Salt-Affected Soils: Their Cause, Management and Cost

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Damage to the productive capacity of soils by salts is a major form of land degradation in the winter dominant–high rainfall areas of southern Australia. Recent surveys suggest that at present, 2.5 million hectares of non-irrigated land are affected by salt, while approximately 360 000 hectares of irrigated land in the Murray–Darling Basin are salt-affected. Increases in salt in the root zone of irrigated crops and pastures have occurred by the over-application of irrigation water, or by the application of salt in irrigation water where insufficient leaching occurs. Conversely, dryland salting has occurred mainly as a result of clearing of native vegetation and its replacement by agricultural crops and pasture species, resulting in a net deficit of water use and an increase in groundwater recharge. Due to the cause of dryland salting, the amount of land affected is likely to increase with time.

Whereas the causes are well understood, the socio-political mechanisms to prevent salt damage or to restore salt-affected land are less clear. Farmers are frequently accused of causing salt problems and are then expected to pay. However, salting is also a community problem. Therefore this problem should be shared financially by the three major stake holders: government, consumers, and agricultural producers.

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PERSONAL COMMUNICATIONS

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1 INTRODUCTION

Despite low birth rates in many western countries, the global population is increasing at an exponential rate. A major challenge facing agriculture is to increase food production at an equivalent rate. The mission of the International Centre for Agricultural Research in the Dry Areas (ICARDA) is '... to improve the welfare of people through agricultural research and training in the dry areas in the poorer regions of the developing world, by increasing the production, productivity and nutritional quality of food to higher sustainable levels, while preserving or improving the resource base' (ICARDA 1998). This charter is shared by many national and international agricultural institutes, and several common channels exist by which this generic charter can be globally achieved. In the exploration of new opportunities to increase food and fibre production, a lot of effort has recently been spent reviewing the agricultural development potential of semi-arid and arid regions of the world. Not only is there a perception that there are gains to be made by improvements in agronomy and in the induction of improved plant varieties, but gains may also be achieved by the strategic application of water.

Great success has been made in converting inhospitable desert regions to prime agricultural lands with the supplementation of irrigation (Bresler *et al.* 1983). However, in some of these areas which are either unsuited to irrigation, or when irrigation has been poorly managed or water of poor quality used, a major problem frequently encountered has been the accumulation of soluble salts (Fig. 1). These impose an additional stress on growing crops which frequently results in yield loss. In addition, clearing of native vegetation and subsequent inappropriate agricultural development in the non-irrigated, higher rainfall zones has lead to loss of substantial tracts of land to dryland salting. This has been particularly apparent in the zones of southern Australia where rainfall patterns are winter-dominant and which receive greater than 500 mm of rainfall per annum.

Although there are many different forms of degradation caused by salts, all forms share one feature in common; the overriding effect of electrolytes on soil properties. In this respect, the concentration of electrolytes, when above a threshold level, affects the physical, chemical and biological properties of a soil, which in most instances renders the soil to be of low fertility and limits the agricultural usefulness of the affected land (Szabolcs 1989). Typical agricultural methods such as the use of agrochemicals or fertilisers generally have little effect in ameliorating the effect of a particular salt. The impact of these electrolytes is so manifest that until the concentration of electrolyte is reduced, agricultural production will be unduly affected.

Fig. 1a. Salt accumulation on the surface of soil in the north of the Middle East as a result of irrigation with saline water (photo used by courtesy of Dr M. Zöbisch).

Fig 1b. Salt accumulating on the banks of irrigation bays by the process of capillary rise of salts from saline groundwater; the result of over-irrigation (photo used by courtesy of Dr M. Zöbisch).

Fig 1c. Salt accumulating at the edge of irrigation paddocks as a result of capillary rise from saline groundwater in the Murrumbidgee Irrigation Area (photo P. Eberbach).

Fig 1d. Saline discharge area in the Mallee Region of north-western Victoria (photo P. Eberbach).

Salts in soil are not uncommon and are in fact essential for plant growth. However, excessive quantities of electrolytes, as in the case of salinisation, or above threshold levels of particular ions, affect the production of sensitive species and may affect the soil architecture; affecting soil hydraulic and other physical properties, frequently detrimental to the production potential of soil. The purpose of this chapter is to consider the environmental issue of salt in the landscape, with particular emphasis on types of salt damage, the extent of salt-induced land degradation in Australia, cause and effect and how salt damaged land may be managed. Salt-affected soils in dryland agriculture, irrigation and the urban environment are considered. Many monographs have been devoted to various aspects of salt as a vehicle of land degradation but few have considered the issue of who should pay in terms of preventing salt damage and in correcting salt-induced degradation. The final section of this chapter considers the cost of managing salinisation and who should pay.

2 TYPES OF SALT-AFFECTED SOIL

Numerous types of salt-affected soil exist. But each type reflects the effect either of the total concentration of dissolved electrolyte in the soil solution or the concentration of a particular ion. The key features of some of the more common salt-affected soils are summarised in Table 1. Although Table 1 summarises several aspects of four categories of salt-affected soils, in Australia, saline, sodic and combination saline-sodic are the most common categories observed.

Saline soils

Saline soils have in common an excessive concentration of dissolved salts. Although saline soils are not necessarily affected by a particular electrolyte species, the most commonly occurring electrolytes in these soils are sodium, calcium and magnesium with chloride, sulfate and bicarbonate. The problem these soils present to plant growth is the effect of an excessive concentration of electrolyte which reduces the osmotic potential of soil water, hence reducing the availability of soil water for plants (Szabolcs 1989). In these soils, plants are typically water stressed more readily than when growing in non-saline soils. In addition, one or more of the ions may be toxic. However, an important feature of saline soils is that frequently they are well structured as the high electrolyte concentration assists in flocculating clays by compressing the diffuse double layer and reducing repulsion between opposing clay faces and allowing attractive forces to dominate (Mullins *et al.* 1990).

