Land Treatment of Dairy-farm Effluent Using Short Rotation Forestry

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Jonathan K. F. Roygard

Massey University
New Zealand

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Abstract

Under the Resource Management Act (1991) New Zealand dairy farmers are required to dispose of dairy-farm effluent in such a manner as to have no adverse effect on the receiving environment. This study investigated the land treatment of pond treated effluent to short rotation forestry (SRF). The study involved both field trials and modelling work to assess sustainability of these systems in terms of nitrogen leaching to groundwater.

A lysimeter study investigated 3 SRF species, 2 evergreen species of Eucalypts (*Eucalyptus saligna*, *E. nitens*) and a deciduous willow (*Salix kinuyanagi*) in the treatment of dairy farm effluent. Trees were grown in lysimeters (1.8 m diameter, 1.0 m depth) to enable measurement of water and nitrogen balances. A bare-soil treatment was used as a control. The application of dairy-farm oxidation-pond effluent totaled 218 g N lysimeter^{-1} (equivalent to 872 kg N ha^{-1}) over 2 irrigation seasons (December 1995-June 1996 and September 1996-April 1997). Effluent was applied weekly during the irrigation seasons at a rate of 21 mm week^{-1}. No effluent was applied during the winter period.

The drainage period of the *E. nitens* was shorter than that of the *S. kinuyanagi*, and rates of leaching were respectively lower. Both these treatments leached for shorter periods than *E. saligna*. Leaching of the bare-soil treatment was consistently high throughout the experiment. Water use through evapotranspiration was found to have a large impact on drainage volume and timing.

The trees were shown to improve effluent treatment because high evapotranspiration rates reduced the volume of leachate passing beyond the root zone. Further, uptake of nitrogen by the trees reduced the quantities of nitrogen available for leaching. In this study both *E. nitens* and *S. kinuyanagi* were more suitable for land treatment than the other 2 treatments evaluated. The low nitrogen concentration in the leachate under the *S. kinuyanagi* is the key criterion which determines the suitability of this tree species for land treatment of effluent. The low total loading of nitrogen to the groundwater of the *E. nitens* treatments is the key criterion in determining *E. nitens* suitability. Although the nitrogen concentrations in the leachate of the tree treatments were generally less than the bare soil treatments, they were still greater than the New Zealand drinking water standard (NZDWS) of 11.3 mg NO_3^--N, during certain periods of the experiment. From the lysimeter experiment it was concluded that the leachate nitrogen concentrations might have been reduced if the amount of nitrogen applied in the effluent was reduced.

Total production of above-ground biomass in the 2.5 years, based on the stocking rate of 4000 stems ha^{-1} was equivalent to 15.6, 30.6, and 21.3 Mg ha^{-1} yr^{-1} for *E. saligna*, *E. nitens*, and *S. kinuyanagi* respectively. Although scaling up biomass estimates from small plot trials and particularly lysimeters introduces associated errors, the estimates fell within the ranges measured elsewhere in New Zealand.

The lysimeter study was complemented by the modelling of the water and nitrogen balances of SRF land treatment systems. Ultimately, the aim of the model was to investigate the effect of changes in management practices on sustainability in terms of nitrogen leaching of SRF systems treating dairy-shed effluent. The model selected for this purpose was a lumped parameter model (LPM). The water and nitrogen balances of

the bare soil and *E. nitens* treatments were simulated with the model to determine the applicability of an LPM scheme to predict system behaviour. The model predicted, with broad agreement, the measured water and nitrogen balances of the lysimeter experiment. The model was then used to simulate the behaviour of a SRF plantation receiving dairy-shed effluent at a rate of 200 kg N ha^{-1} yr^{-1} over 27 years. This simulation predicted the occurrence of high nitrate concentrations in the leachate. This would be a limiting factor for the long term sustainability of such a system. A sensitivity analysis of the model was used to reveal the important parameters of water movement and nitrogen cycling that effect both nitrogen concentration and quantity in the leachate moving below the root zone. Water movement was most sensitive to root zone depth, effective rainfall, available water and crop water use. The nitrogen fate parameters with greatest effect on leachate nitrogen concentration and quantity were denitrification activity and volatilisation. Plant growth parameters of light utilisation efficiency, maximum leaf nitrogen concentration and specific leaf area strongly effected leachate nitrogen concentration and quantity. Mineralisation rates of the soil humus and the senescence rates of plant material also impacted on quantity and concentration of nitrogen leaching.

