cox (1995) is also graphed in Figure 4.2 and exemplifies the trend of concern to Chisholm. The index of real prices fell much more quickly than TFP rose over the period 1953 to 1994. Note that there are weak and strong versions of socio-economic sustainability as well.

[15] Note that this is the sort of adjustment process being recommended for broader measures of economic performance such as GDP.

Figure 4.2: Indices of Productivity and Prices



4.2.1 Salinity

The issue in Australia causing the greatest concern about agriculture's impact on natural resource systems is salinity. An important cause of salinity has been an increase in accessions to groundwater systems and subsequent rises in watertables caused by the substitution, over many decades, of annual cropping and pasture system for perennial native vegetation in the course of the development of agriculture. Salinity threatens not only agriculture through dryland salinity and reduced water quality for irrigation but the community as a whole through its impact on water quality in Australia's major river systems and damage to public and private infrastructure. Water quality is important not only for human consumption but also for plant and animal life in wetlands systems

Salinity is reviewed in more detail in Section 6 and in an important paper by Pannell (2001), however it would seem that there are likely to be continuing high costs to the community in the form of either continuing land and water degradation or in developing and implementing technologies and land use strategies that mitigate contributions to the salinity issue.

The management of salinity would seem to pose difficult dilemmas for those who hold doctrinaire versions of either weak or strong sustainability. On the one hand, it is likely that some important natural resource assets will be threatened by escalating salinity and hence some generalised level of saving and investment in technology by the community will be regarded as an inadequate and inequitable result. On the other hand, simply revegetating the landscape immediately is likely to impose intolerable costs on present and near generations and at least in the short term, prove futile as the encroachment of salinity continues many decades into the future until a new equilibrium in watertables in reached. As Pannell (2001) pointed out easy solutions are not apparent but elements of a pragmatic public response would include investment in the development of technologies based on perennials and in salt tolerant technologies and in engineering solutions to protect particular assets valuable to the community.

The gulf between doctrinaire ‘ecologists’ and ‘economists’ is perhaps illustrated by the bid by an unlikely coalition between the Australian Conservation Foundation and the National Federation of Farmers asking that the Commonwealth Government spend $65 billion over ten years on salinity and the Commonwealth and State Governments’ response to date in the form of a National Action Plan involving an expenditure of $1.4 billion over seven years.

It is interesting to speculate about the nature of agriculture and the Australian economy had agriculture not followed its historical path of development that has led to present salinity problems and which land use strategy future generations would have regarded as being most fair to them.

4.3 The Adoption of Landcare Technologies

In view of concerns by the community and by farmers themselves about land degradation associated with agriculture, there is some concern at the low level of adoption by farmers of landcare technologies (Marsh and Pannell (1997)). The reasons for this low adoption are not well understood. The stock response is that farmers continue with unsustainable technologies because of adverse economic circumstances (from either low prices or poor seasons) - ‘it’s hard to be green when you are in the red’ (Price (1996, p.6)). Often the corollary is that they have to sacrifice long run profitability from sustainable systems to meet short term financial commitments. However several interpretations can be placed on statements like these. For example, it is not clear whether landcare technologies will be adopted on a return to the ill defined good seasons and good prices or whether agriculture is in such long term decline that landcare technologies will never be an option or whether the problem is a failure in financial institutions affecting agriculture. It is important to explore more carefully why landcare technologies may not be adopted widely if we hope to influence their adoption.

One reason for the low rate of adoption may simply be that landcare technologies are unprofitable. Many decisions about resource use on farms are the prerogative of farmers acting in their own best interest. However the term landcare technologies usually denotes a concern on the part of the community about the environmental outcomes of resource use decisions by farmers. In Section 3 the nature of externalities and their implications for government were discussed in detail. Almost by definition, the existence of externalities means that there are some technologies that the community would like to see farmers employ, landcare technologies, but which are relatively unattractive to farmers. This divergence means that it is not in the self interest of farmers to adopt landcare technologies to the extent desired by the community.

Godden (undated) identified three broad causes of market failure where the interests of the community and farmers diverge. First, imperfections in the land and capital markets may lead to exploitative land use practices. Second, poor decisions about land management systems may arise because of unreliable information about their long term impact on both the natural resource base and the financial viability of the farm. Third, as discussed in Section 3, attenuated property rights mean that farmers have little incentive to consider the impact of their land use decisions in the form of externalities on their neighbours.

These three alternative sources of market failure suggest different courses of action on the part of government. In this Section the sources of market failure and their implications for government intervention are reviewed. Further discussion of these issues, particularly with respect to externalities and public goods in the form of research and extension services, can be found in Mullen, Vernon and Fishpool (2000). More detailed discussion of environmental policy and policy instruments can be found in a range of environmental policy texts.

4.3.1 Landcare Technologies are Unprofitable

Perhaps the most important reason why there is low adoption of at least some landcare technologies is that farmers do not perceive them as enhancing their welfare. The term welfare has been used to avoid the narrow connotations often associated with the term profit. The welfare of a farmer encompasses changes in income flows and in the value of assets such as land and allows farmers to sacrifice some income from production for service flows from natural resources that are not related to production. However it is important to recognise that farmers, like the rest of society, are strongly influenced by self interest in making decisions about resource use. As the opportunity cost in terms of income foregone of landcare technologies increases, the adoption of such technologies is likely to decline. Landcare technologies may also require a higher level of management skills on the part of farmers and they may expose farmers to a higher degree of weather or pest related yield risk.

Putting this more clearly in a land degradation context, as for any asset, land yields a flow of services through time. Implicitly farmers are trying to allocate natural resources through time in a way such that the value of using the resource now is compared with the present value of a future stream of benefits that has to be foregone if the resource is depleted now. Note that the value of using a resource encompasses more than its value as an input in the production of food or fibre. Farmers are regularly confronted with investment decisions of this nature that involve sacrificing income in the short term, for gains in wealth in the longer term.

Consequently the rate at which the pH of the soil and the depth of top soil, for example, are rundown, are choice variables. At any point in time, the short run benefits of continuing with an exploitative technology must be compared with the longer term benefits of moving to a land care technology. At some point in time, soil attributes may have degraded to a level at which the present value of production from exploitative technologies will be less than that from landcare technologies and it will be in the interests of farmers or society to move to landcare technologies.

Hence one reason why farmers do not adopt landcare technologies is that it is not in their interest to do so at a particular point in time. Moreover in some situations, landcare technologies may never be profitable from the viewpoint of either the farmer or of society and it makes sense to exploit these resources (at least from an efficiency viewpoint) as we exploit other non-renewable resources such as coal and iron ore.

The fact that farmers seek to enhance their welfare through resource use decisions has a couple of implications for public research and extension agencies. First, research needs to focus on technologies that are relevant to farmers. Whilst publicly funded research should deliver benefits to the broader community, it must also deliver technologies that are likely to enhance the wealth of farmers if they are to be adopted without other forms of intervention by government. Marsh and Pannell (1997) pointed out that a reliance on suasive methods, such as in the Landcare program, was not enough to ensure adoption. Perhaps less obvious is an obligation on the part of research and advisory institutions to inform farmers of the on-farm implications, including profit implications, of the technologies that they are promoting.

4.3.2 Imperfect Markets

Some have argued that land degradation is associated with imperfections in the capital and land markets. With respect to capital markets, it is argued that finance is not as readily available to the farming sector as to other sectors of the economy and hence farmers are forced to adopt unsustainable technologies to meet financial constraints. However the 'Mid-term Review of the Rural Adjustment Scheme' (McColl, 1997, p42) concluded that there was little evidence of deficiencies in the debt markets for rural borrowers. Perhaps there remain concerns that banks may discount the long term benefits of investments in conservation strategies as compared to lending for more traditional technologies.

With respect to land markets, it has been argued that land values do not always reflect the degraded state of important dimensions of land quality and hence this failure encourages degradation. Again the empirical evidence to support this hypothesis is limited. A study by King and Sinden (1988) of land values in the Manilla Shire of NSW, where soil erosion was a problem, found that the land market appeared to be working satisfactorily but the hypothesis remains to be tested for less visible soil degradation problems such as structural decline and soil acidity. If this is a problem then one remedy is the provision to buyers of objective information about land quality. There seem to be few barriers to the emergence of such a method of selling.

4.3.3 Imperfect Knowledge

Clearly from either the viewpoint of society or individual farmers, decisions about land use are both extremely complex, in that they require knowledge of biological relationships over many years, and risky, in that outcomes are dependent on uncertain climatic and economic conditions. Hence one reason for the low adoption of landcare technologies may be the lack of reliable information about their long term impact on both the natural resource base and the financial viability of the farm. The impact on the attributes of soil of some land use technologies is complex, long lived, dependant on the climate and, sometimes unforeseen and not discernible without scientific testing. For example, crop rotations based on conventional tillage and nitrogenous fertilisers yield high levels of plant and animal production in the short term but through declines in soil structure and pH, yields may decline and the range of species that can be grown may narrow. The less vigorous plant cover may then be associated with soil erosion, invasion by weeds and dryland salinity. This process may take thirty years to become apparent in yield losses and reduced cropping options. The technology may have originally been adopted to counter erosion as well as to increase production and the acid soil consequences may have been unforeseen. Pannell (2001) discusses uncertainty in the context of salinity management.

It is important to provide better information to farmers and society about the long run economic and physical consequences of alternative land use strategies. There is a lot of research being undertaken into modelling such complex systems which will allow more components of decisions about landcare technologies to be explicitly valued in a benefit/cost framework[16].

The traditional rationale for the public provision of research and extension services has been that information about production technologies has the characteristics of public goods - non-rivalry in consumption and non-excludability- and hence the private sector would underinvest in research and extension from the community's viewpoint. The establishment in Australia of RDCs which levy industry members to fund research and extension, mitigates to a large degree 'free riding' associated the nonexcludable characteristic of research and extension information. The RDCs address the non-rival characteristic of information by making research information freely available to levy payers. In effect 'public' goods are transformed into 'industry' goods and the members of the industry can act collectively to provide them although still not at levels optimal for every individual. The private sector is also providing an increasing range of research and extension services to agriculture.

[16] The constraint to better information for resource use decisions appears to be lack of knowledge about important biological and hydrological processes rather than the capability to build and solve models.

5. Valuation Techniques

It is important for efficient resource use that both farmers and the community know the benefits and costs associated with alternative resource use strategies. It has been noted above that one reason farmers allow land degradation to continue is that because land degradation is often associated with gradual changes in productivity over long periods, they do not appreciate the marginal user costs of current patterns of resource use. Governments are expected to devote resources to combating land degradation because of divergences in community and private interests in the use of natural resources and hence also need to know the benefits and costs associated with alternative resource management strategies. The deficiencies of estimates of the value of agricultural production forgone as a result of various types of land degradation as a guide to the management of these problems are well known and restated below.

