Chapter 1

Introduction

1.1. Contamination of inland water systems

Water is a most important part of life. Contamination of inland water systems has become a severe problem and has attracted increasing concern from governments, research institutes and communities. Water bodies are polluted by both point and non-point (or diffuse) sources. Australian communities discharge about 4500 megalitres of treated municipal effluent into rivers and oceans each day (Bennet 1993). The discharge constitutes the largest point source of nutrients, particularly nitrogen and phosphorus, entering these water bodies. Other point contamination sources include industrial effluent, fish farming and aquaculture, feedlots and piggeries, urban stormwater etc. Non-point source pollutants are carried mainly in surface runoff from land and sometimes in subsurface flows into water bodies. Non-point sources may be very significant in their impact on water quality (Cullen 1991).

Excess nutrients entering groundwater and waterways have been implicated in health risks and water pollution. Australian standards for nitrate-nitrogen in drinking water are 10 mg L^{-1}, set by the World Health Organization (Phillips and Grant 1994). Nutrient pollution in waterways has resulted in toxic blue-green algal blooms, mostly during the warmer months of summer and autumn. Massive blooms occurred over 1000 km of the Darling River and in other waterways during the summer of 1991-92

(Bennet 1993). While nutrients are not the sole cause of algal blooms they have promoted them. Phosphorus and nitrogen are the most important nutrients for algal growth and phosphorus is considered to be the limiting nutrient for blue-green algae (Lawrence 1993).

1.2. Non-point source contamination

Non-point source contamination has received increasing attention in recent years. Non-point sources are often less obvious than point sources, because they are much more variable, occur over wider areas and generally at lower concentrations (Cullen 1991). However, non-point source pollution is one of the greatest forms of contamination affecting the quality of inland water systems. For example, in the United States, agricultural-derived contaminants constitute the single, largest diffuse source of water quality degradation (U.S. Department of Agriculture 1985, cited by Osborne and Kovacic 1993). In Australia, there is also considerable evidence of significant degradation of waterways related to runoff from rural and urban lands (Cullen 1991).

The important pollutants from non-point sources include: 1) phosphorus and nitrogen from fertilisers and animal excreta, 2) sediment from soil erosion, 3) organic matter from plant material and animal wastes, 4) agricultural chemicals and their breakdown products, 5) microbial contaminants from sewage and animal wastes, and 6) heavy metals and salts from road runoff. These pollutants can dramatically degrade the quality of the receiving waters if excessive loads enter aquatic systems.

Unlike point source pollution control programs, there is no single strategy that is effective in restoring water quality conditions in waterways suffering from non-point impacts. Waterway pollution from non-point source contaminants occurs because human activities have changed the structure of the landscape and increased the quantity of substances (such as nitrogen and phosphorus) in the catchments, thereby altering the rate at which natural processes operate and the ways in which energy, water and matter cycles in the system. Therefore, catchment management strategies must promote best management practices. In this way, natural physical and biological processes can be incorporated to filter, convert, or store pollutants on the land before they move to the aquatic systems (Petersen *et al.* 1992). Such a bio-assimilation strategy is an ecologically sound, sustainable, and cost-effective approach for controlling water quality conditions in waterways involving high proportions of non-point pollution.

1.3. Vegetation-soil systems as an alternative to dealing with point and non-point source pollutants

Both point and non-point source pollutants constitute a threat to our water systems. Pollution from point sources, such as municipal or industrial wastewater is obvious and attracts most attention of the public. Consequently, many technologies have been developed and used for wastewater treatment. However, present day wastewater treatment technologies have grown increasingly complex with the requirement of relatively sophisticated and expensive plants (Droste 1997). In addition to capital cost, considerable outlay is required for operational and maintenance expenses. Therefore, over the past three decades, natural treatment systems, particularly vegetation-soil systems have increasingly been promoted as a viable alternative to the conventional, more engineering-oriented ways of treating wastewater.

Most forms of natural treatment systems are generally preceded by some form of mechanical pretreatment. A minimum of fine screening or primary sedimentation is necessary to remove gross solids that can clog distribution systems and lead to nuisance conditions. The objectives of a natural treatment system will determine whether additional pre-treatment is required. Quite often, a secondary (biological) treatment is required for treatment of wastewater by using a vegetation-soil system. The most commonly used method of recycling secondary-treated effluent in Australia has been flood and spray irrigation of pastures that have been grazed by sheep and cattle (Stewart *et al.* 1986). Many of these schemes have been developed for inland regions because, (1) reusing wastewater can increase pasture product, and (2) there has been ample and relatively cheap land available for disposal of effluent by irrigation (Johnson 1984). However, effluent irrigation of tree plantations is becoming more popular in recent years, mainly because of possible higher rates of growth and water use by trees over other agricultural crops and pastures, and because they produce products that do not enter the human food chain (Bennett 1993; Boardman 1991). Many studies in the United States (Urie 1986) and in Australia (Stewart *et al.* 1986; Myers *et al.* 1996) have found that wastewater irrigation of forests is an economical and safe sewage effluent reuse method.

Control of non-point pollutants is difficult and little progress has been made so far. The basic strategy for managing a catchment to minimise non-point pollution is to develop best management practices (BMPs) that limit contaminants migrating from their source (Cullen 1991; Bosch *et al.* 1994). Although BMPs provide solutions to many problems, during high intensity and high volume rainfall events, surface runoff, soil erosion and deep percolation can still result and produce adverse environmental impacts, even with BMPs in place. Therefore, retained buffers that can slow and sometimes prevent movement of contaminants from agricultural catchments have recently received renewed interest. Vegetated buffer zones have been recommended as important techniques to restrict contaminant migration. Vegetation may include trees, grasses and wetland plants. Many studies have demonstrated that these buffers play important roles in filtering, reducing and storing agricultural pollutants (e.g. Delgado *et al.* 1995; Dillaha *et al.* 1988; Lowrance *et al.* 1985; Osborne and Kovacic 1993).

1.4. Potential pollution associated with feedlot effluent in Australia

Australian lot-feeding of cattle expanded dramatically between 1988 and 1994, although the industry started in the 70s. Official survey figures in 1996 recorded 867 feedlots in Australia with a one-time capacity of over 800,000 head (Meat Research Corporation 1996). As cattle are fed for average periods of only about 120 days, the potential throughout of Australian feedlots is, therefore, much higher than the one-time capacity. This will depend on utilization or occupancy rates which varies with market conditions, grain supply, cattle supply and profitability. For example, in 1994, the estimated occupancy was 73%, while in September 1996 the occupancy was down to 39%. These changing rates reflect the difficulties in the Australian beef industry resulting from high feed-grain prices and intensive competition from the USA in the Japanese beef market. Despite the lower feedlot occupancy, it was estimated that 1.2 million head of cattle passed through registered feedlots in 1996.

Feedlot operations have attracted considerable attention because of their potential to cause environmental degradation, despite the lack of strong evidence that feedlots are a major source of pollution [Senate Standing Committee on Rural and Regional Affairs (SSCRRA) 1992]. The principal threats arise from the intensive nature of the production system, the generation of large volume of waste and the problems related to waste management. Feedlots generate significant amounts of wastes that are a source of phosphorus, nitrogen and salts. Liquid wastes include urine, wash waters and rainfall runoff from cattle pens. Solid wastes comprise livestock faeces and spilled feed. Feedlots concentrate large amounts of nutrients and water into a relatively small land area. On the one hand, these liquid and solid wastes provide a ready source of fertilizer if they are properly managed; while on the other hand, they comprise a significant potential pollution source to groundwater and surface water systems.

Government agencies with statutory responsibilities and industry associations in Australia have formulated a series of codes of practices to control the impact of feedlots on the environment. For instance, The Australian Lot-feeders Association formulated The Code of Practice for Protection of the Environment. The code provides detailed information in relation to management practices necessary to maintain good standards of environmental performance. In particular, current environmental approval requires that feedlot waste be disposed of by land application in such a way that they are used on site and do not contaminate river and groundwater systems.

1.5. Study aims

The Cooperative Research Centre for Meat Quality developed a beef cattle research facility at Tullimba, located 50 km west of Armidale, NSW, Australia (Figure 1.1). The facility comprises feedlot and infrastructure capable of holding up to 1000 head of cattle. The facilities are used for research into meat science, cattle breeding, animal health issues, and feed technology as well as research into feedlot waste recycling, environmental monitoring, and the development of feedlot waste management practices. To meet the environmental requirements of the NSW Environmental Protection Authority, a 20 ha liquid waste disposal area was established for the disposal of liquid effluent from the feedlot by spray-irrigation. The area is also licensed for disposal of some solid waste. The experimental site for this study into the use of vegetated buffer strip for managing non-point sources pollution was established on the lower slope of the liquid effluent disposal area (Figure 1.2).

The overall objective of this study is to address the question: how effective are vegetated buffer strips of tree and improved pastures in intercepting lateral groundwater movement and nutrients from effluent disposal area? This question is significant for the Tullimba feedlot where the soil is strongly duplex in character with highly significant differences in hydraulic conductivity between the A and B horizons. The specific aims are (1) to examine the survival and growth of *Eucalyptus camaldulensis* Dehnh. and *Casuarina cunninghamiana* Mq. trees and improved pastures in the buffer strip irrigated periodically with feedlot effluent; (2) to determine the water use of the trees and the improved pasture, and (3) to determine the effectiveness of trees and pastures in intercepting lateral groundwater flow and nutrients, particularly nitrate-nitrogen.

The field experiment was supported with a glasshouse simulation experiment. The main objective of the glasshouse experiment was to examine the effectiveness of *E. camaldulensis* and *C. cunninghamiana* and pasture removing nitrate-nitrogen from nutrient solution released in the soil A-B interface. In the experiment, the growth, nutrient (mainly nitrate nitrogen) removal and water use by trees and pasture were investigated.

1.6. Thesis outlines

The background and literature review for this thesis is presented in Chapter 2. The literature review emphasises buffer zones, including the terminology and prime function of buffer zones and describes the mechanisms, purpose, implementation and management of vegetated buffer strips (VBSs). The status, practice and study of vegetated buffer strips in Australia are also reviewed. Chapter 3 describes the study design and experimental methods for this project including both the field and glasshouse experiments. Chapter 4 presents the results and discussion on the growth and water use of the trees (*E. camaldulensis* and *C. cunninghamiana*) and improved pastures. Chapter 5 examines the effectiveness of trees and improved pastures at intercepting lateral groundwater flow and in removing nutrients under field conditions. Chapter 6 presents the results and discussion of the glasshouse experiment. The general discussion integrating the key point from the previous chapters and conclusions are presented in Chapter 7.

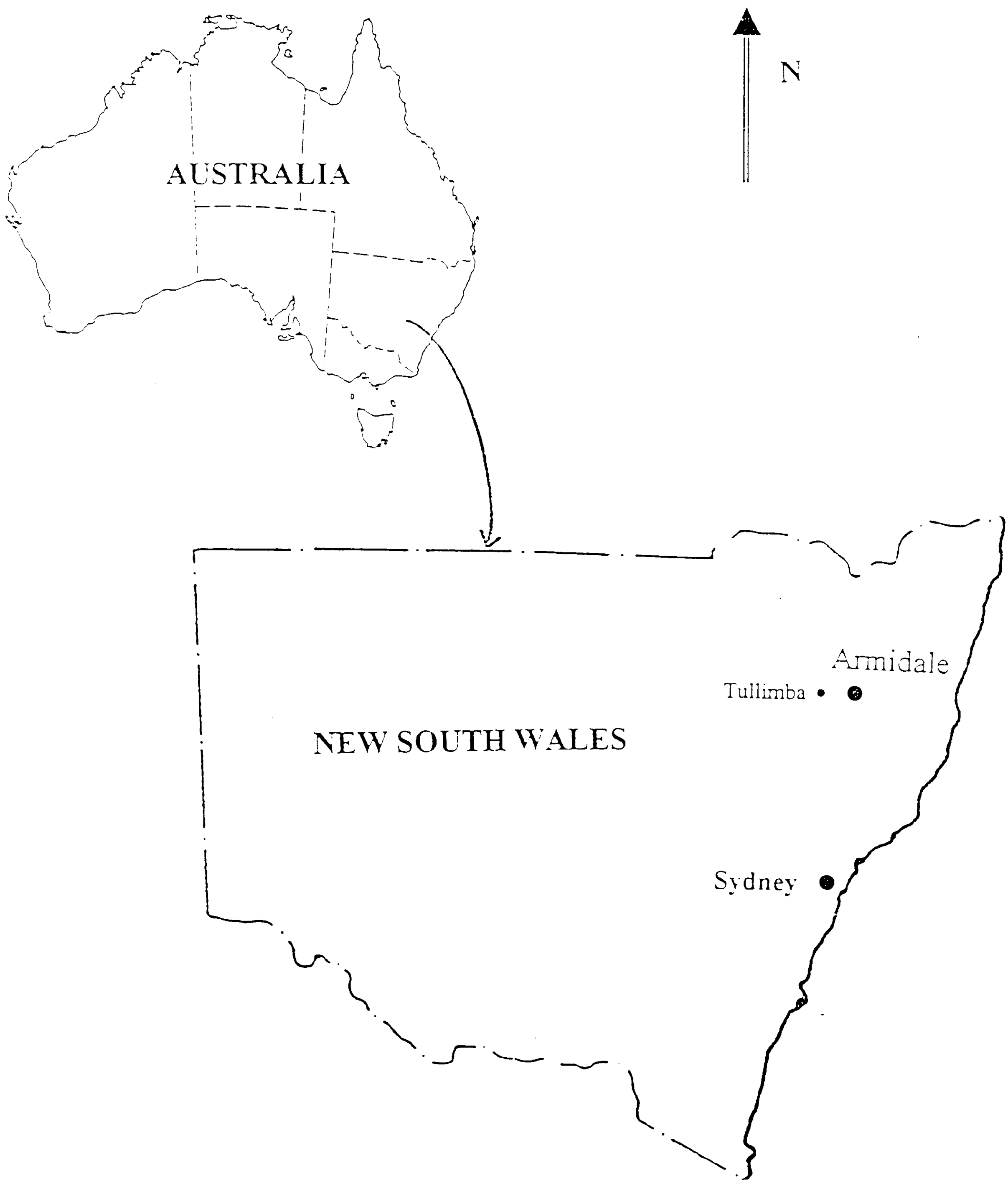


Figure 1.1. The location of Tullimba, west of Armidale, NSW.