Chemical diagnosis of a saline soil can be quite time consuming and expensive. However, Scofield (1942) showed that a good relationship existed between electrical conductivity (EC) of the soil solution and the level of dissolved salt. Although numerous methods exist to determine EC, the two most common procedures are saturated soil paste extract and the soil water (1:5)

Table 1. Grouping of salt affected soil

Major electrolytes causing salting problem	Type of salt-affected soil	Main adverse effect on production	Method of reclamation	Electrical conductivity[A] ($dS\ m^{-1}$)	Exchangeable Sodium Percentage (ESP)
Any cation and anion, particularly sodium, magnesium and chloride or sulfate	Saline	High osmotic strength of soil solution, may also be toxic	Removal of excess salt by leaching	>4	<15
Sodium ions causing alkaline hydrolysis	Sodic	Alkaline pH, structural breakdown reducing root penetration, water infiltration and aeration	Replacing sodium with calcium	<4	>15
Magnesium ions	Magnesium	Toxic effect, high osmotic strength of the soil solution	Chemical amendment with calcium	>4	NA
Ferric and aluminum ions (mainly from sulfate)	Acid sulfate soils	Strong acid-producing, Toxicity of Al	Liming	NA	NA

[A]saturated paste extract.

NA, not applicable.

Source: adapted from Szabolcs, I. (1993) *Salt-Affected Soils*.

mass/volume method of determination. Soils considered saline using the saturated paste extract have an EC of greater than 4 dS m^{-1}, whereas when the soil water (1:5) dilution method is used, soils considered saline have an EC in excess of 0.4 dS m^{-1}.

Although the ECs of these soils are considered to be threshold levels, where a soil is declared saline, the response of plants to salinity is very dependent on climate, soil water holding capacity of soil, salt composition, plant species and the plant's stage of development. For example, in cool climates, where transpiration is low, plants tend to be more tolerant of salinisation than where transpiration demands are high, largely because their demands on available soil water are low.

Sodic soils

Much of the Australian landscape is derived from ancient marine sediments, and hence it comes as no surprise that sodium plays a key role in the behaviour of many Australian soils. The inimical influence of exchangeable sodium on soil physical properties as a result of its effect on clay dispersion is widely known (Mullins *et al.* 1990). In the presence of high concentrations of exchangeable sodium, clay particles tend to disperse. The presence of sodium in the Stern-layer of layer silicate clays increases the radius of the diffuse double layer surrounding the clay particle, increasing repulsion between the clays. Dispersion of clay ultimately leads to structural deterioration, and once dispersed, the affected soil horizon is likely to form a hard surface crust or a hardset subsurface layer on drying. This apedal layer hinders infiltration of water and air and greatly impedes the growth and development of plant roots.

Overseas recommendations suggest that an exchangeable sodium percentage (ESP) of 15% be used to delineate between non-sodic and sodic soils. However, much Australian data show that dispersion of soils can occur at an ESP considerably below 15. Emerson (1977), from an investigation of 51 surface soils (0–10 cm) showed that the dispersive effects of sodium commenced at an ESP of about five and were complete at about 10. Similarly, McIntyre (1979) suggested that an ESP of five was more appropriate for Australian soils. A critical value for an ESP of six has been accepted for Australian soils to be considered sodic. The much lower ESP necessary for dispersion to occur in Australian soils is thought to be due to the high proportion of clays in Australian soils and to the lower calcium to magnesium ratio.

Sodic soils fetter the growth and development of plants in two ways:

- the poor structure and hydraulic properties of sodium-affected soils physically restrict the development of roots and offer a very poor biological and chemical environment for plant roots (Mullins *et al.* 1990); and
- chemically, sodic soils can be highly alkaline due to the hydrolysis of exchangeable sodium from the surface of clays (Szabolcs 1989). Also soluble sodium may be toxic to the plant.

Saline-sodic soil

Saline-sodic soils satisfy both criteria; being that of higher than threshold level of dissolved electrolytes and a preponderance of exchangeable sodium. Similar to non-saline sodic soils, saline-sodic soils can be highly alkaline; however, unlike sodic soils, their structural characteristics conform more closely to saline soils.

3 ORIGINS OF SALT IN THE LANDSCAPE

A commonly asked question is 'where do salts originate?' Salting frequently affects soils of the arid and semi-arid zone where salts concentrate in the root zone. This occurs as rainfall in these climatic zones is insufficient to flush the soil of accumulated salts. Another condition by which salts accumulate in the surface layer and may affect plant production is in areas of high saline groundwater. When highly saline groundwater is close to the surface, or when rising groundwater mobilises salts stored beneath the root zone, salts move into the root zone with the upward capillary movement of water from the groundwater body (Szabolcs 1989). Capillary rise of water occurs in response to evaporative demand and salts concentrate in the root zone as the carrier evaporates or is transpired (Abrol 1986; Abrol *et al.* 1988). These are the two major mechanisms of dryland salting and will be considered later in this chapter.

The movement of water is integral to damage of soils by salts, however, it is important to acknowledge that water is not the actual source of the salts; these are derived mainly from historic reserves, weathering of parent materials, rainfall, and the addition of water via irrigation.

Historic reserves

Like much of the Middle East and Africa, most of the Australian continent is marine in origin; laid down by sedimentation. Given the low leaching environment which exists over a considerable proportion of inland Australia, much of the salt entrapped with the former marine sediments remains within the regolith, frequently in the surface layer of soils of the arid zone. As an example of salt storage, McFarlane and George (1992), in a paired catchment study, calculated that between 1204 and 4528 tonnes of salt per hectare were stored within the unconsolidated regolith of the two catchments. Taking into account the different sizes of each catchment, the average concentration of salts between the two catchments varied between 6.9 and 9.9 kg m^{-3}. Provided these reserves of salt remain beneath the root zone, they present little problem to agricultural production. However, when circumstances change and they are mobilised, these reserves may concentrate within the root zone. Such circumstances may result from a change in local hydrology (which will be discussed later in this chapter), or in weathering of a protective layer within the regolith (Bresler *et al.* 1983).

Mineral weathering

Many of the world's arid and semi-arid regions are in a relatively unweathered state due to the lack of soil moisture. Although this unweathered material provides an excellent source of plant nutrients, it also provides a source of salts which may accumulate and adversely affect plant growth (Bresler *et al.* 1983).