In this section the key issues in modelling the impact on agricultural production of land degradation and associated management strategies are reviewed. The focus of the review is on so called bioeconomic modelling using either optimisation or simulation techniques based on 'dose response' functions which account for the user cost of resource use in both temporal and spatial dimensions within a benefit-cost framework. While recognising the importance of non-use values of environmental resources in decisions about resource use at a community level, the focus in this Section is on changes in use values associated with different natural resource use strategies.

Before commencing this review, the question of efficient resource use is placed within the broader context of the theory of production economics and the valuation of environmental resources. Other approaches to valuing changes in environmental resources are briefly reviewed. These approaches include hedonic analyses of land values and the econometric estimation of a profit function where in both cases a measure of land quality is an explanatory variable. There is a sense in which these approaches are positive in nature in that they attempt an empirical analysis of how land values or profit respond to a marginal change in a measure of land degradation. This contrasts with the normative modelling approaches which this section focuses on which are based on the results of dose response experimentation and assumptions that farmers use resources in ways that maximise profits over time.

5.1 The Broader Production Economics and Environmental Valuation Literature

Techniques for valuing changes in environmental resources, such as soil quality, need to be seen within the broader context of the traditional theory of production. Quoting from Freeman (1993, p. 274):

> 'Properly specified economic models of the effects of pollution on producers make use of cost functions or production functions to link the physical effects of changes in environmental quality to changes in market prices and quantities, and ultimately to changes in consumers' and producers' surpluses. Either directly in the case of the production function

> approach, or implicitly in the case of the cost function approach, these models incorporate the whole range of possible producers' responses to changes in pollution levels (through, for example, material substitution, increased protection activities, and changes in maintenance and repair schedules)'.

The theory of production economics is thoroughly reviewed in Freeman (1993) who closely followed the approach of Just, Hueth and Schmitz (1982) in analysing the welfare implications of changes in demand or supply conditions in markets for products and inputs. This approach has been the basis of much of the large literature concerning the economic evaluation of new agricultural technologies developed in the public and private research sectors which is reviewed in Alston (1991) and Alston, Norton and Pardey (1995). However while attention has been devoted to problems of joint production and transmission of price signals through marketing chains, much of the theory and most applications have been presented in a static rather than dynamic context.

There is also a considerable literature (Freeman, 1993; Hodge 1995; Willis and Corkindale, 1995; Cropper and Oates, 1992; Thampapillai 1993) in which techniques for valuing environmental resources are described and assessed. A common classification system (Freeman, 1993) is a two way system where techniques are classified by whether they use market or hypothetical data and whether they directly value the change in resource in question or whether this value is indirectly inferred from the way in which other resources are used. Thampapillai (1993) classified techniques by whether they involved estimating welfare changes associated with the demand for an environmental resource or whether they are based on supply side concepts of opportunity cost. A review of these papers reveals that some classifications such as damage functions, opportunity cost and averting behaviour techniques do not have unique definitions. Again this literature is largely of a static nature. As for the evaluation of traditional production technologies, the theory of valuing changes in environmental resources outstrips the practice.

5.1.1 Other valuation approaches

A particular problem is that often resource service flows are not valued directly in markets and hence much of the literature is concerned with techniques to identify these implicit values. One such technique is contingent valuation. Whilst contingent valuation methods can conceptually be applied to any valuation problem, market data are more reliable than hypothetical data and hence the techniques are generally restricted to eliciting non-use values. Non-use values can only be elicited using these techniques. Since the focus of this review is on use values, there is no further discussion of contingent valuation methods, already the subject of a large literature.

Opportunities to econometrically estimate production or cost functions of the nature described by Freeman that allow an assessment of the impact on production, profit or cost of a marginal change in an environmental variable are rare. Freeeman (1993) refered to studies by Mathtec Inc. (1982) and by Mjelde, Adams, Dixon and Garcia, (1984) which examined air quality issues. The latter study dealt with crop losses in agriculture. The literature on the impact of air quality and global warming (ABARE (1997); Nordhaus (1990)) continues to grow but with the possible exception of the contribution of livestock enterprises, the changes in environmental resource quality under consideration in these studies are exogenous to agriculture and not reviewed in detail here.

An econometric approach to assessing the impact on profit of a change in various forms of land degradation in NSW can be found in Gretton and Salma (1996) who stressed the tentative nature of their findings. Their objective was to estimate the impact on profit of a marginal change in various types of land degradation. They used cross section data from 148 Statistical Local Areas (SLAs) classified into seven regions including an irrigation region[17]. The data consisted of ABS and ABARE data on production and costs for the 1992/93 year and data on land degradation from the New South Wales Conservation Service for 1987/88. They estimated as a system, a normalised quadratic restricted profit function and associated output supply (2 products) and input demand (4 variable and 2 fixed inputs) equations in which indices of dryland salinity, irrigation salinity, soil structure decline and induced soil acidity were included as fixed factors.

By structuring the model in this way, Gretton and Salma (1996) were able to estimate how farm profit responded to changes in the different types of land degradation allowing the combination of inputs and outputs to change in a way consistent with profit maximisation. Their intention was for the model to be flexible enough to allow farmers to tradeoff between current production and degradation and a conservation strategy allowing higher future production.

They found that outputs and farm profit declined if there was a deterioration in soil structure and soil acidity. In fact a unit decrease in soil structure had about twice the impact on the profit of NSW farmers as a unit decline in soil acidity. However profit increased for a deterioration in both irrigated and particularly dryland salinity. The explanation they proffered for this apparent anomaly was that whereas the effects of salinity were localised, even within a particular farm hence allowing production to expand on the rest of the farm, the effects of acidity and soil structure decline were so pervasive that there was no opportunity to tradeoff a deterioration in these conditions on part of the farm for increased productivity elsewhere on the farm. Surprisingly Gretton and Salma did not also note that the extent of externalities is likely to be much greater for the salinity issues than for soil structure decline (excepting erosion) and acidity and which could perhaps explain the positive relationship at an individual farm level between output and an externality such as salinity .

The appeal of this approach is that it provides some information on the marginal benefits to farmers[18] (the change in restricted profit) of a unit change in each measure of degradation that takes account of the ability of farmers to change input and output mixes and is not related to an arbitrary benchmark technology. Gretton and Salma did not incorporate their findings in a benefit/cost framework nor did they account for the dynamic nature of land degradation issues in a satisfactory way.

Some methodologies, such as the repair cost approach and hedonic pricing, focus on asset value. A widely quoted repair cost study is that by Woods (1983) which is reviewed below. Another approach to valuing changes in asset values associated with land degradation has

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[17] Regions were represented by a dummy variable in the econometric model.
[18] It seems that the actual unit of observation is an ABS SLA rather than a farm although this is not absolutely clear from Gretton and Salma (1996).

been hedonic price analysis. Hedonic pricing methods are used to identify the contribution to total land or property value made by characteristics of land such as the degree to which it is subject to noise and air pollution or, in an agricultural setting, to soil erosion. At least conceptually this approach can provide an estimate of the value of a marginal change in a soil characteristic such as topsoil depth and hence provide a basis for decisions about investments in soil conservation strategies.

There have been several Australian and US studies (King and Sinden, 1988; Miranowski and Hammes, 1984; Palmquist and Danielson, 1989) that have used hedonic analysis to analyse whether agricultural land values reflect the value of soil quality characteristics such as erosion. There is concern that if land prices do not accurately reflect differences in soil quality characteristics, there will an incentive for landholders to 'mine' important soil quality characteristics at the expense of future generations. These three studies focussed on areas of land where erosion was a concern and their conclusions were generally positive with respect both to the methodology and the performance of land values in reflecting quality characteristics such as soil erosion[19]. The qualifications to these findings include the fact that the nonmarket value placed on conserving topsoil by society is not captured and continuing doubts that market prices for land do not reflect less obvious forms of degradation such as soil acidity[20]. Because these studies have focussed on one aspect of land degradation, usually erosion, they provide little information about the relative importance to land values of other types of degradation.

5.2 The Cost of Land Degradation

Techniques for valuing changes in resource use in a benefit/cost framework are reviewed in detail below. However the significance of land degradation has often been assessed in terms of 'lost' agricultural production and hence an explanation of this technique and its pitfalls is presented now.

Valuing degradation in terms of production lost, sometimes referred to as the 'production equivalent of degradation' approach, involves estimating the change in production at a farm or regional level and applying a value to this change in production. Problems with this approach are discussed in Gretton and Salma (Appendix E, 1996) and in Reeves et al. (1998).

A key issue is establishing the benchmark from which production losses are measured. The benchmark is difficult to define but is generally taken to be an estimate of the production that could be achieved from the land in its most likely use were there no degradation, although some would argue that the appropriate benchmark is the preagricultural condition of land[21].

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[19] These conclusions hold for the two US studies reviewed here but no attempt has been made to review completely the many US studies and hence it is not clear whether these conclusions hold generally.

[20] An hedonic study in an area subject to acidity would seem to be a valuable contribution to knowledge.

[21] Other key assumptions in this approach are that changes in production have no effect on input or output prices and that neither prices nor production are regulated.

While such measures offer some broad indication of loss, there are a number of reasons why they are not an accurate or helpful measure of loss (Gretton and Salma, 1996,p. E3). First, the benchmark may be associated with a technology that is unsustainable. Second, such measures ignore the benefits of the increased production from the technology in its initial years. Third, the degradation may be associated contemporaneously with a productive technology, for example acid sulfate soils while causing damage to the fishing industry may have had little adverse impact on agriculture. Fourth, and perhaps most importantly, these measures provide no indication of either the cost or the practicality of reversing the degradation. In a sense, the existing level of degradation is a sunk cost and the focus should rather be on the benefits and costs of alternative resource use strategies to be implemented from now.

To this point I have drawn attention to the inherent difficulties associated with the concept of a benchmark. Additionally there are practical measurement problems (that are not unique to the benchmarking approach). There are two issues here. The first concerns the statistical precision with which the relationship between a measure of degradation, such as soil acidity, and production is estimated. The range is from subjective estimates of yield loss through simple linear relationships to statistically estimated models that allow diminishing marginal productivity and a wide range of substitution possibilities. In the literature these approaches are referred to as damage function or dose response techniques which approach a fully specified flexible production function as more substitution possibilities are allowed.

The second concerns the accuracy with which the extent of degradation is estimated. This may range from subjective estimates by local experts to the use of geographical information system (GIS) data as in the study of soil erosion in the Lachlan Valley by Mallawaarachchi et al. (1994) and Mallawaarachchi and Young (1994).