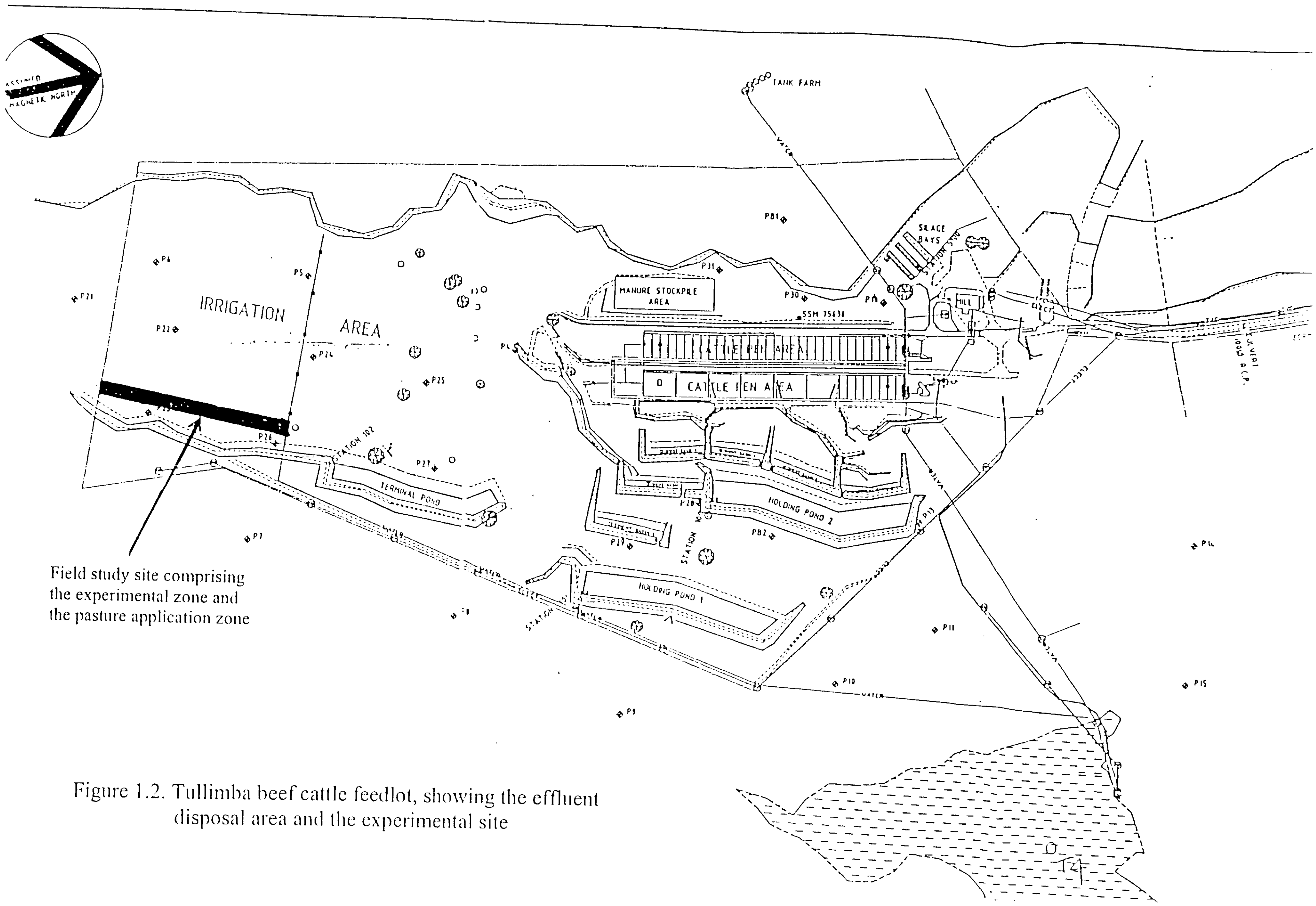


Figure 1.2. Tullimba beef cattle feedlot, showing the effluent disposal area and the experimental site

Chapter 2

Literature review and background

2.1. Introduction

The study and practice of buffer zones for protecting water quality from non-point (diffuse) source pollution has increased over the past 2-3 decades. Detailed research has been carried out for over 10 years. According to Correll's report (1996), over 400 relevant papers were published by 1996 and the rate of publication was about 30-35 papers per year. The International Conference on Buffer Zones (Buffer Zones: their processes and potential in water protection), which was held in UK in 1996 (Haycock *et al.* 1996), indicated new milestones in this research field. More and more countries around the world have initiated research programs and now use various buffer zones to control non-point pollution for water quality protection. The fact that 21 nationalities were presented at the above mentioned conference illustrates the level of global interest in this technology. Although the information is still more qualitative than quantitative, vegetated buffer zones have widely been accepted as part of Best Management Practice (BMP) for conservation of soil and water.

2.2. The terminology and prime function of buffer zones

What are buffer zones? This is a basic question for further understanding of their mechanisms (processes) and potential use. Many similar and even synonymous terms to 'buffer zones' are used throughout the literature. Some key words include 'buffer', 'riparian', 'strip' and 'filter'. Researchers use these words in various combinations, forming terms such as 'buffer zones' (eg Haycock *et al.* 1996; Muscutt *et al.* 1993), 'buffer strips' (eg Barling and Moore 1994; Brison 1993), 'riparian zones' (eg Gregory *et al.* 1991; Lowrance *et al.* 1985), 'filter strips' (eg Dillaha *et al.* 1986; Foster 1996) etc. Using a range of different terms has created some confusion to the point where the terminology needs to be clarified (Addiscott 1996). In most cases, the words 'buffer' and 'filter' describe similar functions, namely to reduce the impacts of both diffuse and point source pollution on waterbodies. The words 'zones' and 'strips' are used synonymously to describe the area where the function is performed.

The term 'buffer zones' has different meanings in different research and practical fields and for different people. For example, in forest operations Clinik (1985) defined a buffer zone (strip) as 'a stream protection zone that comprises the land and riparian vegetation which remains undisturbed during and following forest harvesting'. To cope with non-point source pollution to water sources, Phillips (1989a) viewed 'buffer zones as vegetated strips of land separating runoff and pollutant contributing areas from surface waters'. Some authors view buffer zones as wetlands and floodplains that delay and purify water adjacent to rivers and streams (Brison 1993; Petersen *et al.* 1992), while others consider them as riparian zones or stream riparian zones that protect bank and channel stability and have primary roles in determining the structure and function of stream and river ecosystems (Lowrance *et al.* 1985; Osborne and Kovacic 1993; Schultz *et al.* 1995). Fleischer *et al.* (1996) reported the potential role of ponds as buffer zones in Sweden. A functional definition of buffer zones has been given by Gregory *et al.* (1991) to be an area of direct interaction between terrestrial and aquatic systems involving exchanges of energy and matter.

Buffer zones, however, have recently and more regularly referred to as 'vegetative filter strips' (VFS) or 'vegetated filter strips', which are defined as areas to be managed to reduce the amount of pollution suspended in runoff (Foster 1996). A working definition of 'vegetated filter strips' is defined as bands or areas of vegetation designed for the removal of sediments, organic matter, nutrients, agrochemicals, and bacteria from runoff and wastewater (USDA-SCS 1991, cited by Delgado *et al.* 1995). A VFS is most often installed downslope of areas from point source (e.g. feedlot) or non-point source (e.g. cropland) pollution that could impair off-site water quality. However, it need not necessarily be located in a riparian zone.

Even at the International Conference on Buffer Zones in1996, no clear definition for buffer zones was provided. For instance, Dillaha and Inamdar (1996) defined buffer zones as 'bands of planted or indigenous vegetation situated between pollutant source areas and ephemeral and perennial streams and other waterbodies. The editors (Haycock *et al.* 1996) of the conference proceedings did not give a clear definition either, basically describing 'buffer zones' to be 'a range of habitats' within a catchment which can 'buffer the impacts of both diffuse and point source pollution' on waterbodies. This description is function-based (ie reducing the impacts of both diffuse and point source pollution and protect water quality) and includes all the habitats as buffer zones, such as ponds, wetlands, floodplain, wet grasslands, riparian woodland and forests, hedgerows, shelterbelts etc. However, in most cases, a buffer zone is featured as some vegetated area, which is, just as Muscutt *et al.* (1993)

described, 'a permanently vegetated area of land perhaps 5 - 100 m in width, most likely but not exclusively adjacent to a watercourse and managed separately from the rest of a field or catchment'. Vegetation may include trees, grasses and wetland plants.

In summary, buffer zones can be categorised into the following three groups in terms of their habitats or location in landscape: 1) terrestrial types, which are often located close to waterbodies (such as riparian vegetated buffer zones) or on lower slope (such as forest or grass buffer strips); 2) terrestrial-aquatic types, which are adjacent to water bodies, such as floodplains, wetlands; 3) aquatic types, which include a waterbody such as detention ponds and lagoons. Buffer zones can also be categorised into two groups: natural and artificial system. Quite often, natural systems (eg forests, grasslands, floodplain etc.) are selected and managed as buffer zones, although environmental scientists and government agencies are increasingly encouraging land holders to establish or restore buffer zones, because many natural buffer systems have been dramatically damaged, destroyed or lost their functional efficiency (Dickson and Schaeffer 1996; Downes *et al.* 1996; Riddell-Black *et al.* 1996).

From the above review on the definition of buffer zones, it is apparent that the prime function of buffer zones is to protect water quality through reducing sediment, nutrients (mainly nitrogen and phosphorus), pesticides and pathogen mainly associated with diffuse pollution sources (Dillaha and Inamdar 1996; Haycock *et al.* 1996), although they may perform other functions such as channel stability, provision of habitat for wildlife (Barling and Moore 1994), producing other products (eg timber, hay, fruits) etc. (Gillespie *et al.* 1995). Whether they are ponds, wetlands, wet grasslands, riparian woodlands or shelterbelts, buffer zones can reduce the connection between the potential pollution sources and the receiving environments and may provide a biological and physical barrier against pollution inputs from sources located further from the environments they are intended to protect.

2.3. Mechanisms and potential role of vegetated buffer strips in removing pollutants

2.3.1. General principles of VBSs in removing pollutants

Diffuse source pollution is spatially ill-defined with the principal pollutants being nitrogen, phosphorus, pesticides and sediments (Haycock *et al.* 1996). These pollutants are delivered to the stream environment mainly through two pathways: surface and subsurface transport. Use of VBSs as a means of removing sediment and other pollutants from diffuse point sources is a relatively new practice. Conventionally, sediment control efforts reduced off-site pollution through reducing upland erosion and on-site surface runoff. VBSs, on the other hand, are intended to remove sediment and other pollutants from runoff once it has left the upland area.

Although the detailed mechanisms of pollutant removal by VBSs are still not clear, the basic processes have been documented by many researchers (e.g. Barling and Moore 1994; Dillaha *et al.* 1988; Delgado *et al.* 1992; Hill 1996). VBSs remove pollutants associated with surface runoff and subsurface lateral flow through changes in flow hydraulics that enhance the opportunity for runoff and pollutants to infiltrate into the soil profile, through deposition of total suspended solids (TSS), through filtration of suspended sediment by vegetation and adsorption on soil and plant surfaces, and through the assimilation by plants (Dillaha *et al.* 1998).

An important removal mechanism affecting buffer strip performance is infiltration, because dissolved pollutants from surface runoff enter the soil profile in the vegetated zones as infiltration takes place. The pollutants, particularly N and P, are trapped by a combination of physical, chemical and biological processes when they enter the soil. Infiltration can also reduce surface runoff, which in turn reduces the capacity of runoff to transport pollutants out of the VBS.

VBSs purify runoff through deposition. VBSs are usually composed of grasses or other dense understorey vegetation (in the case of forested buffer strips) that offer high resistance to shallow overland flow, thus decreasing runoff velocity in the strip. If the transport capacity is less than the incoming load of suspended solids, excess is deposited and trapped in the buffer strips (Dillaha *et al.* 1988). Filtration is possibly significant for larger soil particles, aggregates, and manure particles.

Many studies have suggested that VBSs can purify water, particularly subsurface water, through uptake of nutrients (mainly NO_3-N) by plants (e.g. Lowrance *et al.* 1984; O'Neill and Gordon 1994; Peterjohn and Correll 1984). Most of these studies have been based on riparian VBSs. However, considerable uncertainty exists about the relative importance of NO_3-N removal mechanisms, through denitrification or vegetation assimilation.

2.3.2. Sediment trapping in VBSs

Whether they are grass or forest vegetation, VBSs have been widely reported to play an important role in trapping sediment. As early as 1967, Wilson reported appropriate distances required for grass buffer strips to trap sand, silt and clay in floodwater on flat slopes. Neibling and Alberts (1979) showed that 0.6 to 4.9 m wide grass buffer strips reduced sediment discharge by over 90%. Clay transport was reduced by 37, 78, 82 and 83%, for the 0.6, 1.2, 2.4 and 4.9 m buffers, respectively. They observed that significant deposition of solids was just upslope of the leading edge of the buffer strip and 91% of the incoming sediment load was removed within the first 0.6 m of the buffer strip. Through a comprehensive research on sediment transport in grass buffers, Barfield *et al.* (1979) demonstrated that high sediment trap efficiencies could be obtained as long as the grass vegetation was not submerged. However, trapping efficiencies decreased significantly at higher runoff rates which inundated the vegetation. They also observed that most sediment deposition occurs just upslope of the buffer and within the first meter of the buffer, until the upper portion of the buffer was buried in sediment.