The model's applicability as a decision support tool was demonstrated by examining the impact of various effluent loading rates on the leachate concentration and quantity. Based on leachate nitrate concentrations being on average lower than the NZDWS, the key finding was that the sustainable loading rate for the simulated system was found to be around 75 kg N ha^{-1} yr^{-1}.

The major finding of both the lysimeter experiment and the modelling study was the high nitrogen concentrations leaching from SRF dairy-shed effluent treatment systems. The LPM model clearly provides a platform from which to investigate many other possible scenarios of management to minimise the leaching of the high concentrations of nitrogen into the ground water.

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List of Symbols

Definition	Symbol	Unit
Water balance equations		
Evapotranspiration	ET	mm
Potential Evaporation (Penman monteith)	E_p	mm day^{-1}
Evaporation	E	mm day^{-1}
Drainage	D	mm
Irrigation	I	mm
Soil water storage	W	mm
Total rainfall	R_t	mm
Effective rainfall	R_e	mm
Soil water content	θ	m^3 m^{-3}
Constant (Eq 3.11)	c	mm day^{-1}
Time	t	day^{-1}
Water movement		
Maximum water content	θ_s	m^3 m^{-3}
Field capacity	θ_f	m^3 m^{-3}
Wilting point	θ_w	m^3 m^{-3}
Saturated hydraulic conductivity	K_s	mm day^{-1}
Beta constant	β	
Soil bulk density	ρ_b	Mg m^{-3}
Root zone depth	z_R	m
Maximum soil evaporation	E_s	mm day^{-1}
Nitrogen parameters		
Nitrate adsorption	k_{DN}	L kg^{-1}
Ammonium adsorption	k_{DA}	L kg^{-1}
Nitrification	k_2	day^{-1}
Denitrification	k_3	day^{-1}
Denitrification zone below θ_f	δ_D	m^3 m^{-3}
Volatilisation (days time evaporation rate)	k_v	days
C:N ratio	r_o	
Critical N content for growth	N_{crit}	kg N ha^{-1}
Decomposition of litter	k_{lit}	day^{-1}
Decomposition of humus	k_{hum}	day^{-1}
Crop parameters		
Crop factor	k_c	
Drought tolerance	τ	
Light utilisation efficiency	ε	g DM MJ^{-1}
Senescence rate of roots	γ_R	day^{-1}
Senescence rate of stems	γ_W	day^{-1}
Senescence rate of leaves	γ_F	day^{-1}
Maximum leaf N content	N_f	mg N g^{-1}
Allocation to leaves	A_l	
Allocation to stems	A_s	
Allocation to roots	A_r	
Specific leaf area	σ_f	ha-leaf kg-DM^{-1}

Chapter 1 General Introduction

1.1 Statement of the problem

Rising population numbers in a world of fixed size, are increasing environmental pressures on the Earth's ecosystems. Political, scientific and market forces are all pushing further towards sustainable management of the World's resources. This is all happening at a time when farming systems are becoming more intensive, in order to keep up with demand, and to provide better economic returns. To achieve these goals, high inputs of agrochemicals such as fertilisers and pesticides are becoming increasingly common. Assessing the environmental effects of such practices on the long-term sustainability of farming has become an important area of environmental research.