Many of the observations based on rudimentary production economics made by McInerney (1996) concerning the cost of animal disease seem applicable to land degradation. He pointed out that the simple price times quantity approach overestimates the cost of disease because producers respond to disease by changing both inputs and outputs hence moving to a different point on the production surface. This is related to the choice of a benchmark. It is not a one dimensional choice because not only does it involve choosing the base level of production but also the level of inputs or the farming system. From this it also follows that strategies to control degradation involve changes to the farming system as well as to the land resource. McInerney used the concept of a ‘disease loss – expenditure’ function to demonstrate how the efficient management of disease involves selecting that combination of disease loss and disease control expenditure that minimises the total costs of disease.

Related to this value of lost production approach are methodologies such as the repair cost approach which focus on asset value[22]. A widely quoted repair cost study is that by Woods (1983) which is reviewed in Reeves et al. (1998) and Gretton and Salma (1996). The Woods study is the most recent attempt at an assessment of the extent of land degradation at a national level. The approach taken was to estimate the cost of returning land to a near preagricultural state. This cost was estimated to be $2.4b in 1997 dollars for Australia

[22] Another approach to valuing changes in asset values associated with land degradation has been hedonic price analysis. Hedonic pricing methods are used to identify the contribution to total land or property value made by characteristics of land such as the extent of ASS. This approach becomes less valid as externalities such as acid discharge become more important.

(Reeves et al. 1998, p.65). The study estimated that about 16 percent of agricultural land required some treatment, with water erosion in NSW being the largest single contributor. Apparently Woods did not consider soil acidification and structural decline. It would probably not be economic to repair much of this degradation under the present technology and terms of trade facing agriculture. This repair cost approach to valuing land degradation suffers from the same drawbacks as the lost production approach.

There have been few comprehensive national studies that have applied these techniques. Gretton and Salma (Appendix E,1996) referred to a DEST (1995) estimate of the annual cost to Australia of $1.15 billion. The technique has been used more frequently in regional studies or studies of particular types of degradation. Some of these studies are reviewed in Gretton and Salma (1996), Reeves et al. (1998) and LWRRDC (1995, 1998) and in the next section. The NSW EPA(1993) for example estimated that land degradation cost NSW $250m annually with soil structure loss and then soil acidity accounting for much of this[23].

Another example of this approach is an unpublished 1991 study by DPIE which is reviewed in Reeves et al. (1998). A feature of this study was its narrowing of focus to land at risk of degradation rather than land already degraded. The potential lost production from land ‘at risk’ of degradation was estimated to be $1b in 1997 dollars (Reeves et al. 1998, p. 66) with the losses from soil structure decline being much larger than the next most important source of potential degradation, acidification. Reeves et al. did not make clear what the benchmark was.

In the preceding paragraphs we have tried to demonstrate the nebulous nature of this concept of the cost of land degradation issue. It would seem that physical measures are likely to be as valuable as indicators of the significance of the problem.

5.3 Bioeconomic Modelling of Land Use Management Strategies

The objective here is to review some of the key modelling issues in the use of bioeconomic models to compare the implications of alternative land use management strategies for efficient resource use from the viewpoint of farmers and the community. The basis of any evaluation of a strategy to control land degradation is a relationship between land degradation and plant or animal production.

There have been a number of relatively unsophisticated benefit/cost analyses of particular soil health issues where production equivalent measures of degradation associated with a small number of highly constrained management options are the basis of estimated benefits. Gretton and Salma (Appendix E,1996) reviewed studies of irrigation salinity in the Murray River Basin (Murray-Darling Basin Ministerial Council, 1987), dryland salinity in the Campaspe Catchment of Victoria (Oram and Dumsday, 1994) and in the Neridup Catchment, Western Australia (Campbell, 1994) and soil acidity in eight priority areas in Australia (AACM, 1995). Van Bueren and Pannell (1999) reviewed regional studies of dryland salinity, many of which had these characteristics.

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[23] The estimated annual losses in the EPA and DEST studies amounted to five percent of the local value of agricultural production in NSW and Australia in 1992-93 and 1994-95 respectively.

As noted above, estimates of the cost of degradation are often based on the value of production foregone or the cost of restoring the land, relative to a benchmark in both cases. Generally explicit models of the relationship between degradation and production are not used. Other problems with studies of this nature are that they only examine a small number of alternatives none of which may be optimal; they are highly constrained allowing farmers little scope to alter product and input mixes in response to degradation; they are generally deterministic with respect to climatic conditions and there is limited scope to account for the dynamic relationship between soil health and agricultural production.

As concern about degradation issues has increased, there is a growing interest in applying techniques which explicitly model relationships between production and degradation in the form of damage or loss functions. Damage functions range in complexity from simple linear relationships such as is embodied in value of forgone production estimates, to simultaneous equation models of production and/or cost systems.

Modelling involves choices about how best to abstract from reality. These choices are influenced first by the purpose of the model and second by the resources available in terms of knowledge of the biological and economic relationships and also time and computer resources. As in any other area of resource use some judgement is required about the value of the new information generated by the model and its cost.

Some of the key choices in modelling land degradation issues reviewed below include:

- the balance between the economic and biological components of the model;
- the treatment of the dynamic relationship between a soil health issue, such as acidity, and production;
- the ability of farmers to change input and output mixes in response to changes in land quality;
- the treatment of uncertainty and opportunities for tactical decision making; and
- the treatment of externalities.

5.3.1 *Bioeconomic modelling*

The term bioeconomic modelling seems to have come into vogue as a means of highlighting the value of an appropriate balance between the economic and biological approaches to modelling the economic consequences of complex biological processes. A criticism of economic models has been that they attempt to represent complex biological relationships in reduced form, often as single exogenous coefficients. The representativeness and adaptability of such models is highly questionable.

On the other hand there are highly specified biological models of complex systems which model how a crop might respond to different soil, moisture and management conditions. Consequently there has been increasing interest in using simulation techniques as decision aids in pasture, animal and cropping systems. Good examples of this include the GRAZFEED group of models from CSIRO (Donnelly et al., 1997) and the APSIM cropping model from the APSRU group at Toowoomba (McCown et al., 1996). Often such models have only rudimentary economic components in that there are constraints on altering resource use in response to changing economic incentives and changes in profit through time are improperly measured. The term bio-economic modelling is now being used to describe modelling that has significant economic and biological components.

5.3.2 Optimisation versus simulation

An important feature of bioeconomic models is whether they are simulation or optimisation models. Simulation models (with an economic component) estimate an objective function such as net farm income or total farm gross margin for specified management regimes. Different management regimes can be compared by comparing the change in objective function and by searching over a large number of options, the best management regime is approached. The simplest simulation models are spreadsheet based budgeting analyses.

Linear programming models incorporated within a benefit/cost framework have two advantages over these budgeting and simulation approaches. First they allow a much wider range of response by farmers in their choice of outputs and inputs than the limited number of alternatives presented in budgeting studies. Second, LP is a powerful optimising technique in that it selects the combination of enterprises that will maximise profits (or minimise costs) from a specified set of enterprises subject to specified resource constraints. An added advantage of optimisation models is that they provide ‘shadow prices’ indicating the change in profit were additional units of a fixed resource made available or were an additional unit of an enterprise not in the optimal plan forced into the plan.

In its simplest form LP optimises over one period and uncertainty is either ignored or farmers are assumed to be risk neutral. The objective function is either profit maximisation or cost minimisation subject only to constraints on resource use. A well know LP model from Australian agriculture is the MIDAS model from Western Australia and its eastern derivatives, PRISM. Such models have been used to examine the role of particular enterprises in rotations (Faour, Brennan, Scott and Armstrong, 1998). Experience with MIDAS is reviewed in Pannell (1996).

Optimisation requires greater computing resources and hence comes at the expense of either a reduced range of management options or the complexity of the biological relationships. There is growing interest in using biological simulation models of pasture growth, such as GRASGRO, and of cropping systems, such as APSIM to provide inputs to the LP optimising models.

5.3.3 Dynamic versus Static Modelling

Many LP models can be classified as being static and deterministic which are likely to be serious deficiencies when dealing with natural resource problems. Static models have no true time dimension. Even studies of crop rotations represent the sequence of crops as a share of land area rather than explicitly as a time sequence. Hence such models do not account for the cost of lost production in future years from diminishing the stock of a soil attribute such as acidity in the current year. For models to be dynamic there must be a feedback mechanism whereby the stock of a natural resource asset in one period influences enterprise choice and production in that period which in turn changes the stock of the asset in the next period and the decision maker is optimising some measure of wealth over the entire observation period using discounting principles, not just in particular years. A good discussion of these issues (and of dynamic programming) can be found in Kennedy (1986, 1988).

The approaches to this problem have been to extend LP to a multi-period framework or to use optimal control techniques such as dynamic programming (DP). A good example of the use of multi-period LP is Greiner's (1997, 1998) study of salinity on the Liverpool Plains of NSW. Miranowski (1984) used a multi-period LP model to study how farmers in the US cornbelt might change tillage and cropping rotations in response to soil erosion to maximise variable profits over a fifty year period.

LP is not a particularly efficient user of computer resources and this becomes an important constraint when long time periods have to be considered. There is growing interest in the use of optimal control techniques. In a study of soil erosion in the US Walker (1982) used DP techniques to identify when a farmer on a particular soil type should switch to conservation farming technologies. Jones and Medd (2000 and 1997) used DP techniques to compare alternative weed control strategies. Neither of these studies treated climate or price as stochastic variables. The Walker study constrained the farmer to choice from only two strategies.

Optimal control models are based on the mathematical programming techniques of microeconomics with a time dimension but unlike DP cannot accommodate stochastic events. DP has the further attraction of not requiring smooth, differentiable functions. Examples of the use of optimal control modelling include McConnell's (1983) US study of soil erosion and the study of the sustainability of pasture management systems by Farquharson and Mullen (1998).

5.3.4 Stochastic versus Deterministic Modelling

Another characteristic of farming systems that influences the modelling framework is the importance of stochastic rainfall events to farmers' decisions about resource use. There seems to be a growing consensus (Cregan and Scott, 1998) that water use efficiency has important implications for land degradation issues such as acidity and salinity and that responding tactically to stochastic rainfall events may help in ameliorating land degradation.

This is not the place to exhaustively review the techniques of decision analysis (Anderson, Dillon and Hardaker (1977)). However there are two broad issues to be considered. The first concerns how projects that differ in their income and risk profiles are ranked. The second concerns how decision making under uncertainty is modelled. These concerns are reviewed in turn.