A considerable portion of arid and semi-arid Australia is composed of ancient weathered parent materials, much of which are marine in origin. In addition, due to the age of the continent and the complexity of landforming events, many areas have a heterogeneous mix of parent materials. For example, like much of the south-western slopes and eastern Riverina of New South Wales, the Yass River catchment in south-eastern Australia has several principal types of parent material. These include Ordovician metasediments, Silurain acid volcanics and aeolian silts and clays (Nicoll and Scown 1993). Further west, the Houlaghans Creek catchment is similar except for major alluvial deposits in the drainage line (Sutherland *et al.* 1993). Both catchments exhibit saline seeps in specific areas. However, in the Yass River catchment, salt contributions are considered to be in part derived from the weathering and leaching of the metasediments and aeolian deposits (Nicoll and Scown 1993), whereas salts in Houlaghans Creek catchment are thought to be derived from weathering and leaching of the underlying granite as well as the aeolian clays. These aeolian clay parent materials in eastern Australia are a major source of salt as they were originally from groundwater discharge lakes in western New South Wales (Nicoll and Scown 1993).

Rainfall

Whereas electrolytes dissolved in oceanic waters originated as the product of the weathering of rocks, oceans are now a major salt reserve for the terraqueous environment. The major mechanism for distribution from this source occurs when water droplets from oceanic spray and turbulence produce an airborne suspension of salt crystals. These crystals act as a nuclei for condensation, forming raindrops, and are distributed over the landscape in rainfall (Bresler *et al.* 1983). Cope (1958) termed these additions of salt as 'cyclic' additions.

Rainfall is generally considered to be a major contributor of sodium chloride salts in salt-affected regions of inland Australia (Nicoll and Scown 1993). Measurements of salts in continental rainfall have shown that salt concentration varies with distance from the sea, topography, prevailing wind direction, and rainfall intensity. Continental fallout of NaCl in rainfall has been estimated to vary annually from 20 to 200 kg ha^{-1} (Mason 1964), with values in coastal areas of about 100–200 kg ha^{-1}, whereas in inland areas, values of about 10–20 kg ha^{-1} are common (Blackburn and McLeod 1983).

Irrigation water

Like cyclic additions in rainfall, solutes dissolved in irrigation water contribute to the input of salt to irrigation areas. The United States Salinity Laboratory Staff (1954) produced a water classification scheme used to rate quality of irrigation water based on the EC and SAR (sodium adsorption ratio) of the water under consideration. It is interesting to note that in the above publication, even when using the highest quality irrigation water, occasional leaching to remove the imported salt was considered necessary.

4 EXTENT OF SALT-AFFECTED LAND IN AUSTRALIA

When considering the extent of salt-affected land, the distinction needs to be made between primary and secondary salting. Primary salt-affected land is a natural part of the landscape such as saline marshes and salt scalds. As an example, 250 000 hectares of land in the state of Victoria are estimated to be affected by primary salting (State of the Environment Report (SER) 1991). Conversely, secondary salting occurs as the enhancement of natural salting processes by human activities. When considering the extent of salt-damaged land, it is worth considering where possible the distinction between the two forms of salting, particularly given that change in the amount of land affected is likely to be induced by human activity and hence occur as secondary salting.

Dryland salting

The increase in incidence of salinisation of soils and streams in Western Australia was first recorded in the early part of this century by railroad engineers (Bleazby 1917; Wood 1924); however, in contrast, in dryland agricultural areas of New South Wales and Victoria, salinity has been regarded as a more recent phenomenon (State of the Environment Advisory Council (SEAC) 1996). Clearing the landscape of perennial native vegetation and its replacement by short-growing-season annual crop and pasture species has been implicated as the major cause of secondary salting in dryland regions (SEAC 1996).

Recent national estimates suggest that 2.5 million hectares of land are affected by dryland salting (Anon. 1997). As the process of salinisation is temporal, current trends in the process of salinisation suggest that the amount of land affected by salts will increase into the future. In the *Australia, State of the Environment* report (SEAC 1996) the breakdown of this figure estimated that for Western Australia approximately 1.6 million hectares (or 9% of land cleared for agriculture) were salt affected, and 1990 estimates from the Murray–Darling Basin Commission suggested that 200 000 hectares of the basin were salt affected while a further one million hectares were at risk (MDBC 1993). At about this time in South Australia and Victoria, areas affected by dryland salting were estimated as 400 000 hectares (SEAC 1996) and 135 000 hectares (SER 1991) respectively. The figure for

Western Australia appears reasonable, however, other reports have suggested that figures for the Murray–Darling Basin may underestimate the amount of salt-affected land (MDBC 1993).

Predictions of the growth in salt-affected land are difficult as the appearance of secondary salt damage takes time; and although caused by changes in vegetation structure, the discharge areas are often features of topography and underlying geology (Tuckson 1995).

Irrigation salting

Given that the majority of irrigation lies within the Murray–Darling Basin, it is hardly surprising that most of the land affected by salt as a result of irrigation occurs within this region. However, experience gleaned from the Murray–Darling Basin has been adopted by the managers of other irrigation areas, such as in the Burdekin irrigation area in Queensland, to instigate management procedures to reduce the rate of salting. Irrigation can lead to salting by two major pathways: over application of irrigation water, directly recharging the aquifer beneath the irrigation area causing groundwater levels to rise; and by the addition of salts dissolved in irrigation water to land where insufficient leaching occurs to leach away excess salts (both mechanisms will be discussed further in a later section).

Estimates made of the extent of irrigation-induced salting by the Murray–Darling Basin Commission in 1985 showed that about 360 000 hectares had high watertables of which 87 000 hectares of the Victorian part of the Basin were visibly salt affected (SEAC 1996). More recent estimates in Victoria showed that of a total of 465 000 hectares of land currently irrigated, 140 000 hectares were now affected by secondary salting; about 30% of the total land irrigated in Victoria (SER 1991). Of this, the Loddon–Avoca region was the worst affected with an estimated 102 610 hectares of a total of 212 000 hectares of irrigated land affected by salt. In the Shepparton Irrigation region, high watertables have developed over the past century to the extent that now 68% of the area (190 000 hectares) have watertables within 2m of the ground surface, and 30% of the area have watertables between 1 and 2 m of the ground surface (SER 1991). In the Riverina region of New South Wales, major increases in the amount of irrigation-induced salt-affected land have occurred over the past decade (Slavich 1992).

Urban salting

In more recent times, off-farm damage by salting has become apparent, with numerous inland urban communities reporting the presence of high watertables and an increase in the number of salt seeps. This high watertable and increase in salt has accelerated the deterioration in public infrastructure such as roads and sewerage systems and damaged private infrastructure. The damage in the urban environment to public and private infrastructure may be more costly than in the rural environment as salt damage in a rural context is generally production orientated (Christiansen 1995).