Dillaha *et al.* (1989) used a rainfall simulator to evaluate the effectiveness of grass buffers for sediment and nutrient trapping. They constructed plots with both shallow uniform flow and concentrated or channelised flow. The 4.6 and 9.2 m buffers with shallow uniform flow removed 75 and 87 % of the incoming sediment respectively. Buffers with concentrated flow were much less effective, with percentage reductions in sediment loads averaging 23 to 37%. Magette *et al.* (1989)

used a rainfall simulator on field plots to study the effectiveness of 4.6 and 9.2 m grass buffers in reducing nutrients and sediment from agricultural runoff. Sediment losses were reduced 52 and 75% by the 4.6 and 9.2 m buffers, respectively.

Dillaha *et al.* (1989) concluded that the effectiveness of grass buffer strips at increasing sedimentation depends on whether the surface runoff passes slowly and uniformly through the buffer. It was also pointed out that the effectiveness of experimental buffer strips with shallow uniform flow should not be used as a direct indicator of real world buffer strip effectiveness, because channelised and concentrated flow tends to be a major problem with practically-installed buffers (Dillaha *et al.* 1996).

Cooper *et al.* (1987) reported that a forest buffer zone receiving cropland runoff in a Coastal Plain catchment removed 84 to 90% of the sediment eroded from the cropland. Sand and coarse sediments were deposited at the forest edge, while silt and clay-sized particles were trapped deeper in the buffer. They suggested that buffer width should increase as stream order increases because the opportunity for sediment deposition decreases and transport capacity increases as stream order increases. In a literature review, Lowrance *et al.* (1995) reported that forested Coastal Plain buffer zones removed 80 to 90% of sediment loading to receiving waters. Sediment trapping efficiencies in other forest physiographic regions have not been as well investigated and are likely to be lower than those reported for the Coastal Plain because of steeper slopes and channelised flow effects.

Although these results demonstrate that both grass and natural forest buffer strips are effective in reducing sediment inputs, their effectiveness is dependent on a complex interaction between numerous environmental factors. Besides width and sediment input, the factors that are important in dictating the efficiency of VBSs for reducing sediment are micro- and macro-topographic relief, vegetation density and type which in turn determine hydraulic resistance, litter chracteristics, soil characteristics, particle size distribution of incoming sediments, subsurface drainage and slope (Gough 1988). Moreover, many of the models developed to evaluate the efficiency of VBSs in sediment retention assume shallow sheet flow (Dillaha *et al.* 1985), but that is a condition difficult to maintain in the field, particularly under natural rainfall events.

Furthermore, some researchers have noted that VBSs are sometimes sediment sources, although most researchers view them as sediment sinks. For example, Smith (1992) reported that a forested (pine) buffer strip in New Zealand became a sediment source rather than a sink after eight years of growth. Sediment loses were higher by 52 to 219% from the buffers than from the control pasture. Observations noted that the soil surface in the buffer strip was largely bare or covered with a thin loose layer of pine needles. As the pine canopy had closed, the original grasses and weeds in the pasture had presumably died leaving the soil with little protection from the erosive scour of the channelised surface runoff from the upslope pasture. This phenomenon raises the question about the long-term effectiveness of VBSs based on trees only for sediment trapping and consequent surface pollutant reduction.

2.3.3. Nutrient removal in VBSs

2.3.3.1. Surface nutrient removal

Infiltration of runoff and retention of sediment in buffer strips is likely to have a consequent impact on nutrient and other dissolved pollutant transport. An early study by Mather (1969) found that 61 and 39% of nitrogen and phosphorus respectively in a cannery effluent was removed during overland flow. Bendixen *et al.* (1969) conducted a similar study and reported that N and P was reduced by 94 and 81% respectively. Doyle *et al.* (1974) carried out a study to test the effectiveness of forest buffer strips in purifying runoff water contaminated with cattle manure. Although the slopes were very steep ranging between 35 to 40%, soluble N and P concentrations were reduced in runoff by 83 and 91% respectively following natural rainfall events. In 1977, Doyle *et al.* observed a significant removal of total soluble N, P and K within 3.8 m of a grass strip with a 10% slope.

Young *et al.* (1980) studied the impact of crop buffer strips on pollutant transport associated with feedlots. The buffer strips consisted of plots between 21.3 and 27.4 m wide, with a 4% slope, and planted with maize, sorghum, barley, or other food crops. Total N and P reductions were 84 and 83% respectively, and NH_4-N and PO_4-P reductions were 93%, although mean NO_3-N concentration in the runoff increased. Magette *et al.* (1986) evaluated grass buffer strips of 4.6 and 9.2 m in width, using a rain simulator, artificial fertilizers and manure. They reported that the effectiveness of the filter strips in treating agriculturally-derived nutrient contamination varied considerably, decreased with time, and depended to a great extent on the state of the filter at any given time. They concluded that filter strips are viable components of water and soil conservation plans for farms, although they did not recommend them as the sole component of such plans.

Dillaha *et al.* (1985) evaluated vegetative buffer strips for reducing pollution from feedlots. The study included filter strips of 4.6 and 9.2 m in width over a silty loam soil, with simulated rain and cattle manure. Slopes were 5, 11 and 16%. They found the planted strips reduced 58 and 69 % of total P for the 4.6 and 9.2 m strips respectively, compared with the bare-soil strips. Soluble P, however, was not successfully reduced and even increased in some case. Diaz-Fierros *et al.* (1990) conducted studies with a grass filter strip having a 15% slope, using cattle slurry and simulated rainfall. They reported that reductions in dissolved N levels of at least 100-fold were observed at 4 m from the top of the strip. In some cases, 187-fold reductions were observed as close as 2 m from the top of the strip and some cases even reached 300-fold reduction.

In general, buffer strips are effective at reducing nutrient transport in runoff, but some aspects of their performance, however, have not been fully quantified. Many studies (e.g. Arora *et al.* 1995; Dillaha and Hayes 1992; Magette *et al.* 1987) have found that buffers are less effective at removing nutrients than sediments. Concentrated surface runoff may also reduce the effective operation of buffer strips. Most plot studies have examined buffer performance under the shallow uniform flow conditions, but channelised flow tends to be a major problem associated with practically-installed buffers (Dillaha *et al.* 1986).

The fate of nutrients retained in buffer strips, particularly with regard to P, is another concern. Some short-term experiments (Dillaha *et al.* 1989; Magette *et al.* 1989) indicated that results for removal of soluble and extractable P are variable, although transport of total P may be reduced. Moreover, retained sediment may be re-eroded and retained P may subsequently be lost into the adjacent water system. It is not clear if similar results would be produced under natural rainfall conditions. These observations indicate the need for further research on long-term nutrient reduction by VBSs .

2.3.3.2. Subsurface nutrient removal

Phosphorus forms strong complexes with iron and aluminium hydroxides in acidic soils and with Ca in alkali soils. Its transportation is hence usually by adsorption on particles of sediment and most likely to occur only in surface runoff. Stewart *et al.* (1990) suggest that P is unlikely to move below the main root zone (0-35 cm) in the soils of Eutric Cambisols or Um 4.25 (Northcote *et al.* 1975) on a flood plain. Probably because of this reason, most studies on the impact of VBSs on the subsurface pollutant transport have tended to concentrate on N, particularly NO_3-N. A number of studies have investigated the effects of vegetated buffer strips, particularly riparian forest buffers on NO_3-N concentration of diffuse subsurface flow in natural catchments, and indicated that VBSs could significantly reduce NO_3-N in subsurface water.

Removal of NO_3-N in groundwater through riparian forest buffers has been studied in several areas of the southeastern USA Coastal Plain. There is an impermeable layer of clay or plinthite at depths of a few metres in these landscapes, which results in shallow lateral groundwater movement. Peterjohn and Correll (1984) reported a reduction of 90 to 98% in NO_3-N along a 19 m and a 50 m transect respectively in a deciduous riparian forest in spring. Initial NO_3-N concentrations in shallow groundwater entering the forest from adjacent corn-lands ranged from 5.4 in spring to 10.3 mg L^{-1} in winter. Jordan *et al.* (1993) also found that more than 95% of NO_3-N was removed by a deciduous riparian forest adjacent to cropland in the Delmarva Peninsula, Maryland USA. Lowrance *et al.* (1984) reported that NO_3-N levels were reduced from 2-6 to <0.5 mg L^{-1} through a bottomland hardwood forest buffer in Georgia, USA. At another site in the same catchment, Lowrance (1992) found a 94% removal in NO_3-N within a 55 m wide riparian strips. Jacobs and Gilliam (1985) reported NO_3-N reductions of more than 90% in a shallow subsurface flow from cropland through a riparian area in North Carolina.

Grass buffer strips were also effective in reducing NO_3-N in shallow lateral groundwater flow. Cooper (1990) found that of NO_3-N declined by 60-90% in groundwater through a 9-m grass buffer in New Zealand. Haycock and Pinay (1993) reported that 84% of NO_3-N in shallow subsurface water was removed by a 16-m wide grass buffer strip in Southern England. In Illinois, USA, a 39-m wide grass buffer reduced the NO_3-N level in shallow groundwater by 75-90% (Osborne and Kovacic 1993).

Comparison of the effectiveness of forest and grass VBSs is made difficult, as the experimental conditions and study designs for the results reported are considerably different. Consequently, it is still not possible to evaluate whether forest VBS are more efficient than grass VBSs.

Studies frequently demonstrated that NO_3-N losses from groundwater moving at shallow depths appear to take place over quite short distances. Haycock and Burt (1993) found a steep gradient in NO_3-N concentrations at the upslope boundary of a riparian buffer indicating that much of the NO_3-N was removed within the first 8 m of the buffer. Most NO_3-N removal occurred within 20 m of the cropland/forest boundary in Maryland and North Carolina catchments (Peterjohn and Correll 1984; Jacobs and Gilliam 1985). The fact that rapid NO_3-N removal takes place within a narrow zone suggests that many VBS areas have great-unused potential for NO_3-N depletion.

The possible mechanisms responsible for these widely documented retentions of nitrate include denitrification, assimilation and retention by the vegetation. However, few studies have accurately measured the amount of nitrate removed by any one of these mechanisms at a given site and no study has measured the removal rate by all three mechanisms. Denitrification is most often referred as the primary mechanisms of NO_3-N removal from subsurface flow. However, fluxes for denitrification are very difficult to determine accurately because the extreme spatial and temporal variability of denitrification rates in VBSs (Correll 1996). The products of denitrification include dinitrogen, nitrous oxide and nitric oxide and the proportion of these products are highly variable, depending on environmental conditions in the soil. Most studies do not measure all of these products, but express the research as an input-output balance.

In a literature review of studies on nitrate mass balance, Correll (1996) summarized that NO_3-N is effectively removed: a) at all times of the year in temperate climate; and b) from groundwater moving in subsoils at depths of several meters. However, studies found that potential denitrification occurs only in the top few centimeters of riparian soils (Ambus 1993; Pinay *et al.* 1993). Based on the fact that the conditions in the deeper subsoil are low temperature, low pH and low concentrations of dissolved organic matter, some scientists conclude that assimilation by the vegetation is the primary mechanisms of nitrate removal (eg Fail *et al.* 1986). However, this conclusion can not explain the fact that NO_3-N is removed in the winter at sites where the vegetation is dormant in the winter (Haycock and Pinay 1993). In fact, some studies (eg Jacobs and Gilliam 1983) have found that accumulation of nitrogen in buffer vegetation explained a relatively minor portion of the NO_3-N that was removed at their sites. Therefore, assimilation and storage in vegetation are likely to be significant mechanisms for NO_3-N removal, but not the primary mechanism (Correll 1996). Consequently, uncertain still exists as to the importance of the various mechanisms for NO_3-N removal in VBSs.

2.4. Other beneficial aspects of vegetative buffer strips

Apart from their prime function in controlling non-point source pollution, VBSs have a variety of advantages by comparison with the use of other conventional pollution control techniques. Firstly, the technique of VBSs is a more environment-friendly method, which is demonstrated by: 1) in VBS, plants provide a control strategy that utilizes solar energy and stores carbon (Shimp *et al.* 1993); 2) plants, particularly trees, are aesthetically pleasing; and 3) VBSs can provide habitat for wildlife while they exercise their prime function of controlling pollution. For example, Gillespie *et al.* (1995), from surveys based on wildlife habitat use and browsing statistics, found an increase in biodiversity due to VFS use in the Cornbelt Region of the Midwestern USA. Riparian vegetation buffer strips have been recommended as a means to enhance habitat for both aquatic and terrestrial wildlife populations within agricultural ecosystems (Osborne and Kovacic 1993).

Secondly, VBSs can provide other products such as timber, hay, nuts, fruit, berries etc. Gillespie *et al.* (1995), showed that tree planting within a vegetated filter strip system diversified land use objectives including hardwood production and wildlife habitat enhancement without restricting tree growth or the buffer effectiveness while still meeting water quality improvement objectives.

Furthermore, an important advantage of VBSs is that they are relatively cheap and easy to construct and maintain (Dillaha *et al.* 1986). Consequently, using VBSs as an alternative to control some types of point source pollutants (eg from feedlots) and non-point source pollutants (eg from cropland) from reaching water systems has been widely practised and studied in the recent years (e.g. Bingham *et al.* 1980; Cooper *et al.* 1987; Chaubey *et al.* 1993, 1994; Magette *et al.* 1986). Some studies have shown that the use of VBSs is cost-efficient. (Aull *et al.* 1980; Dillaha *et al.* 1986; Gillespie *et al.* 1995; Leeds *et al.*1993; Purvis *et al* 1989).

Clearly, the use of VBSs has many environmental and some economical benefits. However, some disadvantages inevitably occur when agriculture and forestry are associated with implementing and managing VBSs. Cook (1996) has documented the potential impact of buffer zones on agricultural practices and illustrated some of the difficulties associated with management of buffer zones. These disadvantages include loss of production and revenue, increasing loss of efficiency in small fields, restricting land use, increasing complexity of planning and execution of production and losing timeliness and operation, and advancing potential front of invasive weeds into field.