It is likely that both individuals and society are not indifferent to risk but make tradeoffs between expected income and other characteristics of the distribution of income such as variance, skewness and kurtosis. Anderson, Dillon and Hardaker (1977, Chapter 4) demonstrated how the utility of a single attribute such as income could be expressed in terms of the moments of the probability distribution of income. Little is known about the risk preference of Australian producers but studies by Officer and Halter (1968), and Bardsley and Harris (1987) found evidence that woolgrowers are risk averse, although not to a large degree according to Bond and Wonder (1980). Often society is assumed to be neutral to risk but this may not be the case for investments with the potential to significantly change environmental flows for many years into the future.

Many studies are deterministic in nature with perhaps some use of sensitivity analysis to examine the impact of key assumptions such as interest rates, product prices and technical parameters. To more explicitly model risk or the stochastic nature of investments, the values of prices and yields can be drawn from historical or subjectively formed distributions of these parameters using software such as @RISK. This approach allows the use of expected profit, for example, as the decision criterion. LP can be combined with monte carlo simulation, within @RISK for example, to examine the implications of uncertainty about yields and prices in a non-embedded risk setting. The outcome of this approach is a distribution for the value of the objective function reflecting the uncertainty about prices and yields.

If the decision maker is risk neutral or unconcerned about the variance of income, then projects can be ranked according to their expected income. If the decision maker is concerned about the mean and variance of income but not the higher moments of the probability distribution, his utility function can be expressed as a quadratic function and risky decisions can be analysed within what is known as the mean-variance (E-V) model (Anderson, Dillon and Hardaker (1977, p.95)). This framework is also exact if the distribution of income from projects is normal and hence can be completely described by their means and variances. As for other economic problems, this approach is based on equating the marginal rate of substitution between income and its variance in production and consumption. On the production side this requires knowledge of the risk efficient set of investment alternatives or the identification of those alternatives that maximise income for a given variance or minimise variance for a given level of income. On the consumption side the knowledge of how utility responds to changes in income and variance is required. Stochastic dominance tests can be applied when this utility function is unknown (Meyer 1977a and b, McCarl, 1990) although this procedure is not powerful enough to distinguish between investments that have similar income characteristics. Recent Australian applications include those by Jones, Wall and Marshall (1996) and Patton and Mullen (1999).

Turning to the second concern about how uncertainty is modelled, Hardaker, Pandey and Patten (1991) made a distinction between embedded and non-embedded risk where embedded risk is associated with the uncertain events whose outcomes become known during a sequential decision making process. Uncertain events of this nature may change the available soil moisture, for example, and farmers can make tactical responses to these changes. If risk is modelled in a non-embedded way, decision makers make strategic decisions at the start of the period and then await the risky outcome with no opportunity to make tactical responses as uncertain events unfold. Hardaker et al. (1991) classify embedded risk models which allow tactical response through time, as stochastic programming models and non-embedded models as risk programming models.

Risk modelling has been reviewed by Pannell, Malcolm and Kingwell (1995). An important conclusion they drew was that the returns to modelling tactical response to large changes in price and weather conditions are likely to be much greater than to incorporating risk aversion. Kingwell (1994) concluded that ignoring tactical response resulted in farmers' response to price being overestimated as well as underestimating profit.

An important characteristic of stochastic modelling approaches is that the observation period is broken up into stages at which farmers can make tactical responses to climate and market risks as they unfold. A second characteristic is that models of this nature can generally be laid out in a decision tree format and often solved recursively, that is the best outcome in the final period is selected and then the best outcome in the previous period along that decision path is identified. This recursive procedure is followed until the initial period or the first decision node is reached. Note that a variety of techniques can be applied in the analysis of these stages. At one extreme are simple budgeting approaches but mathematical programming and simulation models have also been used.

A variety of approaches have been used that fall within this general category of tactical decision modelling. There is growing interest in this area as the increase in computer power makes possible more realistic representations of the decision problems and more realistic modelling of production and transformation relationships. Despite this increase in computing power and the development of more sophisticated methodologies, the size of models grows very quickly and researchers are forced to constrain their problem to some degree or to maintain certain hypotheses to conserve degrees of freedom, in the language of econometrics.

Hence, while these approaches to modelling tactical decision making have some common characteristics, they also differ in some key areas such as the treatment of risk, the range of substitution possibilities allowed the decision maker, and whether they are optimisation or simulation models. In the discussion below, some studies are referred to that highlight these differences but no attempt has been made to thoroughly review the literature in this area.

In some of the earlier work in this area, there was an emphasis on risk analysis. The outcomes of uncertain events were replaced by the decision maker's certainty equivalents working back through a decision tree or an explicit utility function was used. Examples of this approach can be found in Makeham, Halter and Dillon (1968). While the examples in this book are far removed from complex land degradation issues, the decision tree concept is a valuable one in clearly laying out the sequential nature of the problem confronting the decision maker. An example of the use of decision trees in a situation of tactical cropping related to soil moisture and salinity can be found in Burt and Stauber (1989). In many problems there are strong feed back effects and hence the decision trees become more circular in nature as in Jones and Medd (2000 and 1997). Once feedback is introduced we are in the realm of dynamic problems. A possible research strategy would be to apply budgeting techniques to highly constrained paths through these decision trees and then to relax some of these constraints by incorporating damage functions and stochastic weather and price events.

LP modelling can be extended to account for stochastic events such as rainfall. Rainfall can be used to define a range of seasons with known probabilities of occurrence and for each season, a module of enterprises from which LP selects to maximise its objective function. The risk modelling framework may be either non-embedded (Marshall, Jones and Wall (1996)) or embedded as in discrete stochastic programming as reviewed in Jones (1997) and applied in Greiner (1997).

Simulation models are not strictly speaking optimisation models but can be used to generate a distribution of payoffs which provides a good indication of the relative profitability of alternative strategies. A good example of this approach is the Burt and Stauber salinity study (1989) which built on research by Zusman and Amiad (1965). The objective was to identify a decision rule about the level of soil moisture on which to sow crops. The research strategy they followed was to vary the decision rule between the extremes that resulted in either fallow or continuous cropping. Other features of this study were that soil moisture after the first crop was a stochastic variable and the range of alternatives could be broadened to consider more than one paddock and more than one crop in a year.

5.3.5 *Dynamic Stochastic Models*

As already noted land degradation issues are often dynamic and stochastic in nature and hence ideally require the application of dynamic stochastic models. At present optimal control techniques cannot incorporate stochasticity. There is growing interest in NSW Agriculture and the Department of Natural Resources and the Environment in Victoria in the application of dynamic programming to land degradation issues such as salinity and acidity. The other technique is multi-period LP modelling extended to include both dynamic and stochastic capabilities. This is the approach taken by Greiner (1998) in her study of salinity in the Liverpool Plains of NSW.

5.3.6 Issues in modelling biological systems when externalities arise

Accounting for externalities requires an even greater understanding of biological systems. The impact of externalities in an agricultural context come in two forms. First there is the loss of agricultural production experienced by ‘neighbours’. As already noted, modelling externalities of this nature requires knowledge of how agricultural technologies at one site have an impact on soil qualities at another site (which may not be on the same farm) and hence on agricultural production at that site. Cregan and Scott (1998) concluded that ‘The water cycle is a unifying concept that links many of the significant land degradation/agricultural productivity problems..’. Hence a better understanding of the hydrogeological processes in catchments would allow better modelling of many externality problems.

Many externalities are confined within water catchment areas. This has implications for modelling and for policy response. From a modelling point of view it raises the possibility of developing a model of the catchment rather than of a representative farm. Greiner (1998, 1997) in a study of salinity on the Liverpool Plains, used both farm and regional models to capture on and off-site effects from hydrogeological processes in this catchment.

The role of externalities in identifying a potential role for government was discussed in Section 3. In that section also an appropriate empirical strategy, applied by Quiggin (1986), to estimate the significance of an externality was briefly described. While the these modelling skills have been developed and applied in farm level modelling exercises, they have rarely been extended to this public policy arena and consequently decisions about government intervention to correct externalities are based on subjective estimates of the extent of these externalities. Often the significance of externalities is judged on the basis of ‘lost’ agricultural production, which on top of the problems described above, has limited relevance for public policy. Modelling land degradation issues such as salinity which extend over large catchments, is an expensive business but has the potential to assist government in directing its limited resources to areas of highest payoff. A contrary view about the value of such large scale modelling can be found in Bromley (2000b).

Another deficiency apparent in the literature is that generally degradation issues are assessed separately even though particular areas of land may be subject to several forms of degradation. It may well be that these multiple types of degradation are related. Soil acidity may lead to dryland salinity and/or erosion, for example. In such situations it would seem to be desirable to model the entire system not just one element.

Second there are the externalities in the form of reduced water and air quality, health effects and amenity values etc. that are imposed on the rest of the community. In most of the agricultural studies reviewed above these externalities are acknowledged qualitatively but are seldom modelled quantitatively. The techniques for estimating these external effects are reviewed in the literature referred to in Section 5.1 but as already noted, little attention is devoted to these issues in this review.

6. Review of Land Quality Issues

6.1 Introduction

Having reviewed arguments relating to the more general question of the sustainability of agriculture, the focus now narrows to examine the use of the land resource in Australian agriculture. There is widespread concern that the land resource is being degraded at a rate that is not in the interests of either farmers or the community. Land degradation is a form of resource use that for the sake of current production, imposes future costs, not only in the form of lost agricultural production but also in the form of undesirable environmental outcomes.

Up to this point, we have talked about land degradation in general terms. However land (or soil) quality has many dimensions of significance, both to agriculture and the environment. In this section attention will focus on a selection of soil attributes critical to agriculture, including soil acidity, dryland salinity, soil depth, and soil structure. They have been selected because they are associated with common agricultural practice and with significant losses both in terms of agricultural production and environmental outcomes. The review of soil structure is much less detailed than for the other types of degradation, largely because little seems to be known about it despite its pervasiveness.

These specific types of land degradation will be reviewed with respect to:

- The biological processes which cause them;
- Potential off-site impacts and externalities;
- The extent of the problem;
- Previous economic analyses;
- Management strategies for landholders and the community.

From the perspective of the community, a desirable outcome from this review would have been an assessment for each type of land degradation of the extent of externalities imposed on the community as an indication of whether government intervention was potentially efficient. In the section above on ‘Externalities’, a conceptual approach to measuring the extent of an externality was identified. Few analyses of this nature have been undertaken in Australia with the work of Quiggin (1991, 1986), Barton (1992) and Greiner (1998) being exceptions. The present review falls well short of this desirable outcome.