With the increased emphasis on maintaining environmental quality, new environmentally based laws have been developed. New knowledge of the environmental impact of certain traditional land management practices has led to more sustainable solutions being sought. This has been the case for disposal of farm effluent in New Zealand. Under the Resource Management Act (1991) through the mechanism of discharge consents, New Zealand dairy and piggery farmers are required to dispose of effluent in such a manner as to have no adverse effect on the receiving environment.

Dairy-farm effluent in New Zealand is most commonly treated via two-pond systems. After this treatment, the ponded-effluent may then be released into streams or rivers. However, levels of nutrient removal, particularly for nitrogen, achieved by the two-pond system are proving insufficient to protect the quality of receiving waters (Hickey et al., 1989). To protect water quality, regulatory authorities are now restricting effluent

disposal into surface waters. Land treatment of effluent is therefore becoming more widespread (Cameron et al., 1997; Bond, 1998). Land treatment of agricultural wastes has the advantage of recycling nutrients back to the land thereby increasing the productivity of the land. Thus there are economic incentives to utilise wastes containing nutrients for crop production. The fertiliser value of New Zealand's dairy-farm effluent, pig slurry, and poultry manure has been estimated in New Zealand to be NZ$36 million per year (Roberts et al., 1992).

Irrigation of dairy-shed effluent to pasture soils is increasingly being practiced by dairy farmers in New Zealand (Di et al., 1998a; Di et al., 1998b; Silva et al., 1999). However, one of the problems with this practice is the potential for contaminants, such as pathogens and heavy metals in the effluent to enter the food chain. Another land-treatment option for dairy farmers is to apply ponded-effluent to soil growing trees. Potentially these systems may meet the needs of regulatory agencies by their use of trees to strip the nutrients from the wastewater as it percolates through the soil. Eucalyptus and Salix tree species have recently been advocated in short rotation forestry (SRF) systems for their land treatment potential, as well as for their biomass production (Myers et al., 1994; Myers et al., 1996; Tungcul et al., 1996; Nicholas, 1997; Nicholas et al., 1997). The fast, initial growth-rate of these species indicates both high water and nutrient uptake. As well by possessing coppicing ability, SRF crops are advantageous in land treatment systems (Nicholas et al., 1997). The biomass harvested at the end of the 2–10 year rotations might be suitable for fuelwood and this may provide an economic return.

The land application of dairy-farm effluent to SRF may limit the loss to the environment of wastewater, which is high in nitrogen. However to be sustainable, it is necessary to determine the fate of the various components in the effluent after it has been applied to land. Nitrate leaching to groundwater is an environmental concern because of its potential impact on water quality, and even human health. This health risk forms one of the major basis for a discharge consent as it uses New Zealand drinking water standard (NZDWS) that potable water (including potable groundwater) should contain less than 11.3 mg L^{-1} of NO_3^--N (Ministry of Health, 1995). Nitrate is poorly retained by most soil, and so it moves freely with the percolating soil water. Thus, nitrate leaching occurs if the water inputs of effluent irrigation and precipitation exceed the soil-water storage and evapotranspiration demand. In such cases water that may contain nitrogen will proceed down beyond the rootzone potentially to contaminate the groundwater.

Previous SRF land treatment research studies have generally focused on municipal effluent application (Hopmans et al., 1990; Myers et al., 1995; Nicholas, 1997) or meat-works effluent disposal (Lowe, 1994; Guo, 1999). These studies have been field based and unable to measure nitrate leaching beneath the rootzone of SRF land treatment systems. Tungcul et al. (1996) studied the application of ponded dairy farm effluent to SRF trees. The study showed high biomass yields and high levels of nutrient accumulation in the trees. However, increasing soil nitrate levels in the root zone were also identified, indicating nitrate leaching could be occurring. The field study was unable to quantify the amount of nitrate leaching (Tungcul et al., 1996).