Bren (1997) reported that increasing riparian buffer widths on the 65 km^2 forested catchment of the Tarago River in southern Australia markedly decreased the area of forestland available for harvesting. The average quality of forest within the area remaining for harvesting also decreased with increasing buffer width, but the effect was not marked until buffer widths approach 150 m. The beneficial and detrimental aspects of VBSs are considered to benefit practical agriculture and forestry.

2.5. Design and management of VBSs

The principal components in designing a VBS are the determination of vegetation types and dimensions (Muscutt 1993). In terms of the prime function of VBSs, when selecting or designing the vegetation type and dimensions, the basic selection criteria should be related to maximizing the function of reducing pollutants. However, design procedures are limited by a series of related factors including both the main and secondary purposes of the VBS, natural conditions such as soil physical and chemical properties, topography of the site, hydraulic load, pollutant load, climate etc, and social factors such as public acceptance of the VBS and cost.

The purpose of designing a buffer strip is very important. For instance, herbaceous vegetation is recommended by many authors for VBSs designed as a tertiary treatment system aimed at reducing nutrients level. For instance, Dillaha *et al.* (1989) suggested grasses or legumes and advised against shrubs. Young *et al.* (1982) recommended plants with high N requirements such as maize and alfalfa. Black *et al.* (1984) considered herbaceous plants with a long growing period, high moisture tolerance and an extensive root system to be the most appropriate. In contrast, forest vegetation may be preferably chosen when designing a riparian vegetated buffer strip aimed at reducing surface and subsurface nutrient transport, as well as playing a role in other functions such as stabilising stream banks, contributing to the maintain of the aquatic system, etc. In fact, most of the current riparian VBSs that have been studied for their impact on pollutant transport, are forest buffers (e.g. Jacobs and Gilliam 1985; Jordan *et al.* 1993; Lowrance *et al.* 1983).

Plot studies have revealed that grass buffers provide sufficient resistance to encourage deposition and the removal of N and P (Dillaha *et al.* 1988; Delgato *et al.* 1995). The effect of a tree-based buffer is probably more dependent on the density of the groundcover and understorey but Phillips (1989b) suggested that resistance would be equivalent to grass in non-submerging flow through dense undergrowth with abundant leaf litter and woody debris. Riparian forests have been reported to remove sediments and associated pollutants from surface runoff (Peterjohn and Correll 1984) but contrasting result indicated that surface runoff was accelerated in a buffer planted with pine owing to the lack of adequate ground cover (Smith 1989). In summary, there are still major questions about whether forest or grass buffers is best and what optimal widths of strips are needed to provide a specific nutrient and sediment load reduction (Osborne and Kovcic 1993).

Regardless of its ecological advantages, implementation of a practical VBS needs to be socially acceptable and economically feasible. As Osborne and Kovacic (1993) pointed out, herbaceous vegetation was more compatible with the short-term nature of many government agricultural support programs in USA. Alternative forage crops such as oats and clover are commonly grown in these temporary set-aside programs. Thus the continued use and possible expansion of riparian grass VBS can be anticipated. Implementation of VBSs to control pollution entering a water system is a rather new ecological engineering idea and to date is supported by government programs to be practiced in most cases. For example, in the USA, VBS implementation has been conducted through the so called 'cost-share program' (Dickson and Schaeffer 1996). Very few examples have demonstrated that farmers are driven by economic incentives to implement VBSs without support from government, although in most cases, farmers are voluntarily involved in the programs (Cooper 1996).

The determination of appropriate buffer widths is still a debatable question. Very few studies in the literature provide clear answers to this question, principally because so many hydraulic and soil conditions must be considered in relation to potential volumes of water and the chemical content that may need to be handled. Phillips (1989b) suggested that buffer width should be between 15 and 80 m for the conditions found in North Carolina. His calculation was based on a model of water detention times taking account of soil hydraulic conductivity, soil moisture storage capacity, slope, and Manning's roughness coefficient. In summarising the reports presented to the International Conference on Buffer Zones, Haycock *et al.* (1996) pointed out that "The challenge for the scientific community is to engage policy and operational staff into the debate of how, why, where and what type of buffers could be created rather than taking the short-cut route of asking how wide. Given the complexity of buffer zone processes and their various purposes, a useful retort to question of 'how wide' is 'how wide do you want it'!".

Some new ideas and practices have been developed in recent years to implement and manage integrated VBSs. For instance, innovative designs that use specially selected fast-growing tree species can be grown as short-rotation woody crop systems. These systems produce biomass for energy in 5-8 years and timber products in 15-20 years (Colletti *et al.* 1991). Frequent harvests help to maintain active nutrient and pesticide interception by the woody plant community. On the basis of such systems, Schultz *et al.* (1995) designed a multi-species riparian buffer strip (MSRBS) system for intercepting suspended solids and agricultural chemicals from adjacent crop fields, stabilizing channel movement, and improving in-stream environments, while also providing wildlife habitat, biomass for energy, and high quality timber products. After 4 years of establishment in Central Iowa, this demonstration system seemed to be functioning as expected. Although the system looks promising, the authors noted that there are still many unanswered questions about the functions of MSRBS, such as the long-term effectiveness and efficiency of the buffers, the socio-economic and environmental benefits, and the cost of the system etc.

Welsch (1991) suggested a three-zone concept (Figure 2.1) as developmental guidelines for designing and managing forest buffer zones. Zone 1 is a permanent and undisturbed forested zone immediately adjacent to the stream. Its primary purpose is to stablize the streambank and provide habitat for aquatic organisms. Zone 2 is a managed forest zone, just upslope of zone 1, in which timber is periodically harvested. The primary function of zone 2 is to remove, transform, or store nutrients, sediments, and other pollutants flowing over the surface and through the groudwater. Zone 3 is a managed herbaceous strip, usually grasses, just upslope of zone 2 that is used to help slow runoff, filter sediment and its associated chemicals, and allow water to infiltrated into the ground. This three-zone concept is intended to be highly flexible in order to achieve both water quality and landowner objectives.

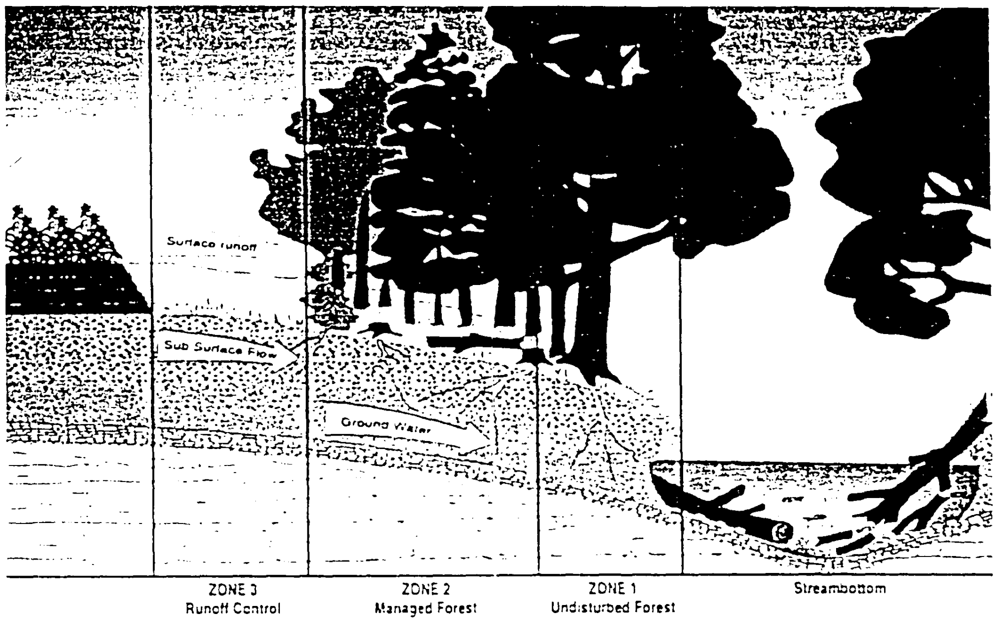


Figure 2.1. Three zoned riparian zone system (Lowrance *et al.* 1995).

Finally, sustainable effectiveness and efficiency is one of the most challenging topics in implementing and managing VBSs. Very few studies can be found about this topic. However, Dillaha *et al.* (1989) have undertaken excellent plot and field experiments for sustainable use of grass buffers. Dillaha and Inamdar (1996) point out the need to consider the medium- to long-term impact of sediment accumulation within a buffer. As more sediment is accumulated the profile of the buffer will change. The leading edge of grass buffers, for example, may become higher, water may start to pond at the edge of the buffer and ultimately flow to the lowest point along this edge, thus concentrating water and sediment, and leading to excess flow into one point of the buffer strip. Therefore, a sediment buffer strip will have a finite life and needs to be actively managed. Dillaha and Inamdar (1996) summarised a detailed guideline for management of grass buffer strips.

2.6. Status of study and practice of VBS in Australia

Australia is the driest continent in the world and has some of the oldest and most fragile soils (Phillips and Grant 1994). The loss and degradation of soil and contamination of water are matters that the country cannot afford in the long-term. As an important part of the integrative and sustainable strategies for catchment management, extensive and appropriate use of VBS systems can play an important role in soil and water conservation and environmental improvement overall. However, little information on both the study and practice of VBSs is available in Australia, particularly for agricultural systems. Just as Prosser *et al.* (1996) pointed out that 'Riparian zones have been a neglected part of catchment management in Australia'. Barling and Moore (1994) recently reviewed the role of buffer strips in the management of waterways, with specific application to Australian conditions. Issues addressed dealt with riparian vegetation, forest systems, agricultural systems, and natural buffer strips. They highlighted the limited information currently available on buffer strips for Australian condition, and the studies reviewed mainly focused on forest systems.

Buffer strips have been advocated as a method for protecting streams in forestry systems in Australia for more than two decades. Borg *et al.* (1988) defined a buffer in forest systems as a strip of undisturbed forest comprising overstorey and understorey left along a watercourse to protect water quality. Some studies have been undertaken in Australia examining the questions of buffer function and design in forestry (e.g. Borg *et al.* 1988; Cameron and Henderson 1979; Clinnick 1985; Grayson *et al.* 1992). These studies mainly address buffer strip dimensions and the extent of buffer along drainage networks. Based on a study with a deep porous red clay-loam soil that is common through out the lower hills of north-eastern Victoria, Bren and Turner (1980) suggested that 20 m width of undisturbed forest is required either side of the channel to ensure that most runoff enters streams by infiltration and percolation through the soil profile. In presenting environmental protection guidelines for forest harvesting, Cameron and Henderson (1979) recommended that the minimum width of filter strip should be 10 m either side of small streams in areas of low erosion hazard, and 20-30 m beside larger streams or areas with high erosion hazard. Bren (1997) studied the effects of increasing riparian buffer width on timber resource availability and indicated that increasing buffer widths markedly decreased the area of forest land available for harvesting and consequently the timber value with each coupe. In

Australia, the most commonly recommended buffer width for stream protection in forest operations is 30 m but is dependent on specific site conditions (Clinik 1985).

Some studies have addressed the required length and extent of buffers. Bren and Turner (1980) showed that a stream buffer should extend to the springhead or runoff-confluence point of any sub-catchment. This conclusion is further supported by the work of other Australian researchers (O'Loughlin *et al.* 1989, Finlayson and Wong 1982). Cameron and Henderson (1979) recommended that buffer strips be required where the catchment area exceeds 100 ha and would frequently be required in much smaller catchments, particularly in areas of high rainfall and erodible soils.

Hairsine (1996) compared the sediment and sorbed phosphorus filtering capabilities of grass filters with those of near-natural riparian zones. His results showed that riparian zones have an overall trapping efficiency and were capable of significantly reducing sorbed nutrient transport when the pollutant was carried by sediment in aggregate size classes greater than 20 mm. Hairsine and Prosser (1997) reviewed the role of perennial grasses in reducing nutrient movement within Australian agricultural landscapes. They suggested developing an integrated system, where grass buffer strips could be used along drainage lines in conjunction with near-natural riparian forest zones adjacent to streams to improve in-stream ecosystem and water quality.

In general, there is a lack of knowledge on vegetated buffer strips in Australia, particularly in relation to the mechanisms involved in reducing nutrient pollution reaching water systems. Cullen (1991) emphasised "This lack of effort into catchment management to minimise non-point discharges is surprising given the extensive degradation of waterways coming from the mismanagement of agricultural lands." He listed buffer strips (at least 10 m each side) with natural vegetation along waterways as one of the several basic strategies for managing a catchment to minimise non-point pollution. It is imperative to conduct further research on the functions, mechanisms, design and demonstration of VBSs under specific conditions in Australia, particularly at the agricultural catchment scale, combined with research into alternative best management practices.

2.7. Conclusion

Vegetative buffer strips may perform multiple functions but the basic function results from their beneficial impacts on surface and subsurface pollutant transport in catchments to protect water quality. The effectiveness of VBSs will be dependent on a complex interaction between numerous environmental factors and the VBS structure and composition. Although much work has been undertaken on the functions, mechanisms and design of VBS, information is still limited on several critical aspects. Some questions and controversies relating to the utility and efficiency of VBS still remain unanswered.

Firstly, questions are often raised about the efficiency of vegetation in VBSs and include: (1) whether forest VBSs are more efficient than pasture VBSs; (2) what is the effective width of a VBS for specific conditions; and (3) does species composition and structure make a difference in the functional effiency of VBS. Although many studies relating to these questions have been conducted, to some extent there are still few good answers. Such questions have important implications in adopting VBS programs and thus need to be further investigated, particularly in related to local conditions.

Furthermore, information on the long-term performance and management of VBSs is particularly limited and desperately needed. Although many observations have indicated that both natural and artificial VBSs are effective in reducing pollutants, most of these results are obtained from short-term studies. Some researchers argue that riparian areas might achieve equilibrium such that pollutant inputs are equal to or less than pollutant outputs. Therefore, the implementation and management of VBSs must consider and reflect these concerns. Further research is imperative on the long-term sustainability of VBSs in pollutant removal to protect aquatic systems.