For each land degradation type, the potential for externalities is discussed and the significance of degradation is indicated, in most cases, by estimates of foregone production or by some physical measure. Important empirical studies that relate the costs of degradation to management strategies intended to alleviate degradation are reviewed with respect to their methodology and findings.

In general, the focus of past studies has been on agricultural impacts of land degradation and this focus is continued here. However, some estimates put the off-site costs or externalities of soil degradation as higher than the on-site costs (Policy Development Planning, 1992; Upstill and Yapp, 1987).

Issues not examined include the contamination of soil by heavy metals and pesticides, the role of soil biota, the loss of native vegetation in general and in riparian areas, invasion by weeds and animal pests and loss of agricultural land to urban uses. Soil contamination is reviewed in Reeves, Breckwoldt and Chartes (1998) who pointed to fertilisers as being important sources of contamination. The loss of native vegetation is reviewed in LWRRDC (1995) which pointed to the loss of habitat for flora and fauna as posing a threat to biodiversity as well as causing losses in agricultural production through induced salinity and erosion.

6.1.1 Catchment hydrology and externalities

Cregan and Scott (1998) concluded that ‘The water cycle is a unifying concept which links many of the significant land degradation/agricultural productivity problems…’. This echoes the views of other scientists, Passioura and Ridley (1998), for example. Gretton and Salma (1996, p.C12) noted correlation between the Statistical Local Areas (SLAs) in NSW that experience the most severe soil acidity with those that experience the most severe dryland salinity.

Land degradation seems often to be associated with the inability of farming systems based on annual crops and pastures to use rainfall as it occurs leading to accessions to groundwater at recharge sites which re-emerge in a more saline form at discharge sites (presuming that the soil though which the water passes is salt-laden). Pastures and crops on acid soils use even less water because they are less productive. Whether the discharge sites are on- or off-farm depends on the hydrology of the catchment. Where groundwater aquifers are largely of a ‘local’ nature, recharge and discharge sites are often on the same farm and hence management of land degradation is an issue for the landholder who weighs up over time, the benefits of higher production from recharge areas against lost production in discharge areas. Where groundwater aquifers are largely of a regional nature, externalities have the potential to be far more significant and the management of land degradation becomes a community concern.

Pannell et al. (2000) argued that the hydrogeology of many catchments in Western Australia was such that there was little prospect with present technologies of efficient intervention by government to correct externalities. However landholders did have some ability to control the watertable under their properties if that proved to be in their interests[24]. The consensus in NSW is that the hydrogeology is such that recharge and discharge sites are widely separated and hence that externalities are significant. However little is known of the hydrogeology of many catchments and sub-catchments. Whether government can intervene efficiently is another question.

6.1.2 Some Key References

There are a number of valuable references concerning land degradation issues. The most recent is a review of soil health issues commissioned by the Land and Water Resources

[24] They argued that some landholders were not taking appropriate action because of the prevailing view that salinity was an externality.

Research and Development Corporation (LWRRDC) from the Centre for International Economics (Reeves, Breckwoldt and Chartres, 1998). The report has sections on the extent and severity of various soil health issues and on possible forms of government intervention but of perhaps greatest interest is the section on defining and estimating the costs of degradation. Another valuable resource from LWRRDC is a publication titled ‘Data sheets on natural resource issues’ first published in 1995 which also provides information on the extent and severity of degradation and ongoing research. This publication was revised in 1998 and again in 1999 but pays less attention to soil health issues that the first edition.

A key reference for both soil and water quality issues is the ENVALUE database of studies of environmental valuation assembled by the NSW Environment Protection Agency (EPA). This database was first released in 1995 and a revised edition is now available on the Internet. The computerised database comprises a listing and summary at three levels of detail of a wide range of environmental valuation studies. The accompanying handbook discusses environmental valuation principles and methods and provides background information about broad categories of environmental issues such as land and water quality issues. The extent to which valuation estimates can be transferred between sites is also discussed both in general and for specific studies or issues. However ENVALUE made little attempt to discuss the relative significance of key environmental issues to Australia either in physical or economic terms.

Other key references are the reports and working papers from the Industry Commission and the Productivity Commission Inquiry into Ecologically Sustainable Land Management including Gretton and Salma (1996), Industry Commission (1997), and Productivity Commission (1997). A useful reference providing explanations of soil health issues that people like economists can understand is a report by Lines-Kelly (1995). Extension material explaining the nature and management of soil health issues is also available from state Departments of Agriculture, for example, the AGFACT series from NSW Agriculture.

None of these reference sources consistently report for each type of degradation, the area affected and the value of lost production, probably because of data limitations. Further, the references are sometimes difficult to compare because data have been collected in different years using different classification rules, methodologies and values. These issues are discussed in Reeves et al. (1998).

6.2 Salinity

As already noted in Section 4, salinity is currently regarded as Australia’s most significant natural resource management issue. Important references are the Salinity Audit (Murray-Darling Ministerial Council, 1999) for a detailed assessment of the extent of the problem, the Basin Salinity Management Strategy 2001-2015 for the current policy response (Murray-Darling Ministerial Council, 2000) and Pannell (2001) for a critical review of arguments concerning the externalities associated with salinity and the range of solutions proposed to manage salinity.

6.2.1 The Nature of Salinity

In Australia, salinity problems are associated with both irrigated and dryland agriculture. The problems associated with irrigated agriculture have been recognised for many years and have been the subject of extensive research, extension and policy initiatives as evidenced in NSW for example, by the Land and Water Management Planning process. However it is only since the late 1990s, particularly as a result of the Salinity Audit of 1999 commissioned by the Murray –Darling Basin Ministerial Council, that the importance of dryland agriculture to salinity, particularly in surface water systems, has become more widely appreciated. Prior to this the focus seems to have been on the salinisation of some land in discharge areas. The awareness of dryland salinity has been greater in Western Australia.

According to the LWRRDC (p.7,1999): 'Salinity occurs when changes in land use ….. causes changes in water balance resulting in rising watertables and salinisation'. Rising watertables mobilise salt which is normally deep enough in the soil profile not to affect plant growth. Agricultural practices have contributed to increased accessions to the watertable through changes in vegetation such as the replacement of trees and perennial grasses by annual pastures and crops which use less water where it falls. Walker et al. (1999) have suggested that leakage under farming systems is greatest in high rainfall regions (> 600mm). Inefficient irrigation practices also lead to rising watertables.

6.2.2 Dryland salinity and externalities

These increased accessions have two broad classes of consequences with the potential to impose externalities on neighbours and the community. First, land that is hydrologically related to the recharge area may suffer production losses and a narrowing choice of salt tolerant species as the watertable brings salts into the root zone of plants. The second class comprises all land or surface water systems affected by the increased salt in the stream flow. Increased salinity in surface water reduces the quality of drinking water and damages infrastructure such as water delivery systems, roads and buildings. It increases the frequency of irrigations required to leach the soil profile of salt delivered during irrigation and hence leads to accessions to the watertable. Additionally it threatens the biodiversity of water systems such as the Macquarie Marshes by favouring salt tolerant plant and animal species.

One of the most contentious issues in the debate about the management of salinity concerns the extent to which particular types of land use at specific locations in the landscape, change the level of watertables and the mobilisation of salt throughout a catchment, as well as locally, and whether these externalities are sufficiently large, given the long time frame it takes for changes in land use to result in changes in salt mobilisation, for the costly investments in some of the land use changes being suggested to ever be recovered. The best discussion of the issues can be found in Pannell (2001).

The answer is largely of an empirical nature requiring on a more detailed knowledge of the hydrogeology of catchments than is currently available. The hydrology of catchments determines the efficacy, at least in a biological sense, of alternative management strategies or land uses. This issue is pursued further below.

6.2.3 The Extent of Dryland Salinity

According to the LWRRDC (1998):

> 'Dryland salinity currently affects almost 2.5 million hectares in Australia and a further 8 million hectares could be affected over the next 30 years if current land use practices are maintained….. The latest estimate of the cost of dryland salinity is $270m per annum, comprising $170m in lost agricultural production, $100m in damage to rural and township infrastructure and $40m in reduced environmental asset values. (pps.14-15)'.

Turning to the Murray-Darling Basin (MDB), it was estimated that by 1987 96,000 hectares of the irrigated land in the Murray-Darling Basin (Murray-Darling Basin Ministerial Council, 1999) were salt-affected and 560,000 hectares had water tables within 2 metres of the surface. By 1998 about 5 per cent of the catchment had shallow water tables, with over 15 million hectares of rising ground water. By 2010 all the irrigation land within the southern Basin will have water tables within 2 metres of the surface unless there are further changes in practice.

The most recent Salinity Audit (1999) concluded that reliable estimates of the extent of dryland salinity could not be made because of a lack of critical data about changes in watertables and surface contour data, particularly in NSW. The Audit noted that in 1996 an estimated 300,000 ha were salt-affected in the Basin and that estimates of the salt-affected area once a new hydrologic balance was reached, ranged from 6 – 9 million hectares.

The Audit noted that currently about 2m ha in the wheatbelt of Western Australia is salt affected (p. 27). Pannell (2001) pointed out that Western Australia currently accounted for about 80 per cent of salt affected land in Australia and even with the rate of salt mobilisaiton in the MDB, Western Australia was still likely to account for half of dryland salinity in 2050.

The Audit however, did spell out its expectations about the future course of water quality through the MDB in the absence of attempts to halt the mobilisation of salt, particularly from dryland agriculture. The Audit pointed out that the MDB Salinity and Drainage Strategy of 1989 had had some success in reducing salt exported from irrigated agriculture. Average salinity in the Lower Murray at Morgan had declined from an average of 721 EC and exceeding 800 EC forty two percent of the time in 1989, to an average of 569 EC exceeding 800 EC only eight percent of the time in 1998. However largely because of salt export from dryland areas, salinity is expected to increase to 670EC, 790EC and 900EC in 2020, 2050 and 2100[25].

Projected salinity levels at other locations in the MDB were higher than unexpected and concerning. According to the Audit (p. vi): 'The Macquarie, Namoi and Bogan Rivers will exceed the 800 EC threshold within 20 years….The Lachlan and Castlereagh Rivers will exceed 800 EC within 50 years. The Condamine – Balonne, Warrego and Border Rivers will exceed 800 EC before 2020. The Avoca and Loddon Rivers already exceed 800 EC on average'.

[25] 800EC is the threshold defined by the World Health Organisation for drinking water, 1500EC is a threshold at which it is risky to irrigate many crop species and where adverse biological impacts occur in waterways. Water starts to taste salty at 1700 EC and at 5000EC it is regarded as saline rather than fresh.