Although the recharge of water in the urban context may in part be derived from recharge off adjacent rural holdings, land practices in the urban environment are likely to be major contributors (Oliver *et al.* 1996). A survey of about 40 state government agencies and 240 local government bodies in the Murray–Darling Basin was conducted by the Australian Bureau of Agricultural Resource Economics (ABARE) during 1994–95 to assess the costs of salinity to government bodies and public utilities in the basin. This survey revealed that of all the respondent local government bodies in the Murray–Darling Basin, 52% reported that salts were causing off-farm problems in their municipalities. Victorian respondents reported the highest incidence of salt damage (58%) and Queensland the lowest (9%). It is interesting to note also that whereas in New South Wales and Victoria, few councils rated the salting problem as serious (5 and 7% respectively), in South Australia, considerably more rated the problem as serious (16%). Further, there was a general perception by councils that on-farm practices were moderate or major causes (76%) of off-farm salting and the rise in watertable height (Oliver *et al.* 1996), despite the belief by scientist that urbanisation generally results in a significant increase in off-farm groundwater recharge (Campbell 1995; Gates 1995).

5 MECHANISM OF SALTING

Salt damage to land manifests itself in numerous ways, and as an artefact of different causal factors such as irrigation, or dryland agriculture in high rainfall areas, however, the mechanism by which salting occurs is common and well known. In principle, the hydrology of the catchment has changed, leading to an increase in height of local groundwater which mobilises deep salt reserves and transports them into the root zone. The symptoms of dryland and irrigation salt damage are similar, but the mechanism by which groundwater is recharged may differ.

Dryland salting

Secondary (or man-induced) dryland salt damage occurs as either of two forms; saline seepage and saline scalds. Saline seepage is intrinsically linked to groundwater processes and these areas are currently expanding due to disturbance of the hydrologic balance of a catchment. Scalds are not necessarily linked to groundwater processes, except perhaps where salts from saline watertables rise via capillary action and concentrate at the soil surface (Evans 1990). Given this, salt seepage as an artefact of man-induced alterations in the hydrologic cycle has attracted much interest over the past decade.

Human intervention has had a major impact on the hydrology cycle of catchments, particularly in areas of high rainfall. In many of these regions in Australia, there has been a major increase in the height of groundwater and, as a result, this has led to a redistribution of salts in the regolith. The increase in height of groundwater has occurred largely due to the clearing of native vegetation, particularly trees, native grasses and other understorey species, for the purpose of primary production (Peck 1978). Many native species may not be as water-use efficient (dry matter produced per unit of water used) as their European replacements, but their deep rooting habit and the fact that they transpire throughout the year, compared with only winter and spring transpiration by introduced annuals, suggest that native ecosystems used more water than currently occurs. As a product of clearing and replacement by monocultures composed of annual species, more water now leaks from beneath the root zone, ultimately contributing to groundwater (Peck 1978).

In regions where groundwater bodies are composed of highly conductive materials such as sands or gravels, and except in circumstances where a geological barrier may restrict flow, an increase in recharge is unlikely to constitute a problem (Szabolcs 1989). However, in regions where there is some sort of restriction to downslope lateral groundwater flow, such as silts or clays, the increased inputs of water destabilise the previous groundwater equilibrium, leading to increases in the height of groundwater (Shaw 1991). This rise continues only as long as inputs exceed the groundwater bodies' ability to conduct the additional water out of the system or where an obstruction to groundwater flow exists. Once the imbalance between input and the catchment groundwater system's capacity to output water is met, groundwater heights stabilise, achieving a new equilibrum level. However, in some areas or parts of the catchment, where hydrologic restrictions exist and limit flow, groundwater levels may rise close to the soil surface, creating a discharge zone (Shaw 1991).

Where the groundwater body and associated regolith are low in salt, the major problem associated with the high groundwater is water-logging, or conditions in plants brought about by a deficit of oxygen in the root zone. However, where significant concentrations of salts exist within the groundwater body or at depth in the regolith, the rise in groundwater height mobilises these salts upward toward the root zone. Where groundwater rises to the soil surface creating discharge, salt seeps form. But more commonly, capillary rise from watertables near the soil surface conducts suspended salts into the root zone, and the evaporative loss of water within this zone further concentrates these salts.

Irrigation salting

Irrigation has been and currently is a major arena for agricultural development. The supply of irrigation water has enabled the expansion of agricultural development into semi-arid and arid areas, and the intensification of production industries which have provided a major boost to global agricultural production, and improved the food security for developing nations. The importance of irrigation to global food supply can not be understated. The World Bank estimated that the expansion in irrigation between the mid-60s and mid-80s accounted for over 50% of the commensurate increase in global food supply (World Bank-UNDP 1990). However, the onset of salt damage in irrigation areas threatens these production levels.

> *FAO estimated that by the year 2000, the amount of land devoted to irrigation on a global scale would expand to 320 million hectares, given an expansion rate of*

5 million hectares per year. However, in more recent times, the rate of development of irrigated areas has dropped to about 2 million hectares per year and coincidentally, this growth is counteracted by 2–3 million hectares of irrigation land lost to salting each year.

(Hydraulics Research 1990, cited from Umali 1993)

Like dryland salting, in irrigation areas water serves as the mobile phase, transporting salt into and out of the root zone. The balance of salt moving in the system in irrigation water and moving out in drainage water dictates whether mobile salts concentrate within the root zone or are leached from beneath the root zone. Hoffman produced an equation to describe the balance of salt at a specific location:

$$S_s = D_r C_r + D_g C_g + D_i C_i + S_m + S_f - D_d C_d - S_p - S_c$$

Where S_s represents salt balance at a particular point in time, D_r and C_r are the volume per unit area and salt concentration in rainfall, D_g and C_g are the volume and salt concentration of groundwater, D_i and C_i are the volume and salt concentration of irrigation water, S_m is the salt load in the soil or derived from weathering of soil minerals, S_f is the salt added to soil as fertiliser or soil amendments, D_d and C_d are the volume and concentration of salt in the drainage water; S_p is the quantity of salt precipitated, and S_c is the mass of salt removed in the harvested product (Hoffman 1990).