Finally, VBSs must be viewed as an integral component of best management practice (BMP) in agricultural systems. VBSs are an important catchment practice, however, the in-field BMPs such as conservation tillage, contouring, strip cropping, erosion control works, controlled grazing, fertiliser and pesticide application, irrigation management, etc. should be given priority over buffer strips to control non-point pollutant transport and protect stream water quality (Barling and Moore 1994; Dillaha 1996). Very few studies have targeted the integrative use of VBSs with other BMPs at a catchment scale. Therefore, further research in this aspect is needed.

Chapter 3

Study design and methods

3.1. Field experiment

3.1.1. Introduction

A 20 ha liquid effluent disposal area was established at Tullimba (Figure 1.2) to retain and use nutrients by crops on the application area. Lateral movement of shallow (A-B interface) groundwater and associated nutrients may occur from the area down slope if rainfall or effluent irrigation rates are higher than the hydraulic conductivity of the soil B horizon and the A horizon is saturated. The general operation strategy of the field experiment site is to use trees and pasture to form a vegetated buffer strip to intercept the lateral moving flow and associated nutrients. This experimental vegetated buffer strip was located about 20 m from the bottom of the effluent application area. The experimental site contained 15 plots with two tree species (*E. camaldulensis* and *C. cunninghamiana*) planted at two densities and improved pastures. Two operational ways were designed for the experiment: one was to compare the water use and nutrient depletion for the tree and pasture treatments under conditions of rainfall and irrigation for the entire area; and another way was to determine the interception of lateral moving groundwater and nutrients by the experimental zone under the condition of effluent application only for the area above the experimental zone.

The life of the experiment is planted to operate for 15-20 years. The overall objective of this experiment is to compare the ability of tree plantation and improved pasture in reducing the lateral groundwater movement and nutrients. The specific aims are (1) to examine the survival and growth of *E. camaldulensis* and *C. cunninghamiana* trees and improved pasture in the buffer strip irrigated periodically with feedlot effluent; (2) to determine the water use of the trees and the improved pasture, and (3) to determine the effectiveness of trees and pasture in intercepting lateral groundwater flow and nutrients, particularly nitrate nitrogen.

3.1.2. Site description

3.1.2.1. Climate

Tullimba is located about 50 km west of Armidale, NSW (Figure 1.1). The altitude of the area is about 720 meters above sea level. The area experiences summer dominated rainfalls, cold winters and warm summers. Mean monthly maximum temperature is up to 28 ^{0}C in January and mean monthly minimum temperature is as low as 0 ^{0}C in July. The average monthly radiation level reaches 29 MJ m^{-2} in summer and is as low as 11 MJ m^{-2} in winter. The average monthly relative humidity is up to 80 - 85% in winter and as low as 65% in November and December. An average monthly maximum of 15 frosts occurs in midwinter during July and August at Tullimba.

Tullimba receives approximately 805 mm of rainfall per year. A mean monthly maximum rainfall of 110 mm and minimum rainfall of 35 mm occur in January and in either April or June respectively. For a one in fifty year storm, maximum rainfall intensity has been calculated as 52 mm per hour and this has been used as the design criteria for the feedlot (Dr Simon Lott, personal communication). Evaporation is estimated to be at an average monthly highest of 215 mm in December and lowest of 50 mm in June (Greene 1993).

3.1.2.2. Vegetation

Tullimba is typical of grazing properties in the Torryburn-Kinstown district on the western side of the New England Tablelands (Whalley and White 1993). The land was extensively cleared and the vegetation further modified by pasture improvement and cattle and sheep grazing. The liquid effluent area is grassland with scattered eucalypt trees including *Eucalyptus albens*, *E. melliodora*, *E. moluccana*, *E. dealbata* and *Angophora floribunda*. The dominant species in the area reflects the past management history. The dominating species are the introduced species *Eleusine tristachya* with the native *Bothriochloa macra* and herbaceous species of *Danthonia*. Several species of introduced clovers (*Trifolium glomeratum*, T. *arvense* and *T. repens*) also were patchily dominated species.

3.1.2.3. Soil properties

3.1.2.3.1. Physical properties

The experiment site was established about 20 m from the bottom irrigation bay of the liquid effluent disposal area (Figure 1.2) with an average slope of 3%. The land surface was basically uniform except for a small depressed area at about 200 m from the northern end (Figure 3.1). However, the depressed area was excluded from the experimental plots (Figure 3.2). The soil depth to rock exceeded 2.2 m over all the plots.

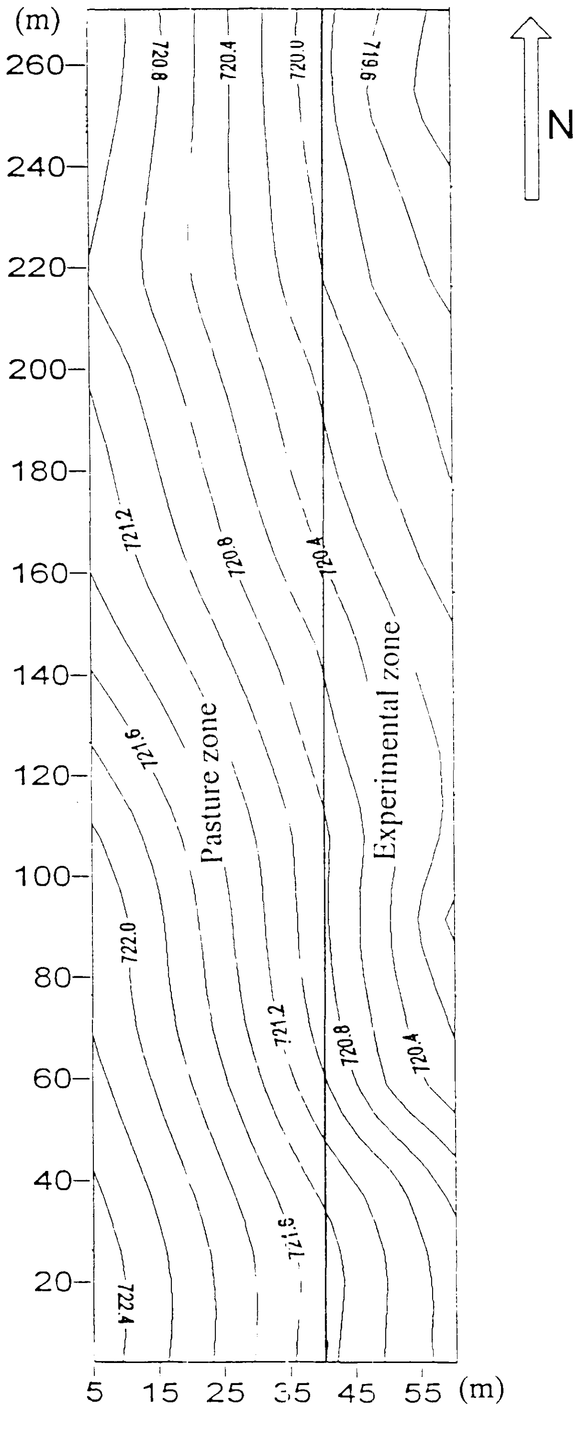
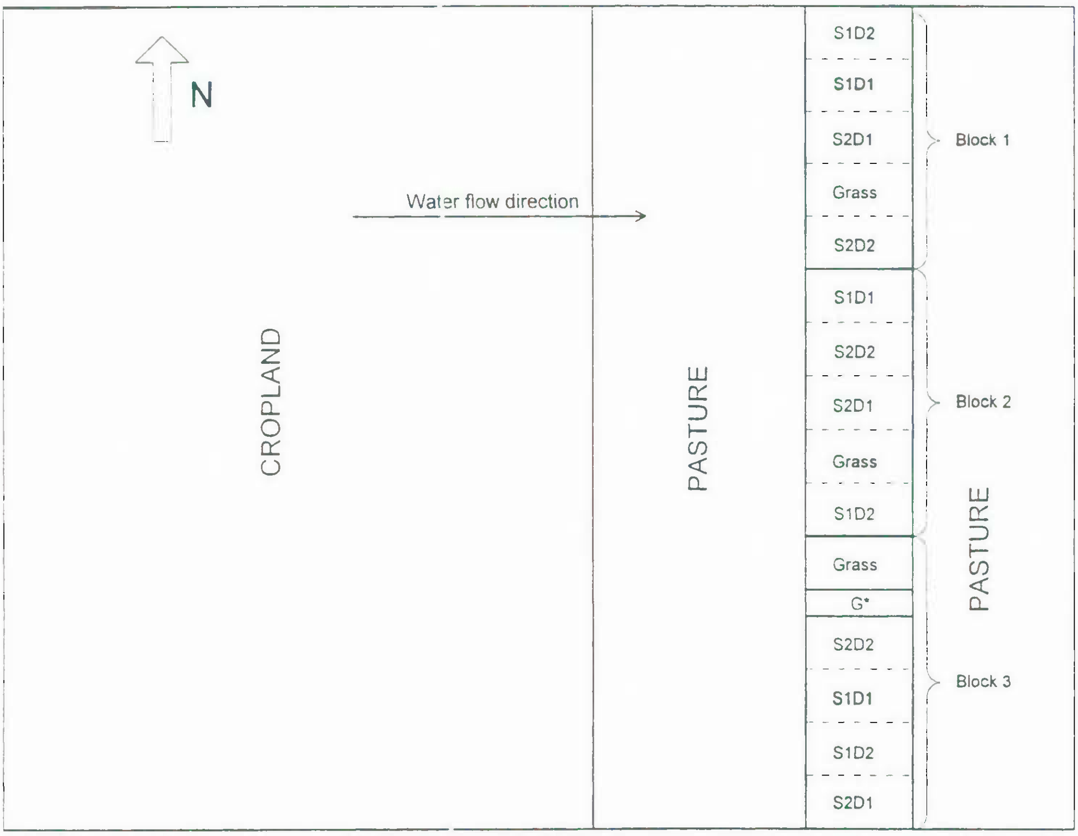


Figure 3.1. Topographic relief of the experimental site at Tullimba, NSW.



Note: * G - a depressed area, which was not included in the experiment, but was managed as pasture to prevent soil erosion.

Figure 3.2. Field experimental layout at Tullimba.

The soils are yellow solodic (Nothcote 1979), where the A-horizon is a sandy clay loam (Plate 1), which shows a tendency for dispersion in the A_2 layer. The B-horizon is light-medium to medium clay with traces of gravel and sands. The clay content increases with depth. The bulk densities increase from an average of 1.35 g

Plate 1. Showing the marked difference in texture between the soil A- and B-horizon in the experimental site at Tullimb. A- horizon is a sandy clay loam and B-horizon is light-medium to medium clay.

cm^{-1} at surface to about 1.70 g cm^{-1} at the B-horizon. (Table 3.1). The dense clay subsoil is much less permeable than the surface soil. Infiltration capacity is low, with surface rate (measured with a Disc Permeameter) of 7.0-15.0 cm per day and B-horizon rate of 0.5 –1.0 cm per day. The marked differences in texture and infiltration capacity between the A-horizon and B-horizon promote groundwater movement laterally along the top of the B-horizon layer (Burton 1993b).

Table 3.1. Soil physical properties of the experimental site at Tullimba, NSW, Australia.

Physical properties	Soil profiles			
	A1	A2	B1	B2
Depth (cm)	15-35	5-15	30-50	50 cm +
Texture grades	SCL*	SCL	LMC, MC	LC, LMC, MC
Bulk density (g cm^{-3})	1.30-1.40	1.50-1.60	1.65-1.75	1.65-1.75
Infiltration (cm day^{-1})	7.0-15.0	7.0-15.0	0.5-1.0	0.5-1.0

Note: *SCL - sandy clay loam; LMC - light medium clay; MC - medium clay; LC - light clay.

3.1.2.3.2. Chemical properties

The surface soil (0-15 cm) is acidic (pH 5.0-6.0) and has moderate to low nitrogen and phosphorus concentrations (Table 3.2) and low organic matter levels with total carbon 1.0-1.5% (Lawrie 1993). The exchangeable cations are reasonably well balanced, although sodium levels are higher than desirable, with an average exchangeable sodium percentage (ESP) of 5.2 %. The soil pH becomes less acidic and the ESP increases in the underlying A_2 horizon, but the overall level of plant nutrients is lower than at the surface. The soil pH rises in the clay subsoil with sodicity increasing to an average ESP of over 10%. The salinity level is very low over the whole soil profile indicated with low electrical conductivity (EC), but the average value of EC increases from 0.039-0.043 in the A horizon to 0.087- 0.127 dS/m in the B horizon.

In general, the soil in the effluent disposal areas, including the site of this experiment, can be described as light textured, shallow with bleached light grey A-horizon and yellow gravelly clay B-horizon, and these can be judged to be only marginally suitable for irrigated crop production (Collett 1993).

Table 3.2. Soil chemical properties of the experimental site at Tullimba, NSW, Australia. Values in parentheses are standard errors of the means (n=30).