6.2.4 Economic Analyses of Dryland Salinity

The Salinity Audit (1999) referred to studies commissioned by the MDBC which put the cost of a one unit change in EC at Morgan at between $93,000 and $142,000 per year. It noted estimates of the total economic impact of salinity to be $46m per year. Gretton and Salma (1996) reviewed benefit cost analyses of dryland salinity in the Campaspe (Oram and Dumsday, 1994) and Neridup (Campbell 1994) catchments. Both studies found that while some rotations could be identified that were profitable to producers and lowered the rate of spread of salinity, there were no rotations that were profitable and reduced the rate of spread to zero. Other regional analyses of salinity were also reviewed by van Bueren and Pannell (1999). The Audit referred to the inadequacies of such studies and one of its recommendations concerned the need for better economic analyses of strategies to manage salinity.

Important economic analyses of salinity in Australia are those by Quiggin (1991, 1986), Greiner (1997, 1998) and Barton (1992). Quiggin's study concerned water quality in the Murray River rather than land degradation per se. However, as noted above, the study is important for defining a conceptual approach to the assessment of the extent of externalities or the cost of degradation. Quiggin divided the catchment into six sections and compared the change in wealth from the six farms when they operated with no concern by upstream for downstream users (open access) and when the asset, water quality, was managed to maximise the income from the six farms operated jointly (common property). Quiggin found that when the river was managed as common property, water use and salinity decreased and the value of the asset increased from $65m to $84m. The land use change suggested by a common property approach involved a shift from irrigating low- to high-value crops. Barton (1992) followed the spirit of Quiggin's work in evaluating some hypothetical salinity scenarios and management strategies in Western Australia

Greiner studied dryland salinity on the Liverpool Plains of NSW using four farm models representative of the major production and hydrological systems in the area. The objective was to optimise income using multi-period linear programming where production was influenced by accessions to the watertable throughout the catchment through its impact on the depth to groundwater on farm. Greiner (1997), in running the farms independently, found that while profitable on-farm strategies could be devised to control on-farm recharge, salinisation continued to increase on those farms where the watertable was largely influenced by water use elsewhere in the catchment. Greiner (1998) also ran her model as a catchment model optimising the income of the catchment rather than the individual farms although she did not pursue Quiggin's appealing strategy and hence it is difficult to judge the significance of the externalities from her work. She did however observe that under her preferred optimisation scenario where rainfall was stochastic, the amount of land affected by salinity would stabilise at about 62,000 ha, about double the area under threat of salinisation at the time of her study. Preferred land use strategies included less use of long fallows, more use of perennials such as lucerne and more use of salt tolerant species such as saltbush, in discharge areas. Tree planting did not feature strongly but the commercial opportunities for forestry that are now emerging, in the form of carbon credits, for example, were not apparent when Greiner conducted her analyses.

6.2.5 Management strategies

The management of salinity revolves around the use of deep rooted perennials to reduce accessions to the watertable and to lower the watertable and, in addition, in salt affected areas, the use of agricultural enterprises that tolerate salt. Engineering solutions, such as drainage systems, may also be applicable in some areas.

Some (for example, Walker et al. (1999)) have argued that large scale revegetation, particularly in the high rainfall areas bordering the MDB, is the only solution. However, Mullen, Helyar and Pagan (2000) argued that a simple blanket recommendation that does not account for soil type effects and the amount of salt in the pathway of the deep drainage, may lead to control measures that are not economically optimal. In addition to other biological and hydrological uncertainties about the effectiveness of such a strategy, a practical means of implementing such a strategy, expected to be unprofitable for most farmers, is not obvious.

It would seem that the practical management of salinity is site specific. A general discussion of the factors of influence can be found in Pannell (2001). In some high rainfall areas it may be profitable for farmers to consider forestry options. There are extensive areas of private forestry in Western Australia. Further impetus to forestry development depends in part on whether forestry is recognised under the Kyoto Protocol covering greenhouse emissions as a means of earning carbon credits.

In other areas of the MDB, farmers may find it profitable, or at least not very costly in terms of foregone income, to make greater use of perennial pastures. It is uncertain how effective perennial pastures can be in reducing groundwater accessions across the Basin, either because of too much water in the high rainfall areas, or because of the difficulty of establishing and maintaining a productive perennial system in the low rainfall areas. In many areas annual cropping systems will remain far more profitable than perennial pasture systems for livestock enterprises. Pannell (2001) noted the need for R&D to develop more profitable perennial pasture systems and also to develop profitable agricultural systems for salt affected land and those are two of the objectives for the recently funded Cooperative Research Centre (CRC) for Plant-Based Management of Dryland Salinity

As already noted there are potential externalities associated with salinity leading to a divergence in the interests of farmers and the community. This divergence in interests or potential market failure suggests an important role for government in managing salinity in addition to its long accustomed role of providing research and extension services to counter a perceived failure by industry to invest at an appropriate level of these services. To date there has been a reliance of suasive tools under the Landcare Program to increase the awareness of farmers and the community about salinity and how it might be managed. Pannell (2001) reviewed the weaknesses of relying on such an approach when the profitability to farmers of recommended strategies was uncertain or non-existent. However Randall (1999a) and Marshall (1999) warned that the role of local communities in dealing with these ‘isolation’ issues or externalities should not be overlooked.

There seem to be few practical policy tools to deal with a pervasive non-point externality such as salinity. At present there is no cost effective technology to measure the export of salt and/or the accessions to groundwater from particular farms which would allow an assignment of property rights necessary for regulatory or market based approaches directly related to the externality. Instead such approaches would presently have to focus on less direct targets such as land use, and hence, are likely to be both less efficient and less equitable. There is little evidence that the community is prepared to fund the level of financial incentives required by farmers to adopt the level of revegetation proposed by some.

Pannell (2001) argued that publicly financed revegetation implemented indiscriminately across the landscape would be inefficient in the sense of sufficient future benefits accruing to offset present costs at even low discount rates. Note that this is an efficiency rather than an equity argument as discussed in Sections 2.4 and 4.2.1. He argued that it was more sensible to protect valuable public assets and the means to do this may include engineering solutions. Stoneham et al. (2000) proposed auction mechanisms as efficient ways for the community to ‘purchase’ land use change in locations of high conservation value.

Given the limited range of practical management options available to both farmers and the community, it is not surprising that most commentators (MDBC (1999) and Pannell (2001)) expect the area of salt affected land to increase and the quality of surface water systems to continue to deteriorate.

6.3 Soil Acidity

6.3.1 The Nature of Soil Acidity

The current acidity status of soils in Australia is the result of a combination of a natural process and agricultural impact. Natural processes operate on a near geological time scale. The impact of modern agricultural practices is much more rapid. For example, in some situations 20-30 years of agricultural practices have resulted in a similar amount of acidification as in tens of thousands of years of natural processes.

Acid and acidifying soils occur extensively in Australia especially in high rainfall crop or crop/pasture areas. Soil acidity, where surface soil or subsoil has a pH_{ca} of less than 5.2[26], results in poor performance of crop and pasture systems. Soil becomes acid through three mechanisms involving the nitrogen and organic carbon cycles. Cregan and Scott (1998) identified these mechanisms as the use of legumes, the harvesting or removal of agricultural outputs and wastes and the use of acidifying fertilisers. In general, the greater the productivity, the greater the potential soil acidification rate.

With the decrease in soil pH, that is, with the increase in soil acidity, imbalance in macro and micro nutrient elements occurs which seriously affects plant growth. It can cause aluminium (Al) and manganese (Mn) toxicities while inducing deficiencies of calcium (Ca), magnesium (Mg) and molybdenum (Mo). Phosphate availability in acid soils is low and added phosphate is rapidly rendered unavailable. Imbalances in soil nutrients can cause restricted root growth, adversely affect legume nodulation and can reduce the over-summer survival of rhizobia. Limited root growth restricts production of some crops and reduces animal production from perennial pastures (eg lucernes). To the extent that pH in the topsoil can be manipulated by the application of lime, soil pH is a renewable resource.

Acidification of the topsoil eventually leads to acidification in the subsoil. The development of toxicities in the subsoil causes the loss of deep rooted perennial plant species.

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[26] Soil is acidic when the pH falls below 7 but there is little effect on plant growth at this point.

Ameliorating acidity in the subsoil is a much more difficult problem and is generally regarded as being non-renewable although Cregan and Scott (1998) refer to a claim by Sumner (1995) that the technology now exists to ameliorate acidity in the subsoil. Other management strategies to be used in conjunction with lime include the selection of more acid tolerant crop and pasture species, and a reduction in stocking and fertiliser rates to reduce the rate of acidification.

6.3.2 Soil Acidity and Externalities

The temporal off-site effects of soil acidity are well known. Soil pH in the current period influences the choice of crop and pasture enterprises and the level of production from these enterprises, which in turn influence soil pH in the next period. As explained more fully below, farmers have to manage the acidity status of their land to maximise income over time. Soil acidity in this temporal dimension becomes an externality when it imposes unanticipated costs on future generations.

As discussed above, there is empirical evidence that the price of land reflects the extent of visible forms of degradation such as erosion. Many believe that the market for land ‘fails’ for less visible forms of degradation such as soil acidity and structural decline, although this hypothesis has not been empirically tested. The argument is that land values do not reflect soil pH status and hence landowners have an incentive to run down pH. One remedy is for information on soil quality to be available at the time of sale. It is not clear why such ‘sale by description’ could not emerge with a minimum of government intervention.

Acidification can also cause direct off-site effects in a spatial dimension such as the drainage of acid leachate causing fish kills in lakes or streams but indirect off-site effects are likely to be far more significant. The lower productivity and persistence of deep rooted perennial plant species on acid soils means that there is greater opportunity for invasion of weed species and erosion and greater accessions to the watertable which may result in salinity problems elsewhere in the catchment. These arguments are explained in more detail in Cregan and Scott (1998).

6.3.3 The Extent of Soil Acidity

According to the LWRRDC (1998) there may be 24m ha of agricultural land in Australia that is acidic with a pH of less than 4.8 and the value of production losses may be in the order of $134m with the area of acidic agricultural land in NSW and the value of lost production estimated to be 9.5m ha and more than $100m[27]. The 1986-87 land degradation study in NSW reported that about one third of SLAs concentrated in the southeast of the State but extending to the Riverina, suffered from severe induced soil acidity and that there were a large number of SLAs where soil acidity was likely to become severe (Gretton and Salma, 1996, p. C10). A survey conducted by Helyar et al. (1990) showed that in NSW about 13.5 million hectares of lands have a soil pH less than 5.0 which includes 8.5 million hectares of agricultural land. They also found that over 40% of the agricultural land that received more than 500 mm average rainfall was affected by low pH. The extent of the problem is of such concern to the NSW government that it has funded an Acid Soil Action

[27] Note that these numbers have not been revised since the 1995 report and it is not clear what year the dollar values relate to.