The accumulation of salt in irrigation areas occurs as an integration of the processes listed in the equation above. However, in most areas, one or more of the above processes dominate. As an example, many irrigation areas throughout the world are located in the arid or semi-arid zone and on soils in which the substratum is naturally saline. As a result, it is typical practice to irrigate in excess of plant needs to leach indigenous salt, and salt previously applied in the irrigation water from the root zone. Where the volume of water applied is less than that required to leach the soil of salt, salts accumulate in the root zone (Umali 1993). However, irrigation applied in excess of plant demands for the purpose of leaching may increase the height of the groundwater immediately under the region irrigated. This occurs particularly where hydraulic conductivity of the aquifer is low. This effect has been observed in groundwater bodies beneath most irrigation areas in south-eastern Australia, most notably the Riverine Plain (Stannard 1978; Slavich 1992) and the Macquarie Valley (Willis and Black 1996). As a consequence, many of these irrigation areas have elevated groundwater levels directly beneath the irrigation zone (e.g. the Coleambally Irrigation area; Stannard 1978) that dissipate as the distance away from the irrigation zone increases. If these water bodies have high salt loads and exist close to the soil surface, salts concentrate in the root zone via capillary action and evaporation (Umali 1993).

Urban salting

Urban salting is a relatively new phenomenon, about which we currently have very little experience. Hence quantifying sources of recharge is at this stage somewhat of an arduous task. The ABARE survey of 1996 revealed that a widely held perception with local government authorities existed, to the effect that 'groundwater rise and discharge in urban areas was due largely to on-farm land management practices' (Oliver *et al.* 1996).

There is evidence available which suggests that urban contributions to groundwater may be considerably greater than previously thought. In an examination of town water distribution, the NSW Water Supply and Sewerage Performance comparison showed that in most cases leakage of town water supply was underestimated by the local authority (Sarma *et al.* 1994). In this study, leakages from 40 water supply schemes in NSW varied from 7 to 35% of the total supply. Other sources of off-farm recharge have also been identified. In an investigation of dryland salting in the city of Wagga Wagga, sources of recharge highlighted were those from rainfall (diffuse recharge (est. 1–3% of annual rainfall) and household stormwater drainage via rubble pits), over-irrigation of house gardens and public areas, as well as leakage from water supply and sewage pipes (Hamilton 1996) (Table 2).

Table 2. Estimated contribution of recharge to the Wagga Wagga urban environment

Recharge source	Estimated volume for 1993–94 (kL)
Rainfall	
diffuse	29 750 – 89 250
rubble pits	53 120
Pipe leakage	
water supply	46 400 – 190 130
sewer	16 430 – 49 280
Irrigation	4120 – 20 600

Source: adapted from Hamilton, S. (1996), Department of Land and Water Conservation

6 COST OF SALT-DEGRADED LAND

On-farm costs in dryland environments

Assessing the cost of salinity is a difficult task, particularly considering that there are many causal factors and many ways that salt-induced damage can be interpreted by the viewing public. Some members of society frequently view the cost of salt damage in terms of lost production potential of the land affected or in damage to infrastructure. However, others view the cost of salt damage land in terms of the cost of restoration of degraded land.

The Federal Minister for the Environment estimated the current national cost of dryland salting in terms of lost agricultural production to be $243 million per annum (Hill 1997). This amount is appreciable, however, it is not the only cost associated with this form of land degradation. In addition, off-farm costs such as damage to public infrastructure (i.e. roads and bridges, increased salts in waterways, damage to existing woodlands and scattered trees, damage occurring to houses and water reticulation systems), and restorative costs of treating salt-affected land were not considered. Clearly, the annual cost of dryland salting will significantly exceed this figure.

In a paper to the Prime Minister's Science and Engineering Council on sustaining the agricultural resource base, salinity in south-western Western Australia and in the Murray–Darling Basin was noted as being one of the major forms of land degradation. The authors, Hamblin and Williams (1995), made a particularly noteworthy comment regarding the solution to salinization:

> *The hydro-geological situation in the Murray–Darling Basin is such that it is improbable that surface management of soil or vegetation will be able to contain or control saline waters in most affected areas. Acutely affected areas will be retired from agriculture. Smaller district-scale solutions are possible, by greater vegetational use of plant water via improved agricultural plant yield, and extensive use of perennials.*

In a dryland context, this quote fairly well reflects reality and suggests that current secondary salt damage, as occurs in many high rainfall areas of the Murray–Darling Basin, is a function of historic contributions to recharge. Except in small local catchments, it seems unlikely that immediate changes in vegetation management will have much effect on reducing the rate of spread of salt-affected areas, or reducing accessions to groundwater and the height of regional groundwater bodies. The cost of reclamation of these zones using drainage is frequently more expensive than is the loss of production from affected areas, and salt removed in drainage water is difficult to dispose of legally. Until we develop vegetation systems such as perennial pasture–annual crop rotations, plus well managed stocking rates or optimal plantings of native trees in critical recharge areas that can maintain the soil profile in a relatively dry state, it is most likely that we will have to learn to live with salt-affected discharge zones as a part of the landscape in agricultural catchments, located in high rainfall zones. Hence the cost of salt damage from loss of production in association with the expansion of the area affected by dryland salting will certainly increase in the foreseeable future.

On-farm costs in irrigation areas

Numerous reports have estimated the cost of irrigation-induced salting to the community. In the 1991 Victorian State of the Environment report, the costs of salt damage were reported for numerous irrigation regions (SER 1991). Current loss of farm income in the Shepparton region was considered to be $20 million and expected to almost double by 2020 if nothing is done. Similar losses in production were occurring in irrigation areas within the Loddon–Avoca region. To assist in combating induced salting, the release of the Victorian Government's Salinity Strategy Salt Action-Joint Action (Victoria Government 1988) suggested community involvement via the production of community-based management plans for the nine declared salinity regions. Within each of the plans the following matters were to be dealt with:

- an assessment of the economic, social and environmental effects of alternative salinity control measures;
- preferred measures and implementation targets;
- cost sharing procedures;
- incentives and sanctions to be used to ensure adoption of the plan;
- arrangements governing the discharge of salt to the Murray River where relevant; and
- responsibilities for the implementation and review of the plan (SER 1991).

The costs of enacting Land Management Plans in the Shepparton region has been estimated at $295 million over a 30-year period with 50% of the cost being borne by government. Although this cost appears huge, it is substantially less than the estimated accrued cost resulting from the loss of production alone that would occur if nothing were to be done and salt damage were to continue unabated at current rates of expansion.