Soil Properties	Soil Profiles			
	A1	A2	B1	B2
EC* (dS/m) (1:5 = soil: water)	0.043 (0.002)	0.039 (0.003)	0.087 (0.006)	0.127 (0.011)
pH	5.67 (0.05)	6.34 (0.06)	6.96 (0.08)	7.52 (0.16)
K (cmol/kg)	0.30 (0.02)	0.15 (0.01)	0.16 (0.01)	0.14 (0.01)
Na (cmol/kg)	0.26 (0.01)	0.38 (0.01)	1.41 (0.16)	1.77 (0.14)
Ca (cmol/kg)	3.44 (0.18)	3.22 (0.37)	5.81 (0.39)	5.86 (0.32)
Mg (cmol/kg)	1.30 (0.07)	2.66 (0.37)	8.71 (0.53)	9.64 (0.40)
ECEC** (cmol/kg)	5.30 (0.24)	6.41 (0.75)	16.08 (0.88)	17.41 (0.68)
Ca/Mg	2.75 (0.14)	1.39 (0.12)	0.73 (0.06)	0.62 (0.04)
ESP***	5.2 (0.4)	6.4 (0.5)	8.5 (0.8)	10.2 (0.8)
Total N (mg kg^{-1})	1172.5 (65.1)	382.5 (98.2)	271.5 (13.0)	213.1 (11.4)
NH_4-N (mg kg^{-1})	10.3 (0.3)	10.2 (0.3)	10.2 (0.3)	10.1 (0.4)
NO_3-N +NO_2-N (mg kg^{-1})	2.9 (0.4)	2.0 (0.3)	2.4 (0.4)	1.4 (0.2)
Total P (mg kg^{-1})	203.3 (10.8)	123.9 (7.1)	90.4 (3.4)	90.1 (3.9)
Available P (mg kg^{-1})	8.23 (0.9)	3.4 (0.2)	1.3 (0.1)	0.7 (0.1)

Note: *EC = electrical conductivity; **ECEC = effective cation-exchange capacity (=K+Na+Ca+Mg); ***ESP = exchangeable sodium percentage.

3.1.3. Design and establishment

The experimental area was about 300 m x 20 m (Figure 3.2). The experiment was a randomized block design with the two tree species (S1: *E. camaldulensis*; S2: C. *cunninghamiana*), two planting densities (D1: 1250 trees ha^{-2} on a 4 x 2 m basis; D2: 714 trees ha^{-2} on a 4 x 3.5 m basis) and a control pasture plot. Therefore, there were a total of 5 treatments in this experiment: S1D1, S1D2, S2D1, S2D2 and 1 pasture control. A total of 15 plots, each being 18 m x 16 m, were arranged in 3 blocks (replicates). The pasture plots were designed to use a composite mix of the 7 grass species. The sowing rate was 56 kg ha^{-2}, with species ratio of 24:10:10:3:3:3:3 for the 7 species respectively. A pasture (with the same 7 species) zone of 300 m x 35 m was established above the experimantal zone. Above the pasture zone was the cropland, the main effluent disposal area.

3.1.3.1. Selection of tree and grass species for the experimental zone

The criteria taken into consideration for tree species selection included: 1), adaption to the local climatic and edaphic conditions; 2), having some tolerance to soil salinity; 3), relatively high water users but not subjected to drought stress; and 4), having relatively high growth rates (Boland *et al.* 1984; Campell 1990; Cremer 1990b). In terms of the above selection criteria, *Eucalyptus camaldulensis* spp *obtusa* Dehnh. and *Casuarina cunninghamiana* Mq. were selected as the tree species for this experiment. For the sake of convenience, *Eucalyptus camaldulensis* spp *obtusa* Dehnh. is referred to as *E. camaldulensis* through this thesis. Tree seedlings came from the State

Forestry Nursery, Inverell, NSW. The most common grass species used for improved pasture establishment in the Northern Tablelands were selected for this experiment. These included *Festuca arundinacae, Phalaris aquatica, Lolium perenne, L. x hybridum, Trifolium hirtum, T. pratense, T. repens* and *T. subterraneum*. Pasture seeds were obtained from Purkies Seed, Armidale, NSW, a commercial seed supplier.

3.1.3.2. Establishment

The soil was ploughed and deep-ripped before planting trees. Trees were planted in December 1995, with two trees planted at each planting site. The excess trees are planned to be removed when the trees became well established. The pasture plots were established at the same time but needed to be replanted in March 1996. Trees were protected after planting by using milk cartons around them to help tree survival and promote early growth (Plate 2a). Grasses and weeds were controlled by mowing between the rows and manually hoeing around each tree. Grasses were not vigorous during the first growing season after planting trees because the soil was ploughed before planting, only the weeds around trees, therefore, were manually hoed as required. Starting from the second growing season, grasses between the planting lines were mown 2 to 3 times each growing season by using a tractor-drawn slasher/mulcher, with the mulch being left on the surface. Eighty gram fertilizer Starter 12 (12 N, 22 P, 0 K and 3 S) was applied to each tree during the first two growing seasons. Fertilizer was applied into a spade slit about 15 cm on the up-slope side of the tree. Insecticide (Rogors ^{TM}) was sprayed over *E. camaldulensis* when insect damage was evident mostly in mid December each year.

Plate 2a. Showing the established experimental site at Tullimba. Planted trees were protected with milk cartons around them. Observation wells (each instrument station including 3 wells) were established above, within and below the experimental zone.

Plate 2b. The trees (*E. camaldulensis* and *C. cunninghamiana*) and pastures in late November 1997 (in the third growing season).

3.1.3.3. Instrumentation for monitoring soil water storage and groundwater

Soil water storage and groundwater were monitored by using the neutron probe method and observation wells respectively. A series of instrument stations were installed across the experimental site (Plate 2a and Figure 3.3). Fifteen stations, each including 3 observation wells and 1 neutron probe access tube (Figure 3.4), were established, respectively in the zones 4 m above, within and 4 m below the experimental zone. Other 15 neutron probe access tubes were set up in the middle of the application pasture zone above the experimental zone. Fifteen observation well stations (each including 3 observation wells) were also installed along the border between the cropland and the pasture zone. The neutron probe access tubes (inside diameter = 4.5 cm) were installed to depths of 1.4 – 1.8 m according to soil profiles. At each instrument station, 3 wells were installed at 3 different depths. The deep, middle and shallow wells were about 2 m, 1 m and at the interface between the A and B soil horizons respectively (Figure 3.4). In total, 180 wells were installed. The observation wells were used for monitoring water tables and collecting groundwater samples for chemical analysis.

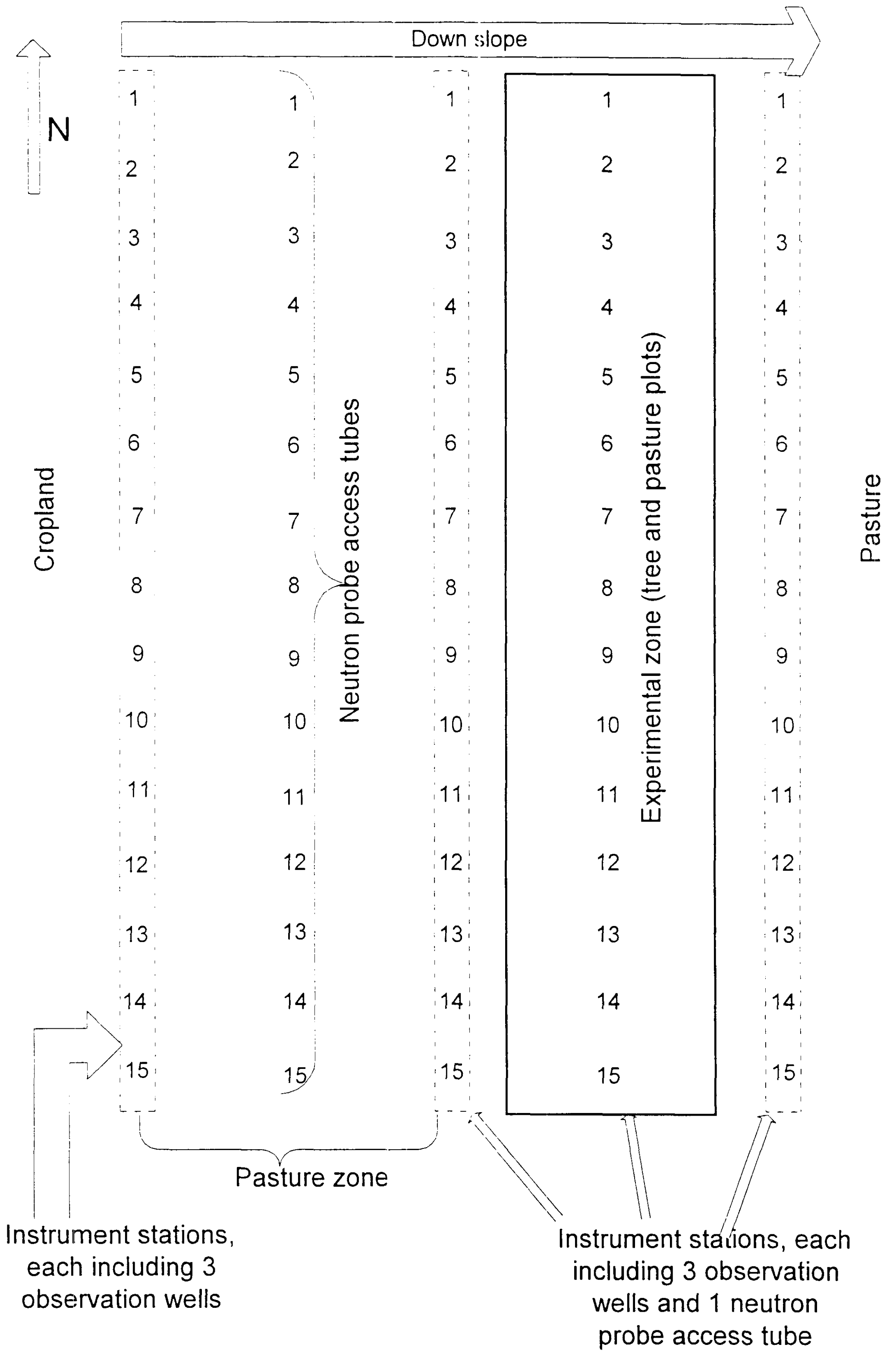


Figure. 3.3. Locations of instrument stations for the field experiment at Tullimba, NSW.

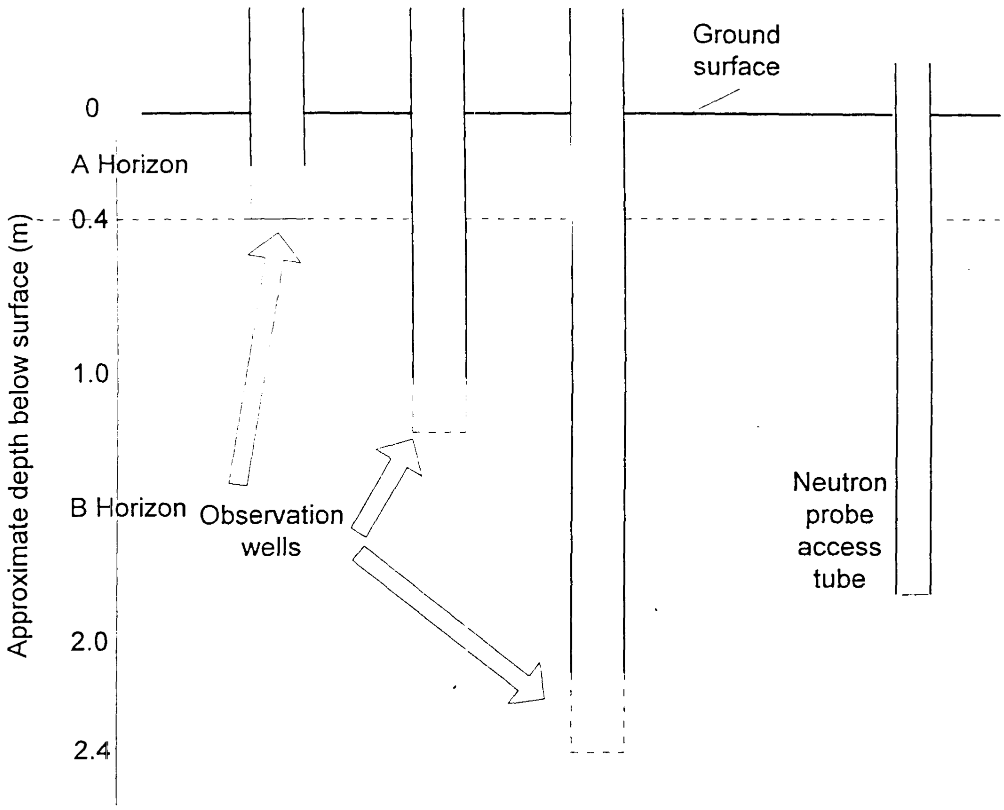


Figure 3.4. The observation wells and neutron probe access tubes at an instrument station.

3.1.3.4. Instrumentation for surface runoff measurement

Surface run-off during heavy storm events was evident during the experimental period. Runoff was measured by using a simple runoff collector, which consisted of a 15 cm diameter PVC pipe with a 4 cm wide slit allowing surface runoff to flow into the collecting pipe. A water storage container was placed at an end of the pipe, while another end of the pipe was sealed. An outside-lean cover was fixed above the runoff-inlet-slit to prevent the direct entry of rainwater. The runoff sample area was 2 m x 2 m, which was trenched around the edge water-proof insulation. Six runoff collectors were installed in the plots that were selected at random among the 15 plots. The water volume collected in each bag was measured following each storm to calculate the runoff produced.

3.1.3.5. Effluent irrigation

The characteristics of the effluent as it was applied to the irrigation area are shown in Table 3.3. Total nitrogen concentration (2.32 mg L^{-1}) and total phosphorus concentration (2.56 mg L^{-1}) were lower by comparison with other reported wastewaters (Cromer *et al.* 1982; Polglase *et al.* 1994; Stewart *et al.* 1986). Total N level and P level in secondary-treated sewage effluent in Australia are generally in the range of 10-30 mg L^{-1} and 4-10 mg L^{-1} respectively (Myers *et al.* 1995). The effluent is held in two holding ponds until required for irrigation.

The experimental plots were periodically spray-irrigated with a travelling irrigator. Irrigation time and quantities were dictated by two factors: one was to promote the growth of the young trees and the pasture by relieving drought stress on some occasions and to encourage their root system development by increasing water deficit on some other occasions; and the other factor was to create specific experimental soil conditions in order to compare water use and nutrient removal by trees and pasture. To avoid run-off, irrigation rates were controlled around 8-12 mm per hour according to the soil permeability and soil moisture condition. Application rates were measured by flow meters associated with the irrigator. In total, an equivalent of 512 mm effluent was irrigated during the period after planting to March 1998.