In the present study, the sustainability of dairy-shed effluent-irrigation to SRF is assessed in terms of nitrate leaching to groundwater. This will be investigated through

both a lysimeter experiment and a modelling exercise. The experimental focus of this project is to obtain measurements of water and nitrogen balances in an SRF dairy-shed effluent-treatment system. These will be used to develop an understanding of the key processes so that modelling tools can be used for the prediction of the fate of applied water and nitrogen in SRF land treatment systems. This second step of modelling will enable quantification of the effects of different effluent loading rates on nitrate leaching thereby aiding the design and management of sustainable land treatment systems.

1.2 Objectives

The objectives of this study were:

1. To measure and calculate the water and nitrogen balances of three species of SRF and a bare soil control receiving dairy-shed effluent irrigation;
2. To parameterise a lumped parameter model (LPM), utilising data from the field experiment and literature values to simulate the water and nitrogen balances of the field experiment;
3. To demonstrate the applicability of the LPM model as a decision support tool.

1.3 Preview of the chapters

Having introduced the study in this Chapter, the lysimeter experiment is discussed in Chapters 2, 3, and 4. These three chapters on the lysimeter experiment are linked. Chapter 2 presents the aspects of biomass production. Chapter 3 focuses on the water balance. Chapter 4 draws together information in the 2 preceding chapters to present the nitrogen balance information. All these chapters possess separate introductions, methods and literature reviews, with each focusing on one aspect of the lysimeter experiment.

Chapter 5 presents the LPM model developed in conjunction with Steve Green and Brent Clothier from HortResearch. This chapter presents both the simulation of the lysimeter experiment, and the sensitivity analysis of the model. The model's applicability as a decision support tool is then demonstrated through an investigation which seeks to determine the sustainable loading rate for SRF plantations receiving dairy-shed effluent irrigation.

Chapter 6 provides a general overview and conclusions of the study, with an emphasis on the management aspects relating to SRF systems for sustainable management of dairy-farm effluent.

Chapter 2 Tree growth and productivity

2.1 Introduction

Efforts to develop cost-effective ways of producing and using biomass resources for heat and power are currently being supported by various national and international programs (Kendell et al., 1997). The rising demands for energy from renewable sources has generated new ideas and turned attention to fast-growing, short rotation forests (SRF) for woody biomass production (Ranney et al., 1987; Lodhiyal, 1995a). Assuming complete combustion, biomass only releases CO_2 taken up during growth and thus provides a CO_2 neutral fuel. To increase the economics of these systems, greater biomass production is desirable (Sims and Riddell-Black, 1998). However, removal of high yields of whole trees increases nutrient removal from the site leading to questions of long term sustainability (Perry and Maghembe, 1989; Stanley and Montagnini, 1999). Production and sustainability, in terms of nutrient depletion, can be improved via addition of extra water and nutrient to the site.

Additions of water and nutrient are known to increase the production of forests (Birk, 1995). The growth potential of most tree species is not normally realised under natural conditions (Ericsson, 1995). Irrigation and fertiliser application during the growing season of *Salix viminalis* (Christersson, 1987) and *Picea abies* (Linder and Flower-Ellis, 1992; Nilsson, 1993) in Sweden, *Pinus radiata* in Australia (Snowdon and Benson, 1982) and *Eucalyptus globulus* in Portugal (Pereira et al., 1989; Pereira et al., 1994) have resulted in 100 to 300 % increases in the rate of biomass production.

2.1.1 Land treatment and biomass production

Sources of extra nutrient and water for these plantations include effluent from sewage treatment plants, dairy farms, and food processing industries. The irrigation of nutrient-enriched wastewaters onto SRF systems is a win-win situation. The inputs of water and nutrient are likely to increase the biomass production and reduce the chance of nutrient depletion of the sites. In addition, the continued application of high nutrient containing effluents can produce a sustainable treatment system as the nutrient added is being removed from the site with the biomass at each harvest. Land treatment of wastewater may divert effluent from disposal into waterways, thus protecting their environmental quality. The combination of effluent irrigation with biomass production is receiving commercial and research interest (Edgar and Stewart, 1979; Hopmans et al., 1990; Lowe, 1994; Myers et al., 1995; Nicholas, 1997; Nicholas et al., 1997; Sims and Riddell-Black, 1998; Guo, 1999; Roygard et al., 1999). Nicholas et al. (1997) and Nicholas (1997) provide reviews of the status of the hardwood trees in land treatment schemes in New Zealand and Australia.