Off-farm costs

Increased salt loads of inland streams and water bodies are an additional casualty of increasing salinisation of agricultural lands. This phenomenon is not restricted to Australia but is occurring on a global scale (Deason 1992; Umali 1993). At present, a great proportion of this increased salt load originates from the disposal of saline drainage waters from irrigation areas and has a major influence on in-stream and riparian ecological processes in the zone downstream of the inflow (Conacher and Conacher 1995). The areas most affected in this way in Australia are streams and rivers of the Murray/Murrumbidgee catchment. Increasing instream salt not only affects instream and riparian processes but influences downstream water users: irrigators, municipal and industrial users. While drainage from irrigation areas contributes a major source of salt, it is worth noting that, as time passes and as groundwater levels and groundwater-driven stream flows increase, the contribution of salt from dryland agricultural zones will increase substantially, as has occurred in south-western Western Australia (Conacher and Conacher 1995). The effect of this will be a further stress on remaining native terrestrial ecosystems (Ghassemi *et al.* 1995), causing tree decline and contributing further to loss of biodiversity (Powell 1993). It is interesting to note, none the less, that in the wetter areas of south-western Western Australia under current land use salts are expected to be flushed out within decades, but in dry areas it will take centuries under current land-use practices (Peck and Hurle 1973; Peck 1978; Peck and Williamson 1987).

Cost of urban salting

Although urban salt damage is a relatively new phenomenon, salt-induced damage in these environments is more easily identified and therefore tangible costs associated with this damage are more easily determined. These costs are manifest as damage to infrastructure such as road surfaces, buildings and utility supplies. However, it is worth considering that intangible costs of urban areas also are substantial. These include declining real estate prices and loss of market and flexibility of the affected landowner. These costs are rarely included in any estimate of damage. The estimated total repair expenditure by local councils in the Murray–Darling Basin during the period 1994–95 was of the order of $8.5 million, with roads and bridges accounting for 86% of that amount (Oliver *et al.* 1996).

7 ROLE OF VEGETATION IN MANAGING GROUNDWATER RECHARGE

In previous sections of this review, the association between clearing native vegetation and the increase in incidence of secondary salting in dryland areas has been explicit. Concordant with this association, it is implicit then that re-establishment of trees and other deep-rooted perennial plant species is essential if secondary salinisation is to be reversed (Johnston 1993). This being true, then several issues need to be considered, some of which will be briefly discussed here:

- importance of episodic recharge versus annual recharge to groundwater;
- effectiveness of native versus exotic vegetation in reducing recharge;
- spatial nature of recharge, locating areas of maximal and nominal recharge and discharge zones;
- where and how much land needs to be revegetated;
- should farms which occupy major recharge or discharge areas be retired from production; and
- economic assessment of viability of new vegetation management programs.

Since the late 1980s, government authorities have advocated restoration of some elements of original vegetation as a management strategy to attempt control of dryland salinisation (Government of Victoria 1988; Clifton *et al.* 1993). Success in achieving this aim depends on the successful integration of two strategies: recharge control and vegetation management of the affected site. In marginal agricultural land, such as hill tops where shallow soils overlie fractured rock and where groundwater recharge processes predominate, strategic plantations of selected tree species may reduce annual recharge. This strategy has been shown to be effective in arresting recharge and within relatively short time periods (i.e. 1–2 decades) reversing the apparent rise in local groundwater heights (Schofield and Bari 1991; Clifton *et al.* 1993). In addition, the time taken to affect groundwater levels may be reduced by increasing planting density (Clifton *et al.* 1993). However, this approach may not be economically viable for controlling recharge in arable areas. In more arable zones, strategic use of exotic agricultural species intermixed with agroforestry may prove effective in controlling recharge, as was shown by Carbon *et al.* (1982) in the Swan Coastal Plains of Western Australia. Their study showed perennial pastures exhibited similar seasonal patterns of water use and could use similar amounts of soil-water as forests indigenous to the area; and 14-year-old *Pinus pinaster* (Alt) plantations transpired more water than the native forests they replaced, reducing deep drainage beyond 6 m (Carbon *et al.* 1982). The findings of this study suggest that strategic use of perennial pastures and woodlots may control annual contributions of water to local groundwater systems. However, in some areas where episodic recharge predominates, establishing tree plantations may be the only means to control recharge in these areas (Nulsen 1993).

Similar to recharge zones, vegetation management is important for groundwater management in discharge zones and in maintaining a vegetative cover to reduce the risk of salt scalds developing. However, where trees in discharge zones use saline groundwater, there is a risk of salt accumulating in the root zone (Kennett-Smith *et al.* 1993); consequently, trees extract less water, further reducing transpiration (Passioura *et al.* 1992). In situations where these salts are not periodically leached, the increased salt load in the root zone may cause trees and other vegetation to die, leaving the area susceptible to erosion.

However, although there is a public perception that tree plantations are the solution to dryland salinisation, widescale adoption of revegetation of the landscape by trees is low. Only the minority of Australian farmers plant trees (20%). Highest proportions of farmers planting trees occurred in the state of Victoria (60%) followed by Western Australia (50%), South Australia (22%), New South Wales (9%) and Queensland (4%) (Prinsley 1991). The reasons given for such low levels of adoption included:

- financial reasons including establishment costs, non-economic benefits, maintenance costs;
- technical reasons including lack of knowledge;
- competition with agricultural interests; and
- lack of interest.

Dumsday *et al.* (1989) concur and suggest that broadscale revegetation is not economically viable as a strategy for groundwater management. They suggested that two major obstacles exist to prevent widescale adoption of revegetation with trees: low commercial viability of properties, and risk in long-term investment in forestry. Hence farmers may be more inclined to revegetate some areas of their properties in a manner consistent with agroforestry practices than to adopt major reforestation programs. Nonetheless, given the low rates of adoption so far of revegetation, yet the seeming necessity for revegetation, community and education programs are likely to be needed if the adoption rate is to increase.