Program through a Treasury enhancement. There seems to be general agreement that soil acidity is both one of the most important soil health issues and that it is becoming more severe particularly in sandy soils, high rainfall areas and farming systems based on ammonium fertilisers (LWRRDC, 1998, p.8).

6.3.4 Economic Analyses of Soil Acidity

Economic analyses of soil acidity are reviewed in Islam et al. (1999). It would seem that most economic analyses of soil acidity have been attempts to measure the annual cost of soil acidity in terms of production foregone. One exception to this was the benefit/cost analysis by AACM (1995) of the use of lime to ameliorate acidity in eight regions in Australia vulnerable to production losses from induced acidification. In this study (also reported in Gretton and Salma 1996, pps E18-E22) it was estimated that the rate of lime application in Australia in 1989-90 was about 0.5 million tonnes and that this was about a quarter of what was required to maintain current pH levels. A further 2.3 m tonnes were required to ameliorate the 1.5 m ha where soil pH was less than 4.5. AACM found that generally lime application only had favourable internal rates of returns in cropping situations.

Other noteworthy research is that by Hochman, Godyn and Scott (1989) and by Trapnell (1998). Hochman, Godyn and Scott (1989) developed a simulation model called “Lime-It” which used marginal analysis to identify the returns from various rates of lime on subterranean clover pastures. Trapnell (1998) in his masters thesis gave an indication of the benefits of lime in north-eastern Victoria and south-eastern New South Wales.

Both Lime-it and Trapnell’s research are simulation models rather than optimising models. They compare alternative management strategies for lime use on the basis of standard investment criteria such as the sum of discounted returns over an investment period. The original version of Lime-it was envisioned as a decision aid for the amount of lime to be used in pasture systems but it has since been extended to cropping situations. The biological relationships are represented explicitly within the model as soil, pasture and livestock modules. There is also an economics module which Trapnell (1989) was somewhat critical of.

Trapnell used a spreadsheet based budgeting approach for three soil types and a range of cropping/pasture rotations. The biological relationships were represented as fixed coefficients in the budgets as opposed to being explicitly incorporated in the model as for Lime-it. These coefficients were derived from research by Slattery and Coventry (1993).

Both these approaches have some dynamic capabilities in the sense that they maximise discounted returns over an investment period where the returns in any time period are a function of soil pH which in turn responds to management options over time. However as simulation models they do not have an optimising capability and hence only by simulating a large number of scenarios varying by lime use, pH and crop and pasture rotation, is it possible to identify strategies approaching optimality.

In general, past economic analyses have not adequately addressed the dynamic nature of the acidity status of soil. Soil acidity can be regarded as a natural resource stock which must be managed through time. As pointed out by McInerney (1976), the key principle in a managing natural resource stock, such as the level of soil acidity, is to equate the marginal benefits from running down soil pH in the current period with the marginal user cost, MUC, of this strategy. The MUC includes the value of production lost in future time periods from the reduction in pH in the current period. There is a dynamic element to this problem in that production in the current period is affected by current pH and in turn has an impact on next period's pH. Ignoring this marginal user cost leads to higher rates of soil acidification than is optimal and a reluctance to apply lime. Islam et al. proposed a truly dynamic approach to this issue in their future research program and this is being pursued by Eigenram of DNRE, Victoria.

6.3.5 Management strategies

Economic analyses have generally found that a liming strategy is likely to be profitable at some level of soil pH in broadacre cropping and more intensive horticultural situations. Many grain growers regularly apply lime although it is not clear whether there remains any economically significant divergence from optimal soil pH from the viewpoint of the community.

However cropping is not part of the farming system in many parts of the slopes and tablelands of south east Australia. A feature of these farms is the limited number of diversification options that are available. Enterprises have traditionally been limited to meat and wool production by grazing sheep or cattle. At least under prices prevailing in recent years, the use of lime in extensive grazing situations is unlikely to have been profitable. Furthermore the ability of graziers to use opportunity cropping to incorporate lime is limited by their lack of cropping experience and infrastructure. From the farmers' viewpoint, the appropriate management strategy may be a low stocking rate, low fertiliser native pasture regime that slows the rate of acidification.

If it is not in the interests of farmers in these areas to ameliorate the rate of soil acidification then consideration needs to be given to whether this land degradation is a private matter or has broader community implications. Significant externalities may arise if the land market does not adequately reflect the acidity status of the land.

6.4 Acid Sulfate Soils

6.4.1 The Nature of Acid Sulfate Soils

Along Australia's coastline but particularly along its northern and eastern coastlines is a layer of soil containing iron pyrite, acid sulfate soil (ASS). There is a risk of this condition wherever elevation is 10m Australian Height Datum (ahd) or less. Normally this layer of soil is below the watertable. However where this layer of soil is close to the surface and is exposed to oxygen in the course of agricultural or urban development there is a build up of acid in the soil. This acid discharges into waterways when the soil is saturated after rainfall events with the potential for fish and oyster kills and other environmental impacts. The oxidation of ASS occurs during dry times.

Because of their low elevation and proximity to major waterways, lands associated with ASS are often wetlands or prone to periodic flooding. Consequently these lands have been subject to extensive public and private drainage works both as a means of flood mitigation and to increase agricultural production. In addition an extensive system of floodgates has been used to restrict saltwater from low lying lands and wetlands.

Many of the larger public drains went deep into the ASS layer which means that there may be significant quantities of ASS spoil along each drain from the initial excavation and subsequent maintenance. In addition the combination of public and private drainage works has meant a significant drop in the watertable and exposure of ASS to oxygen where the ASS is close to the surface. Clearly the depth to ASS is of great significance. Where ASS is less than half a metre from the surface, soil pH is likely to be low and the frequency and extent of acid runoff is likely to be higher than for situations where the pyritic layer is deeper.

The production of sulfuric acid from disturbed ASS is large. It would take hundreds of years for natural processes to neutralise these soils. The production of acid is so large that liming is apparently not an economic proposition at current prices and technology. Because of the tremendous volume of oxidised ASS, it will be impossible to avoid acid discharge in high rainfall events. Hence there is an important stochastic element to this problem in that the amount and frequency of discharge depends on rainfall, as well as the depth of the watertable and the density of the drainage system.

6.4.2 Acid Sulfate Soils and Externalities

ASS is an instance of land degradation where externalities are likely to be of greater significance than on-farm impacts. Costs imposed on the community include poor water quality and an associated loss of wetland biodiversity; lost production from the commercial and recreational fishing industries and the oyster industry; and infrastructure costs from acid corrosion.

Whether the drainage of wetlands and associated development of ASS has imposed on-site costs on agriculture is unclear. On the one hand a build up of acidity in the root zone of plants, which has undoubtedly occurred in places where the ASS layer is close to the surface, results in production losses. However, as pointed out by Mullen and Kaur (1999) and discussed below, depending on how the benchmark is defined, the value of agricultural production may have risen as land has become available after draining. Sturgess, Read and Associates (1996) estimated that most of the benefits from better management of ASS in the Tuckean Swamp flowed to the commercial and recreational fishing industries rather than agriculture.

Because of pervasive externalities and public sector involvement in much of the drainage work, the role of government in managing ASS is likely to be significant. Much of this work to modify watertables was undertaken before the consequences of the oxidation of ASS was understood. In addition much of the drainage and levee works were financed by the public sector and had an objective of flood mitigation as well as increasing agricultural production. This work has resulted in significant quantities of ASS spoil and large areas of acidified land, important sources of acid discharge. Hence there are strong arguments in support of continued public involvement in ameliorating the impact of these past policies. Now that the consequences of ASS are well known, there is a clear role for government in ensuring that any new agricultural or urban developments on ASS bear the full cost of acid discharge from these sites.

Even at the 'farm level' there are strong arguments for some government involvement, although the nature of this government involvement is less clear. Much of the agricultural development that occurred when the public drainage system was implemented was with the encouragement of government through extension and research programs. Hence it is appropriate for government to at least partly fund research and extension programs into ASS issues. In addition, there are issues of externalities and property rights which extend to the broader community.

6.4.3 The Extent of Acid Sulfate Soils

According to Firztpatrick et al. the distribution of ASS in Australia is 'unknown at any scale'. Estimates of the area of ASS in coastal areas range from 5m ha by Fitzpatrick et al. to 2m hectares by Sammut and Lines-Kelly (1997). There is a growing appreciation but little information about the occurrence of ASS in inland areas in association with mining activities but most interest centres on coastal ASS which is the subject of a 'National Strategy for the Management of Coastal Acid Sulfate Soils' (National Working Party on Acid Sulfate Soils, 2000).

Coastal ASS is concentrated on the northern and eastern coastlines. The best sources of physical information about the distribution of ASS in NSW are the ASS Risk Maps and associated databases assembled by the DLWC. The nature of the data available and their interpretation can be found in a publication from DLWC titled 'Guidelines for the Use of Acid Sulfate Soil Risk Maps' (Naylor et al., 1998)[28]. One set of maps classifies land by its probability of having ASS (or PASS) somewhere in the soil profile however deep. A summary of this information indicates that in NSW there are almost 260,000 ha of land at high risk of ASS with about 150,000 ha being on the North Coast of NSW.

6.4.4 Economic Analyses of Acid Sulfate Soils

Economic analyses of ASS include those by Sturgess, Read and Associates (1996) and by Mullen and Kaur (1999). The former was an analysis of alternative strategies to manage ASS in the Tuckean Swamp on the North Coast of NSW. The latter was an attempt to review some of the economic issues associated with ASS in NSW.

As far as I am aware there have been no estimates made of the cost of ASS to either agriculture or the wider community. Mullen and Kaur (1999) pointed out some of the difficulties associated with establishing a benchmark from which to assess losses in agricultural production associated with ASS. The benchmark might be the production from land and waterways prior to any modifications to the watertable where the only acid discharges are the result of ongoing natural processes. An immediate problem is setting the date when watertable modification started inducing ASS problems. This date may well differ between catchments.

Clearly for this benchmark, the value of agricultural production will be quite low as much of the agricultural production we see today would not have been possible without modification of the watertable. Hence using this benchmark, it is most likely that the modification of

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[28] Note that the data we are using is from the First Edition of these maps.

watertables has increased the value of agricultural production despite the unforeseen ASS problem which is likely to have decreased the value of production from other industries.