2.1.2 Sustainability issues

There are several sustainability issues to address with irrigation of SRF plantations using effluents of high nutrient content. Potential environmental effects include degradation of ground water quality (nutrient levels) and altering the depth of the water table. Excessive effluent application may lead to contamination of the ground water (Cameron et al., 1997; Bond, 1998). Water table levels may, rise through over irrigation or lower due to water uptake by the trees. Trees with high water consumption potential, (in comparison to pasture systems) have been known to lower water table levels in and around the plantation areas (Bell et al., 1990; Schofield and Bari, 1991; Bari and Schofield, 1992; Fiekema et al., 1999). These concerns have lead the call for increased

knowledge of nutrient and water balances at effluent land treatment sites (Cameron et al., 1997; Bond, 1998).

2.1.3 Dairy-shed effluent treatment using SRF

As outlined in the preceding chapter, this study sets out to investigate the use of SRF species for renovation of ponded dairy-farm effluent, through a lysimeter experiment and modelling. Water and nitrogen balances of the trial and the modelling will be discussed in later chapters. The current chapter focuses on the biomass production aspects of the field experiment. The biomass harvested at the end of the SRF rotations, is suitable for fuelwood and may provide an economic return (Sims and Riddell-Black, 1998). The Salix species, used in this trial, also offers an alternate end use as a fodder crop under drought conditions (van Kraayenoord et al., 1995).

2.1.4 Species selection

Eucalyptus and Salix tree species have recently been advocated in SRF systems for their biomass production and land treatment potential (Myers et al., 1994; Myers et al., 1996; Tungcul et al., 1996; Nicholas, 1997; Nicholas et al., 1997). The species chosen for this study were *E. saligna* (Sydney blue gum), *E. nitens* (shining gum) and *Salix kimuyanagi* (Japanese shrub willow). *S. kimuyanagi* is also known as *S. viminalis* 'Kinuyanagi' (van Kraayenoord et al., 1995). Previous research identified each of these species suitable for SRF land treatment systems, however further work is required to examine the water and nutrient balances of effluent land treatment systems using these tree species (Hopmans et al., 1990; Tungcul et al., 1996; Nicholas, 1997; Nicholas et al., 1997).

2.1.5 Species performance

Biomass production of these species is known to be high. The previous studies of biomass production and performance in New Zealand and Australian land treatment schemes are outlined here for each species.

S. kinuyanagi clones in New Zealand have yielded 12 Mg ha^{-1} yr^{-1} (van Kraayenoord et al., 1995). In Sweden, irrigated and fertilised plantations of *S. viminalis* have been shown to yield 9-14 Mg ha^{-1} yr^{-1} (3-year-old) (Nilsson and Ericsson, 1986) and 11.4 Mg ha^{-1} yr^{-1} (2 year old) (Christersson, 1987). In Scotland, irrigated and fertilised Salix stands (1-year old) have produced 9-10 Mg ha^{-1} yr^{-1} (Cannell et al., 1987). Senelwa (1997) used small plot trials in the Manawatu region of New Zealand to evaluate SRF species production. All species were planted at a stocking density of 3470 stems ha^{-1} and received no irrigation, nutrient additions or silvicultural practices. In the biomass production trials *S. kinuyanagi's* mean annual increments (MAI) were 23, 29 and 33 Mg ha^{-1} yr^{-1} for 3, 4 and 5 year rotations respectively. The literature values of *S. kinuyanagi* dry matter production rates in New Zealand range from 12 to 33 Mg ha^{-1} yr^{-1}. *S. kinuyanagi* was a top performer in a species trial of 9 willows and *E. nitens* for dairy-shed effluent treatment (Tungcul et al., 1996).