An alternative strategy that may achieve more acceptance by farmers involves the use of perennial pastures composed of deep rooted, summer-active species in parts of the landscape where soils have a reasonable capacity to store soil moisture. Studies have been conducted which show that replacement of shallow-rooted annual species by perennial species can be effective in reducing accession to groundwater (Crawford and Ransom 1993) and secondary salinisation (Halvorson and Reule 1980). This happens as during winter, rainfall in excess of plant requirements can be stored in the soil to be used later in the season by summer-active species when evaporative demand is higher. Also the deeper rooting habit of perennial species such as phalaris and lucerne allows them to extract water from a greater depth than annual pasture species (Beale 1993; Crawford and Ransom 1993; Taylor and Clifton 1993) and therefore effectively increase the soil's capacitance for storing soil moisture. However, in areas where recharge is episodic and via prefered pathways rather than through the soil matrix, recharge control is most likely to be via deep-rooted perennials with a high water use potential, namely trees (Nulsen 1993).

8 COST OF MANAGING SALT-AFFECTED LAND: WHO PAYS?

As discussed in the previous section, the solution to salt-induced land degradation is well acknowledged and quite simple, but given the current socio-political climate, almost insurmountably difficult to implement. A major constraint to management is the issue of cost; who pays for revegetation, and who compensates the landowner? The producer? The government? The consumer? Should some combination of all three be accountable? A further question relevant when the issue of cost and who pays is being considered: 'Is the cost of prevention greater than the cost of the problem?'

Agricultural salting

To explore this area of the debate, the status quo regarding Australian farming systems and previously government-endorsed initiatives needs to be considered in an attempt to ascertain who pays.

A major problem regarding management of dryland salting is the spatial nature of the problem. Recharge and discharge are typically geographically remote events and occur within a catchment, often on land owned by different land owners. Should one land owner curb their agricultural practice, such as annual cropping, and replace it with a lower potential value enterprise, such as perennial pastures, for the benefit of a neighbour? If so, where the substitute enterprise gross income is less than the former enterprise, should the beneficiary pay reparation to the donor farmer? In addition, and complicating this thesis, are commodity prices received by farmers adequate to allow them to farm sustainably and to pay neighbours to forego production? Although these questions may be too difficult to answer, the debate about the environmental impact of agriculture is largely dictated by these issues.

It is often argued that agricultural producers have caused the problem and therefore should be responsible for rectification and shoulder the cost. This argument is only in part correct. It is probably more correct to argue that in the pursuit of increasing food production for the benefit of society, land was historically cleared of native perennial vegetation, frequently with some form of government support (e.g. subsidies, tax relief, soldier settlement projects), and replaced by agricultural species. Although the intentions of agricultural producers regarding land clearing were not entirely philanthropic, land clearing was frequently supported by the government and goods produced satisfied the needs of consumers. Hence the needs of consumers have necessitated development. As a product of development, the additional water that has deep percolated as a result of clearing and replacement by annual crops and pastures has progressively contributed to groundwater levels, causing levels to rise. Now, deep reserves of salt have been mobilised toward the surface and have affected events on the soil surface. Is it realistic for present day farmers to be held responsible and expected alone to pay for historical damage from which society as a whole has been a major beneficiary?

Present day farmers have little latitude in affording further production cost increases while commodity prices are maintained. Over time, the real price received for agricultural commodities has decreased, as reflected in the almost linear decline in farmers' terms of trade over the past 30 years (Cribb 1989). To accommodate declining real farm-gate prices, agricultural producers have had to become more efficient or have had to take a cut in real income. In response, Australian farmers have become more efficient; for example, wheat yield per hectare sown has consistently improved over the past century (Fischer 1997). These improvements in operating efficiency have come about by improvements in farm management techniques such as the timing and application of fertiliser to crops, and the move from grazing to cropping industries. In addition, to make economies of scale, the general size of farms has increased. However, at the same time, inputs such as fuel and fertilisers have increased steadily, reducing the net income earned by farmers. At present the cost price squeeze on most agricultural produce is such that further production costs cannot be easily met.

A similar situation occurs in irrigated agriculture. Irrigation water in Australia is considered cheap by international standards; many proponents recommend increasing the cost of irrigation water to force irrigators to become more efficient and to apply less water. Experience from the United States suggests that this approach is unlikely to reduce water use but instead force growers to switch to higher value crops (Conroy 1995). In a paper discussing water market reform, various scenarios were used to study the impact of increasing the price of water on the gross margin for irrigated rice production. This study showed that given 1995–96 water prices ($13 dollars per megalitre), profitability of rice was estimated to be $924 per hectare, but increasing the cost of water to $75 per megalitre reduced the profitability of rice to zero (Samaranayaka *et al.* 1997). This implied that the rice industry could sustain only minor changes in water pricing before growers would cease growing rice.

The problem of the reduction in farm-gate prices is complicated by the extent of international trade in agricultural commodities. Australia's free trade policy along with improved transportation have led to extensive international trade in these commodities. However, in the global market place, Australian producers are not influential in determining price. This is in part due to the lack of domination in supply to these markets, and competition with subsidised produce from competitors (Douglas *et al.* 1995), which in effect forces the commodity price down. Australian consumers too are the beneficiaries of international trade. For example, where local produce is priced too high, consumers can buy from an overseas competitor, effectively forcing the price of locally produced goods down.

Hence the international nature of agricultural trade tends to buffer the prices received by local producers, and in an effort to maintain cash flow, there are times when the prices producers receive may be below their production costs. Hence, given the commodity price squeeze, Australian farmers have little latitude to change farming practices to more environmentally benign production methods, unless production methods are at least as profitable as the practices they replace. To the contrary, Australian consumers are the beneficiaries of cheap food, but implicit in this indulgence is the downside that this cheap food is produced at a cost to the natural resource base.

Hence it is probably impracticable to expect that farmers alone can effect a change in trends in dryland-induced salt damage. Given this, it could be successfully argued that risk and cost of salt damage to land should be shared between the three major stake holders: the government, consumers, and land holders. Realising this quandary the Australian government has attempted to assist by way of encouraging community involvement in treating salt-affected land. In 1988 the national program Landcare was developed from discussions between the National Farmers' Federation and the Australian Conservation Foundation (NSW Landcare Working Group 1992). The aim of this initiative was to achieve the vision of sustainable land use by educating community groups and acting upon projects to arrest land degradation. A unique aspect of this initiative was its community involvement which gave the community some ownership of the problem and the solution, and that it was endorsed and in part supported financially by Federal and State governments. Since its inception, the Landcare initiative has evolved in an almost exponential fashion and now integrates closely with Total Catchment Management (TCM) (NSW Landcare Working Group 1992). Cooperation and coordination has allowed for surveys of catchments and the development of community Land and Water Management Plans to be achieved so that plans to more sustainably manage the land resource may evolve. This approach provides something of a cost sharing initiative for land degradation, but of particular note salinity, between agriculturalists, the community and government.