Table 3.3. The typical chemical composition of the irrigat:on effluent.

Parameter	Unit	Average value	SE
pH		8.7	0.05
Electrical Conductivity (EC)	ds m^{-1}	0.42	0.03
Total Alkalinity	mg L^{-1}	157	18.22
Chloride (Cl)	mg L^{-1}	92	6.32
Sodium (Na)	mg L^{-1}	39.2	11.59
Calcium (Ca)	mg L^{-1}	16.5	5.32
Potassium (K)	mg L^{-1}	44.6	10.87
Magnesium (Mg)	mg L^{-1}	15.3	2.35
Sodium Absorption Ratio (SAR)		1.75	0.13
Sulphate (S)	mg L^{-1}	7.8	1.23
Total Nitrogen	mg L^{-1}	2.32	0.98
Ammonium-Nitrogen (NH_4-N)	mg L^{-1}	0.25	0.09
Nitrate-Nitrogen (NO_3-N)	mg L^{-1}	0.22	0.07
Total Phosphorus	mg L^{-1}	2.56	1.02
Phosphate (PO_4-P)	mg L^{-1}	2.21	0.78

3.1.4. Survival and growth measurement and analysis for the trees and pasture

Tree survival was assessed within 3 months and 12 months of planting respectively. The few trees that died were replaced to restore the trials to full stocking. Tree height was measured approximately bimonthly. Many trees of *E. camaldulensis* developed multi stems, due in part to frost damage during the first winter, and hence measurements and comparison of the tree diameters and tree crowns became difficult. Therefore, measures of tree basal area and crown cover were made only once in April of 1998 (end of the third growing season), when tree growth lowed with the onset of winter. Tree diameters were measured at 5 cm above ground to calculate tree basal area.

The pasture biomass measurements were made twice in the growing season of 1996/1997 and only once in 1997/1998 because of the dry conditions and the lack of effluent for irrigation. Biomass measurements were taken before commercial harvesting and bailing of the pasture. Four subplots, each 0.5 x 0.5 m, were selected at random in four quarters of each plot. The samples were cut, bagged and fresh-weighed before being oven-dried at 80 ^{0}C for 48 hours. The biomass of the native pasture between the tree rows was also estimated by using the same method.

3.1.5. Estimate of water use by trees and pasture

Total water use E (evapotranspiration, including interception by tree canopy and grass) by the plantation or pasture was calculated from:

$$E = P + I - \Delta W_p - R - D$$

where P was precipitation received; I was irrigation applied; ΔW_p was the change in soil water content over a measured period; R was run-off produced from the experimental plot during heavy rainfall events; and D was the water lost from the soil profile due to deep drainage. Measurements of these parameters are described below.

3.1.5.1. Soil water storage

Soil water storage was measured using a neutron meter (Hydroprobe, 503DR model, C.P.N.®). Measurements were taken at 10 cm intervals for the first 60 cm, and 20 cm to the depth of the access tube. Measurements were taken usually every 2 weeks, starting from May 1996 when the access tubes were installed and the plantation and pasture were at the end of the first growing season. Soil water storage W (mm) over certain depths was calculated from:

$$W = 10 \sum D_i \, \theta_i$$

where D_i was the thickness of the soil layer i (cm); θ_i was the volumetric water content of that layer ($cm^3 \, cm^{-3}$); and 10 was a conversion factor used to express W in mm.

It is necessary to calibrate count rates for a specific soil type to obtain precise volumetric water content values (θ_i). In this experiment, calibration was made by using the field calibration method (Greacen *et al.* 1981). Four locations were selected about 6 meter below the experimental zone. Two 1.5-meter aluminium access tubes about two meters apart at each location were installed. One of each pair of tubes were destructively sampled on May 6, 1997, when the soil was very dry while the other tube was sampled on October 3, 1997, when the soil was very wet. Before sampling the tubes, neutron meter count rates were taken in the soil at 10-cm depth intervals to a depth of 1.0 meters. Counts were also taken in water. Five core samples were then collected for each selected depth, close to the access tube. The soil cores were transported to the laboratory and oven-dried at 105 °C for 48 hours. Bulk density (ρ) and gravimetric soil water content (θ) were determined on each core. Regression analysis was used to compare the calculated volumetric water content to the count ratio (CR_{soil}/CR_{water}). A separate regression analysis was conducted for each of the main soil horizons (A and B).

3.1.5.2. Soil water deficit

Water deficit was used to compare water use potential for the plantation and pasture. Water deficit (ΔW), the depletion of soil water, was calculated as the difference between the tube field capacity (W_f) and subsequent measurements in the corresponding tube (W_i) at each instrument station. Hence,

$$\Delta W_i = W_{fi} - W_i$$

where the tube field capacity (W_{fi}) was determined for each instrument station from measurements taken on occasions when the soil water content had stabilized after periods of heavy rainfall (Honeysett *et al.* 1992).

The change in soil water content over a measured period (ΔW_p) was the change in water deficit during that period:

$$\Delta W_{pi} = \Delta W_i - \Delta W_{i-1}$$

where ΔW_p could be positive or negative depending on whether the soil profile was drying or wetting.

3.1.5.3. Separation of evaporation and drainage

To examine the change in soil water storage that results from evaporation, as distinct from the total water loss from the soil profile, the depth of the root extraction zone (effective rooting depth) needs to be specified (Gregory *et al.* 1978). Evaporation is calculated from summing rainfall and changes of water storage within the effective rooting depth. The water loss in the soil profile beyond the effective rooting is considered as deep drainage. In this study, the effective rooting depth was determined by using the method developed by McGowan and Williams (1980a,b). Graphs were drawn that showed water content against time for each depth of measurement in the profile and by examining the presence of discontinuities and fluctuations in the rate of water loss caused by rainfall. Because the experimental plants were trees and perennial grasses, they have roots at a considerable depth throughout the year. The effective root depth, therefore, was assumed to be the same all the year around.

3.1.6. Estimate of the effectiveness of trees and pastures in reducing lateral groundwater flow

Two methods were used to estimate the effectiveness of trees and pastures in reducing lateral groundwater flow. The first method included monitoring water table depths to determine the effectiveness of the tree treatments and the pasture in intercepting lateral groundwater movement. Depth of water table was measured by using each of the 3 observation wells at each instrument station. Water table measurements were made fortnightly, but more frequently under some conditions such as after heavy rainfall or during irrigation experiments. For the sake of convenience, the groundwater from the deep wells will be referred through this thesis as 'deep groundwater', while the groundwater from the shallow will be referred as 'shallow groundwater'.

The second method included monitoring the change of soil moisture by using the neutron moisture probe method. During the procedure, the pasture zone above the experimental plots was loaded by irrigation to saturated condition, and lateral groundwater flow, particularly the interflow between the A and the B horizon was then generated down slope into the experimental plots. Measurements and comparison of soil moisture at all instrument stations down slope (in pasture zone when irrigation was applied, then above, within and below the experimental plots) resulted in the estimation of relative effectiveness of the tree and improved pasture plots in intercepting lateral groundwater movements.

3.1.7. Estimate of the effectiveness of trees and pastures in removing nutrients

3.1.7.1. Nutrient removal from lateral groundwater flow

Groundwater chemistry was monitored during this experiment to determine the removal of nutrients associated with lateral goundwater flow. The groundwater observation wells were used to collect groundwater samples. Water samples were taken from the deep wells, firstly, by pumping all old water from the wells with a small electric water pump to let fresh groundwater refill, and secondly, collecting water samples after 24 hours from the refilled fresh water. Water samples from the shallow water were collected immediately once the water produced, because usually the water in the shallow wells had relatively short residence time. Water samples were then returned to the laboratory and frozen until analysis.

Water samples were analyzed to determine pH, electrical conductivity (EC), soluble reactive phosphorus (PO_4-P), and nitrate-nitrogen (NO_3-N). EC and pH were selected as indicators to monitor the level of salinity and alkalinity. Once large changes were found in these indicators, analysis of K^+, Na^+, Ca^{++} and Mg^{++} and Cl^- were conducted. PO_4-P and NO_3-N were selected to monitor water quality as they are considered to be the main contamination sources to terrestrial water systems (LWRRDC 1993; Myers *et al.* 1995).

Water pH and EC were measured by using a 744 pH meter (Metrohm™) and a YSI Model 30 Handheld Salinity, Conductivity & Temperature System (YSI Incorporated, USA) respectively. Soluble reactive P was determined using the manual molybdate blue method (Muphy and Riley 1962). NO_3-N was determined by using the manual cadmium reduction method (Rayment and Higginson 1992).

3.1.7.2. Estimate of changes in surface soil chemical properties

Soil samples were collected on 15^{th} of February 1998 to measure chemical change in surface soil against the baseline conditions determined at the time of establishing the experiment. Eight cores (with diameter of 4.5 cm and depth of 5 cm) of surface soil were taken at random from each experimental plots and thoroughly mixed. The distance between a sampling location to any individual tree base was limited to >1 m in the tree plots. Five sampling sites were randomly selected in the middle of the pasture above the experimental zone. The sampling method was as the same as described above. Soil samples were transported back to the laboratory and immediately air-dried.

Thirty soil sampling sites (15 along the upper border and other 15 along the bottom border) were selected for soil baseline sampling in the experimental zone in July 1995 before the tree and pasture plots were established. Soil samples were taken by using an auger for A1, A2, B1 and B2 horizons respectively.

Soil EC, pH, total phosphorus, extractable phosphorus, total Kjeldahl nitrogen (TKN), ammonium-nitrogen (NH_4-N), nitrate-nitrogen (NO_3-N) and cations K^+, Na^+, Ca^{++} and Mg^{++} were analyzed. Soil electrical conductivity was determined by mechanically shaking 12 g of air-dried soil in 60 ml of distilled water (1:5 soil/water (w/v)) for 1 hour, and measuring with a YSI Model 30 Handheld Salinity, Conductivity & Temperature System (YSI Incorporated, USA). Soil pH was measured by using the same soil/water suspension as described above and using a 744-pH meter (Metrohm^{TM}).

Extractable P was determined by the manual colour method of Bray 1 (Bray and Kurtz 1945), by extracting P from the soil by using 0.025M HCl-0.03M NH_4F solution. NO_3-N and NH_4-N were determined using the manual colorimetric methods (Bremner 1965). NO_3-N and NH_4-N were extracted by mechanically shaking 10.0 g air-dried soil in 100 ml 2M KCl solution for 1 hour, settling for >30 minutes and filtering with No: 42 (Whatman™) filter paper. NO_3-N content was analysed by the copperized cadmium reduction method and NH_4-N by the indophenol blue method. Total N and total P were determined by using the Kjeldahl digestion method. A 0.50 g of soil sample was digested in 10.0 ml H_2SO_4/Se mixture at 350 °C. N and P in the digests were then analyzed by using automated methods with a Technicon Auto Analyser II™. Cations K^+, Na^+, Ca^{++} and Mg^{++} were determined by using an atomic absorption spectrophotometer.

3.2. Glasshouse experiment

3.2.1. Introduction

A glasshouse experiment was established to measure the effectiveness of trees and pasture to remove nutrients from laterally flowing water under controlled conditions. This experiment was designed to simulate the field condition and support the field experiment. Fifteen large metal boxes were used and filled with the duplex soil profile and either trees (*E. camaldulensis* and *C. cunninghamiana*) or pastures were then planted. Water and KNO_3 solution were fed into the soil at the A-B interface. Through collecting and analysing soil solution collected at various locations, across and down the profile, the effectiveness of trees and pasture in reducing nutrients transport, was estimated. The growth and water use by the trees and pasture were also measured.

3.2.2. Design and establishment

Fifteen rectangular metal boxes, each measuring 120 cm by 40 cm by 40 cm (Figure 3.5) were constructed. The boxes were sealed with sealer along the edges and corners to prevent water leaking. Soil was collected from Tullimba at a location adjacent to the field experiment site. In order to resemble the field soil profiles, the bottom 20 cm of each box was filled by using the B-horizon soil and the top 15 cm of the box was filled with the A-horizon soil. The B-horizon soil was re-compacted to ensure it resembled field conditions and had no leaks. The boxes were arranged in a glasshouse so that there was a 3% slope to facilitate water movement laterally through the soil profiles.