Another benchmark might be the value of agricultural production either achieved initially or expected after the modification of the watertable. Under this benchmark, the costs of ASS to other industries is probably similar to that under the former benchmark. However it seems likely that the emergence of ASS problems has meant that agricultural production has not increased to the degree anticipated (or declined after an initial spike) especially for the extensive grazing enterprises. While the area of land available for grazing has increased, the land that has become available is much more difficult to manage because of ASS which in some instances may mean that it is actually less productive than when it was a wetland for part of the year. Under both benchmarks an important unquantified benefit from at least some of the drainage works has been flood mitigation.

The study by Sturgess, Read and Associates (1996) appears to be the only comprehensive analysis of alternative approaches to managing ASS in a particular environment, the Tuckean Swamp. The Tuckean Floodplain, part of the catchment of the Richmond River on the North Coast of NSW, is comprised mainly of estuarine and freshwater wetlands. The majority of floodplain has now been drained and cleared for agriculture. The construction of the Baggotville barrage across the Richmond Broadwater in 1971, eliminated tidal flushing and saltwater intrusion in the Tuckean floodplain.

It is believed that drainage of the floodplain and construction of the barrage have lowered the watertable and hence led to oxidation of pyrite in the Tuckean floodplain. In some parts of floodplain, soils are highly acidic with pH less than 3 being found in soil and water samples. There is growing concern that highly acid flows into the streams of the floodplain and the adjacent Richmond River have reduced the abundance and diversity of fish and water birds with adverse impacts on the fishing and tourism industries.

Sturgess and Read were commissioned to evaluate a management plan for the Tuckean Swamp which included reinstating tidal flows, the purchase of some re-flooded agricultural land, drainage works and liming. The objective of the study was to identify and estimate the benefits and costs, to all those with a stake in the Tuckean Swamp, of specific changes to the way in which the Swamp was to be managed. Hence the analysis was conducted at the level of the catchment rather than at the level of individual farms. They found that the proposed management changes were profitable but the benefits largely accrued to the fishing industry rather than to agriculture.

6.4.5 Management strategies

While it appears that with present technology there is little prospect of removing the stock of ASS, there is some prospect that the acid discharge from this stock could be reduced or managed. In summary it would seem that an appropriate objective might be to manage acid discharge into waterways in such a way as to maximise the wealth from a range of industries in a catchment with some regard for the distribution of benefits and costs on different industries within the catchment. Such an approach would require an assessment of the benefits and costs of a range of management options. These benefits and costs will be difficult to estimate for a number of reasons. They are related to rainfall and hence have a stochastic nature. Some of the benefits particularly those related to environmental flows are unpriced. There is great uncertainty about biological relationships between pH in soil and water and production from agriculture and fisheries. Particularly in fisheries but perhaps also in agriculture there are important dynamic effects which must be accounted for.

There are three components to managing acid discharges. One component is the discharge from ASS spoil along the public drainage system. The second is the discharge from urban development. The third component is the discharge from land used in agriculture. Management practices in agriculture include the redesign of on-farm drains and landforming to quickly remove rainfall from land before the watertable rises bringing acid to the surface and the use of lime where profitable. At a catchment level, options in include returning land to wetland and restoring tidal influences in estuaries presently controlled by barrages.

6.5 Erosion

6.5.1 The Nature of Soil Erosion

Soil erosion involves the loss of topsoil either uniformly (sheet erosion) or in channels (rill erosion) or in gullies (gully erosion). Soil erosion is a natural phenomenon that is part of the aging of the continent. Generally soil erosion is classified as a non-renewable resource issue because the natural rate of soil development is extremely slow[29]. Agriculture contributes to soil erosion through the changes it causes to ground cover, soil structure and water flow. Generally we think of this contribution as being negative but conceivably agricultural technologies could be used to slow the rate of soil erosion below its natural rate in some areas if that was in society's interests. The main forms of soil erosion and the factors that contribute to erosion are defined in ENVALUE (p.28).

There may be both on- and off-site costs associated with erosion. The main on-site cost of soil erosion takes the form of lost production, increased fertiliser use and the loss of visual amenity. Following ENVALUE closely, externality costs include the diminished value of recreation facilities because of siltation and turbidity, damage to infrastructure such as roads and dams, a deterioration in the quality of drinking water and damage to aquatic ecosystems. There is some evidence that soil erosion is linked to or caused by other forms of land degradation such as acidity and salinity.

Soil erosion is a dynamic resource issue in that current production depends on the depth of topsoil and current production practices influence the depth of topsoil in the future.

6.5.2 The Extent of Soil Erosion

The LWRRDC (1995, p.10) noted that a Commonwealth-state survey (Woods, 1983[30]) of land degradation found two thirds of cropland, one third of non-arid and over half arid grazing lands were seriously affected by erosion. It noted that the costs of erosion 'are poorly quantified except for specific sites and reported values vary widely (LWRRDC, 1995, p.10). The report pointed to evidence that with the reduction in the rate of land clearing, the rate of soil erosion was probably past its peak. In its more recent report, the

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[29] Miranowski (1984) referred to US research suggesting that the rate of soil genesis may be up to five tons per acre per year.

[30] This study is also discussed in Gretton and Salma (1996, pps 40, a1, c1).

LWRRDC (1998, p.7) seemed ambivalent about the rate of soil erosion suggesting in one section that the rate of soil erosion may be increasing again but in another (p.10) that in some areas the rate of soil erosion is falling.

Gretton and Salma (1996) also provided estimates of both the extent of the various types of soil degradation (Appendix C), largely based on the 1987-88 Land Degradation Survey conducted by the Soil Conservation Service of NSW and of the associated costs - most often in terms of the value of production forgone (Appendix E) - drawn from several sources including the EPA (1993). According to the Land Degradation Survey about ten percent of NSW suffered from severe soil structure decline and gully erosion. The EPA (1993) estimated that the annual value of the decline in agricultural productivity due to gully erosion was $5.2m.

At a regional level using GIS information, Mallawaarachchi et al. (1994) estimated on a production equivalent basis, that the loss in income from sheet and rill erosion in the Lachlan Valley could be valued at $21.1m or six percent of the value of production in the region in 1989-90 at prices for that year. The LWRRDC report (1998, p.7) noted an estimate for the off-site costs of soil erosion in Queensland of at least $31m per year (1989 dollars)

6.5.3 Economic Analyses of Soil Erosion

ENVALUE reported a range of site specific estimates of the on- and off-site costs of erosion. Most studies used a dose-response methodology but the study by King and Sinden (1988) used hedonic pricing methods and concluded that there was evidence that at least for soil erosion in the Manilla Shire, losses associated with soil erosion were reflected in land values.

US studies by Walker (1982), McConnell (1983) and Miranowski (1984) that analyse soil erosion in the broader context of the management of a natural resource have been referred to above. Soil erosion seems to be the land degradation issue that has had the widest range of economic methodologies applied to it (at least in the US). Such studies require physical models to better predict the rate of soil erosion and to relate the loss of topsoil to the level of production. A range of Australian and US models are available but according to the LWRRDC (1995) none are robust for the wide range of farming systems and climatic conditions found in Australia.

Miranowski (1984) used a multi-period LP model to study how farmers in the US cornbelt might change tillage and cropping rotations in response to soil erosion to maximise variable profits over a fifty year period. In year 50 there was a penalty function for the loss of topsoil. Yield in each period was related to the depth of topsoil which in turn was depleted from year to year according to the Universal Soil Loss Equation developed by Wischmeier and Smith (1968). The drawbacks of this model were that while soil loss was modelled through time, tactical response was not possible nor was there any attempt to account for risk.

In a study of soil erosion in the US, Walker (1982) used DP techniques to identify when a farmer on a particular soil type should switch to conservation farming technologies. Neither climate nor price were treated as stochastic variables. The Walker study constrained the farmer to choice from only two strategies. Optimal control models are based on the mathematical programming techniques of microeconomics with a time dimension but cannot accommodate stochastic events. Examples of the use of optimal control modelling include McConnell's (1983) US study of soil erosion.

6.5.4 Management strategies

Soil erosion is controlled by maintaining ground cover, hence one reason for interest in reduced tillage techniques and shorter fallow periods is the contribution they make to erosion control. Cregan and Scott's (1998) reference to the role of the water cycle in erosion, soil acidity and salinity has already been noted. Land suffering from acidity or salinity is likely to have poor groundcover and hence be prone to erosion. In like manner, strategies to combat acidity and salinity based on reduced fallow and perennials will contribute to erosion control.

6.6 Soil Structural Decline

6.6.1 The Nature of Soil Structural Decline

Soil structure decline is often associated with tillage practices and a decline in soil organic matter and refers to an undesirable change, through compaction for example, in the arrangement of soil particles and air spaces that is evident in surface sealing, crusting and hard-setting. The on-site losses from structural decline are in the form of greater seedbed preparation costs and lower establishment rates; and reduced uptake of nutrients and water by plants because of restricted root growth. Minimum tillage and controlled traffic practices are being used to ameliorate the problem. There may also be externalities in the form of increased water runoff and erosion because of the lower rate of water infiltration associated with structural decline.

6.6.2 The Extent of Soil Structural Decline

There appears to be little data about the extent of structural decline and its cost in terms of foregone production. Reeves et al. (1998) quoted an estimate of 55m ha of land in Australia suffering from structural decline. According to the land degradation survey conducted by the NSW Soil Conservation Service (1989), the area suffering from moderate and severe soil structure decline was 14.7m ha. and was most apparent in the wheat-sheep zone. This would make soil structural decline the most widespread form of land degradation although few estimates have been made of the losses it is causing. The EPA (1993) estimated that the annual value of the decline in agricultural production in NSW due to soil structure decline was $144m, suggesting that the problem is more significant that soil acidification.

6.7 Summary of Soil Issues

There have been few comprehensive national assessments of land degradation and their findings are difficult to compare because they have used different methodologies and benchmarks. Similar problems beset cost of production studies of particular land degradation issues. As Reeves et al (1998, p.17) concluded, '…there are no robust data sets on which to make firm conclusions about the extent of soil degradation and whether or not the degree of degradation is increasing’. Perhaps the National Land and Water Resources Audit to be completed in 2002 will prove an adequate benchmark against which to judge the progress of land and water degradation in the future.

The interdependence of major types of land degradation was noted by Cregan and Scott (1998); Passioura and Ridley (1998) and Gretton and Salma (1996, p.C12). Despite this, much research, literature and policy responses, at least implicitly, treat land degradation issues as though they were independent. Stoneham (pers. comm.) has suggested that government could manage land degradation externalities more efficiently were their joint occurrence explicitly recognised.

An important area for future research is to identify the joint occurrence of land degradation issues. It may be that some areas of high rainfall non-arable land along the ranges are a significant source of externalities associated with land degradation. The payoff to the community from land use changes in these areas, induced using some of the mechanisms reviewed by Stoneham et al. (2000), may be quite high.

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