However, direct contribution by government is only part of the equation. As consumers are a major beneficiary of agricultural production and sustaining the agricultural resource base is within the interest of consumers, then the price paid for commodities must reflect at least the real cost of production. However, given the current environment where due to the manipulation of commodity prices by competitors, the true costs of production are not achieved in the market place, then government intervention is required to ensure that the minimum real price paid reflects production costs. Since the early 1990s there have been several calls for the introduction of an environmental levy on food so that everyone shares the cost of protecting our basic food-producing resources (Meyer 1992; Conroy 1995; Douglas *et al.* 1995). In a Research Strategy Position Paper (Meyer 1992), and more recently in popular science publications (Conroy 1995) and the popular press (Meyer 1995), Meyer has argued that the introduction of an environmental levy on food could be used to support or reward restorative work, upgrade irrigation infrastructure, be used to support efficient resource use and to fund research, education and training programs. Coincidentally, in a paper to the Prime Minister's Science and Engineering Council on sustaining the agricultural resource base, Douglas *et al.* (1995) made a recommendation that 'consideration should be given to possible publicly supported funding options for natural resource management'. In this document, Douglas *et al.* suggested that a 3% resource management levy on the wholesales of all foods and beverages would raise $300 million per year (estimated on 1991–92 estimates of sale). However, although this approach would cover domestic as well as imported produce, it may have little impact on the effect of subsidised imports. Perhaps instead a combination of environmental levy on local produce as well as a floating environmental tariff on imported produce may be more effective. A tariff of this type may be imposed against imported produce whose production is directly or indirectly subsidised, or where agricultural commodities are produced in an environmentally degrading manner, in an attempt to neutralise the effect of subsidies. Using this sliding levy, the level of tariff imposed would be zero when applied to non-subsidised commodities produced in an environmentally benign manner, but increase proportionally as the level of subsidy or the degree of environmental damage incurred during production increased. The effect of this may be a more even playing field for Australian producers, allowing them to produce in a more environmentally passive manner and to be more adoptive of new technology. However, the introduction of a new tax is normally distasteful to the public and the role for the government would be to market this concept not as a tax but as a contribution to the sustainability of our natural resource base.

Urban salting

Like on-farm or rural salting, salting in urban catchments is also caused by disturbance of that water balance. However, management and the partitioning of cost in the urban environment is markedly different to other environments due to different land usages and urban community expectations (Hamilton 1996). As the damage of salting in these environments is very expensive, solutions such as groundwater pumping may in the short term be cost effective. However, the long-term solution is a reduction in recharge, and to achieve this goal the local community needs to understand and accept the limitations of the natural catchment in accommodating urbanisation (Hamilton 1996).

The options for control of recharge and the associated cost fall to the local authorities and members of the community, but the success of the program will depend on how well the community unites and develops an integrated strategy for control. Past experience with similar environmental issues has shown that education of community members is pivotal to ensure the wide-scale adoption of salinity management plans (Hamilton 1996).

9 CONCLUSIONS

Salt-induced land degradation, whether on-farm or off-farm, is a major problem in southern Australia. The amount of land affected by salt is expected to expand into the future. Engineering solutions are available to solve the symptoms and some eco-physiological options are available to control recharge, however, the catchment nature of dryland salting causes the solution to move beyond the technical into the socio-political arena. In addition, as reported by Hamblin and Williams (1995), it is unlikely that surface management of soil or vegetation will be able to contain or control saline waters in most affected areas. Hence, under current land practices, we will have to accept that some land will be sacrificed to induced salt damage.

However, to prevent the situation from becoming much worse, there are three major hurdles to be overcome for effective salt management to be realised:

- recognition of the problem by stakeholders and admission of ownership of the problem;
- achieving workable cooperation between landowners within a catchment; and
- agreement on strategies, including optimisation of vegetation restoration works and the location of vegetation in the landscape.

Major gains have been realised in achieving cooperation between landowners. This has in part occurred as a result of government intervention through Landcare style initiatives, particularly education of the community and encouragement of community action. However, the first hurdle is probably more formidable: the recognition of the problem by stakeholders and the acceptance of ownership by stakeholders. In essence all consumers are stakeholders as they depend for their future on the production capability of the land. Encouraging exploitation of the environment by demanding good quality produce at a price beneath the real cost of production is unsustainable. These stakeholders must be educated as to the effect of their actions on the environment and to accept some ownership and, in doing so, be willing to meet the real cost of production, otherwise further degradation of the natural resource base will be inevitable.

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Soil Acidification — An Agricultural and Environmental Problem

Peter Cregan and Brendan Scott

Soil acidity is both a serious agricultural and an environmental problem and affects in excess of 35 million hectares of agricultural land. Acidification is ongoing with rates of acidification, expressed as lime equivalence, varying from near zero in unfertilised low fertility systems to 150 kg ha^{-1} yr^{-1} in high rainfall areas, with rates as high as 1710 kg ha^{-1} yr^{-1} in intensive agriculture. This is driven by the N and C cycles operating through nitrate leaching and product removal. Some fertilisers also acidify. Acid soil adversely affects plants through aluminium and manganese toxicities, hydrogen ion (H^+) toxicity, and deficiencies of molybdenum, calcium, magnesium and phosphorus. Changes in biology of the soil also occur as pH declines, as most organisms have an optimum pH range for growth. A reduction in pH can change the spectrum of diseases and their pathogenicity. The major management strategy in agriculture is lime application, but this appears to be economically unattractive in many situations. For farming/grazing land on the south-western slopes of NSW, lime application alone may represent 1.5 to 3 times the value of the land. Most of the acidification which has affected subsoils (>20 cm depth) cannot be treated profitably.

Soil acidity has environmental consequences. The nature of off-site effects of soil acidity is not well documented nor accepted. We believe that off-site effects of soil acidity are driven by the under-utilisation of water in situ. This excess water can erode surface soil, causing turbidity in streams and siltation of river systems, as well as phosphorus contamination. Excess water lost down the soil profile carries nitrate and contaminates groundwater in addition to raising the watertable to produce waterlogging or salinisation down slope.