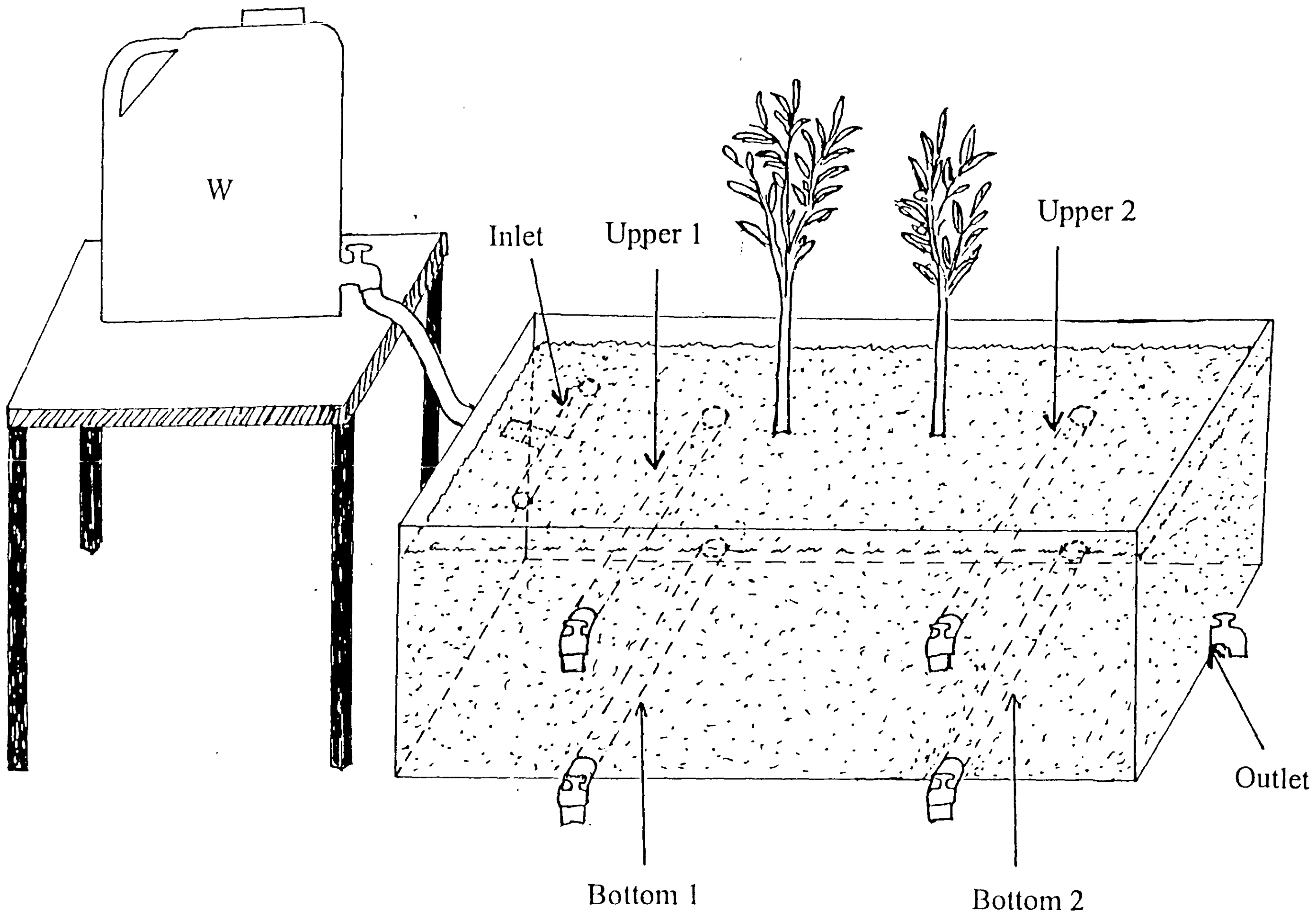


Figure 3.5. Schematic representation of the glasshouse experiment: a two-tree box. W: container for water and nutrient solution; Inlet: pipe releasing water into the soil profile; Upper 1 and Upper 2: water sampling pipes at interface of A- and B-horizon; Bottom 1 and Bottom 2: water sampling pipes 2 cm above the base of the box; Outlet: water sampling pipe at the base of the other end of the box.

A inlet pipe (PVC) was located at the input end of the box and about 10 cm deep from the soil surface in the middle of the A horizon. Five sampling pipes (upper 1, upper 2, bottom 1, bottom 2 and outlet) were installed, to extract water samples, above and below the trees and pasture, and at the interface of the A and B horizons and at depth in the B horizon (deeper and lower). An outlet sampling pipe was at the other end of the box and installed at the base of the box. The upper sampling pipes were set at the interface between the soil A-horizon and B-horizon. The bottom sampling pipes were in the B-horizon, 2 cm above the base of the box. All pipes extended the width of the box and had small holes (diameter of 0.5 mm) along their length and covered with nylon gauze to prevent clogging. The inlet pipe was connected by tubing through the end panel of the box to a 15 L container (W, Figure 3.5). The container was positioned 0.5 m higher than the surface of the soil to develop a hydrostatic head. The side sampling pipes (upper 1 and 2, bottom 1 and 2) were located 30 cm and 90 cm from the inlet pipe respectively. Sampling pipes were fitted with taps to collect water samples.

Two tree species (S1: *E. camaldulensis*; S2: C. *cunninghamiana*) were used in the glasshouse experiment and were planted at two densities (D1: 4 trees in each box; D2: 2 trees in each box). Seedlings were planted uniformly between the sets of sampling pipes. The experiment was carried out in two stages (Table 3.4). The control treatment for the first stage of the experiment was bare soil, while for the second stage the control was sown to improved pasture (*Festuca arundinacae, Phalaris aquatica, Lolium perenne, L. x hybridum, Trifolium hirtum, T. pratense, T. repens* and *T. subterraneum*). The first experimental stage commenced from May 29 and lasted to December 28, 1997. The second stage lasted for about 4 months, finishing at the end of April of 1998. Treatments were replicated 3 times and laid out in the glasshouse as a randomised block design (Figure 3.6). Tree seedlings came from the same source as used in the field plantation and were planted on May 29, 1997.

The minimum temperature in the glasshouse was kept above 5 °C in the winter period, and the maximum temperature was controlled to be below 35 °C in the summer period. Insecticide (Rogor ™) was sprayed once at 5 weeks after the trees were planted, when some signs of insect damage on *E. camaldulensis* were found. Afterwards, no further insect damage occurred. Weeds were removed by hand.

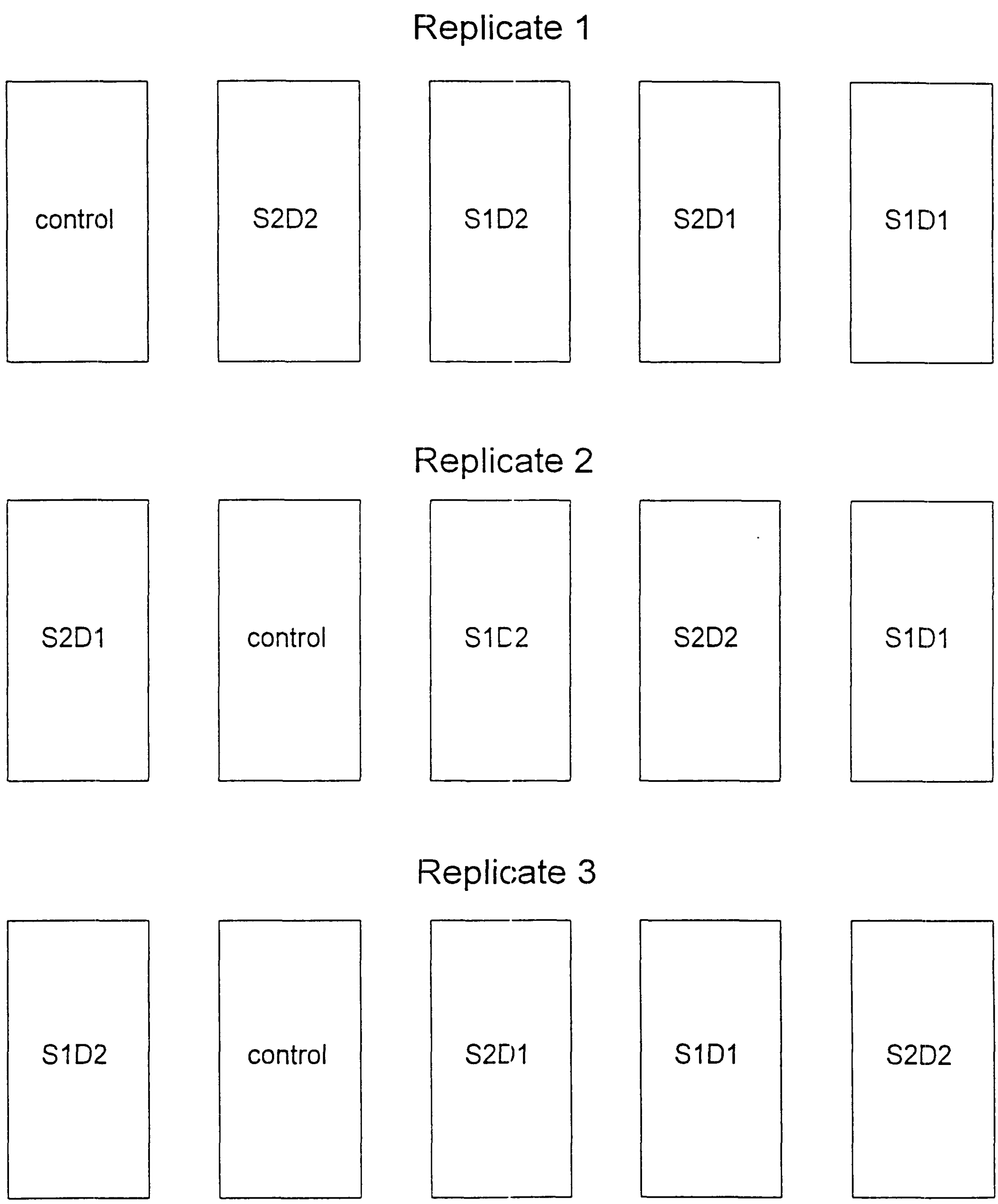


Figure. 3.6. The glasshouse experiment layout.

3.2.3. Experimental periods and nitrate application

The glasshouse experiment was divided into 6 periods, the first three of which were included in the stage 1 with bare ground controls, and the last three of which were included in the stage 2 with pasture controls (Table 3.4). The first period was a soil consolidation and tree establishment phase from May 29 to August 18, 1997. The trees and the control treatments were regularly watered using tap water to keep soil moisture and to promote tree growth during this period. The second period, from August 18, 1998 to October 27, 1998, was a 'soil nutrient depletion period'. Five litres of tap water were supplied to each box per day, without any nutrients. Soil solutions were collected and analysed 6 times during this period. The third experimental period was from November 1 to December 28, 1997, in which, five litres of low KNO_3 concentration solution (10 mg L^{-1} of NO_3-N) were added to each box per day. This period was referred to as the 'low NO_3 addition period'. Three collections and analyses of soil solutions were carried out in this period.

The fourth period was a pasture establishment phase from December 29, 1997 to February 19, 1998, during which, the pastures and trees were regularly watered with tap water without any nutrients. No soil solutions were collected in this period. The fifth period was from February 20 to the end of March 1998. Five litres of high KNO_3 concentration solution (20 mg L^{-1} of NO_3-N) were added to each box per day in this period. This was called a 'high NO_3 addition period'. Three collections and analyses of soil solutions were taken in this period. The sixth period was in April 1998, in which, water use by trees and pasture was measured. Water without nutrients was added to the soils in this period. This period was referred to as a 'water use measurement period'.

Table 3.4. The experimental stages and periods for the glasshouse experiment.

Stages	Periods	Comments
Stage I	1 (29.5-18.8.1997)	Soil consolidation and tree establishment phase.
(Controls were	2 (19.8-27.10.1997)	Nutrient depletion period.
bare ground)	3 (1.11-28.12.1997)	Low NO_3-N application period.
Stage II	4 (29.12.1977-19.2.1998)	Pasture establishment period.
(Controls were pastures)	5 (20.2-31.3.1998)	High NO_3-N application period.
	6 (1.4-30.4.1998)	Water use measurement period.

3.2.4. Growth measurement of trees and grasses

The height of the trees was measured since planting to the middle of November 1997 when all trees had to be trimmed as they reached the roof of the glasshouse. Six height measurements were carried out during this period. Tree diameter was measured monthly since about 4 months after planting until the end of the experiment. The grass biomass was measured by harvesting at the end of April 1998.

At the end of the experiment, trees were harvested and above and below ground biomass was separately determined. Foliage was separated from stems and branches. Tree roots were extracted from the soil by hand digging and washing, and were separated into three categories: tap roots, lateral roots (>0.3 cm dia.) and fine roots (<0.3 cm dia.). Vertical distribution of tree roots in soil was determined by separating the soil into two layers: the surface 15 cm (the A-horizon) and the subsurface 20 cm (the B-horizon). All plant materials were dried in an oven at 75 °C for 72 hours to determine dry weight biomass.

Pastures were harvested at the end of the experiment. Above ground biomass was estimated by cutting and drying at 75 °C for 48 hours. To estimate below ground biomass, a 20 cm x 20 cm sample area was selected in each plot, as the grasses were distributed uniformly in each box. The samples were sorted by washing and roots were dried at 75 °C for 48 hours.

N and P were determined for all plant tissues by using the Kjeldahl digestion method. Plant tissue (0.10 g) was placed in 10.0 ml H_2SO_4/Se mixture and digested at 350 °C. N and P in the digests were then analyzed by using automated methods with a Technicon Auto Analyser II™.

3.2.5. Soil solution sampling and analysis

Solution samples were collected from the water inlet container and the 5 solution collecting taps (Figure 3.5) on a fortnightly basis. About 200 ml soil solution was collected for each sample. All the taps were allowed to drain for about a half minute prior to sample collection. The solution samples were vacuum-filtered and placed in cold storage at 2 - 3 °C until analyzed.

The main chemical parameter determined for the soil water samples was nitrate-nitrogen (NO_3-N), while other parameters including pH, electrical conductivity (EC), soluble reactive phosphorus (PO_4-P), were also analyzed. The analysis methods have been described in section 3.1.7.1.

3.2.6. Water use measurement for the trees and pasture

Water use by trees and pasture was measured from April 1, 1998, when the trees had grown for 10 months and the grasses 3 months. The trial period was 4 weeks. Water use E (evapotranspiration) was estimated from:

$$E = W - \Delta W$$

where W was the water supplied during the trial period; ΔW was the change in soil water content over the period.

Soil water content change ΔW for each plot (box) was estimated by measuring the soil water content (gravimetrically) at the starting and ending dates of the trial. Soil samples were taken in the center of the box for the A and B horizon respectively and then oven-dried at 105 °C for 48 hours. The time to add water was determined by monitoring the soil surface moisture. When the soil surface became dry in some box, twenty litres of water were added to it. The total amount of water added in each box was recorded during the period.

3.3. Data analysis

STATISTICA™ (StatSoft, Inc. 1993) was used for all statistical analysis in this thesis. The data were subjected to the standard ANOVA test for the randomised block design both in the field and glasshouse experiment. Various contrasts and their standard errors were calculated and F-test used to assess the significance. Regression analysis was used for the analysis of tree diameter growth and nutrient dynamics in soil solution in the glasshouse experiment.