E. nitens is the most commonly planted Eucalypt in New Zealand; with its fast early growth, frost hardiness and site tolerance, *E. nitens* looks promising as a successful commercial species (Miller et al., 1992). Kincheff and Carter (1991) reported yields of 32 and 40 Mg ha^{-1} yr^{-1} at 5861 and 17,414 stems ha^{-1} respectively for 7 year rotations of *E. nitens* grown in Northland. *E. nitens* plantations near Rotorua grown at 6470 stems ha^{-1} produced 20 Mg ha^{-1} yr^{-1} at age 5 (Madgwick et al., 1981) compared with 15 Mg

ha^{-1} yr^{-1} at 1675 stems ha^{-1} at age 4 years (Frederick et al., 1984). In the trials of Senelwa (1997) mentioned earlier, the MAI of *E. nitens* were 22, 46 and 59 dry Mg ha^{-1} yr^{-1} for 3, 4 and 5 year rotations respectively at a planting density of 3470 stems ha^{-1}. *E. nitens* has been grown in several New Zealand and Australian land treatment sites (Nicholas, 1997). Much of the following information of *E. nitens* performance in New Zealand and Australian land treatment sites come from the reviews of Nicholas (1997) and Nicholas et al. (1997). In Australia, *E. nitens* is grown in the municipal effluent treatment project at Wagga Wagga, where it was in the top eleven eucalypt performers in a species evaluation. In New Zealand land treatment sites, *E. nitens* has shown good results. In the Horotiu meatworks effluent trial, *E. nitens* was one of the 3 best performing species with *E. saligna* and *E. botryoides*. At Whakarewarewa municipal effluent trial, *E. nitens* far out performed the other species (*Accacia dealbata*, *Cupressus macrocarpa*, *E. saligna*, *E. botryoides*, *E. globulus*, *E. ovata*, *Juglans nigra*, *Paulownia tomentosa* and *Pinus radiata*). At Oringi in the Hawkes Bay, *E. nitens* was trialed prior to establishment of a commercial scale 100 ha plantation for meat works effluent treatment but "...the poor showing of *E. nitens* was not expected as it had performed well in many other areas of the country" (Nicholas, 1997). *E. nitens* has been very successful in the Moutere municipal effluent scheme. The literature values of *E. nitens* dry-matter production rates in New Zealand range from 15 to 40 dry Mg ha^{-1} yr^{-1}.

E. saligna is one of the most widely planted Eucalypts (FAO, 1979). DeBell et al. (1997) showed the MAI peaked at 28 Mg ha^{-1} yr^{-1} for 2-4 year old *E. saligna* plots in Hawaii with high water and nitrogen inputs. Also in Hawaii, the MAI of 16 year old *E. saligna*, has been measured at 20.3 Mg ha^{-1} yr^{-1} (Binkley and Ryan, 1998). In New Zealand yield information for *E. saligna* is scarce. Frederick et al. (1985) measured over

16 Mg ha^{-1} yr^{-1} yield of dry matter for *E. saligna* grown at 829 stems ha^{-1} in the Bay of Plenty. In the trials of Senelwa (1997) mentioned earlier, the MAI of *E. saligna* were 12, 7 and 11 Mg ha^{-1} yr^{-1} for 3, 4 and 5 year rotations respectively, at a planting density of 3470 stems ha^{-1}. The poor performance in this trial was attributed to infestation by *Ophelimus eucalypti*, a leaf gall wasp (Senelwa, 1997). *E. saligna* has also been grown in several New Zealand and Australian land treatment sites as reviewed by Nicholas et al. (1997) and Nicholas (1997). In Australia, *E. saligna* has been a strong performer at municipal effluent treatment sites in Wodonga, Wagga Wagga, Shepparton and Werribee. *E. saligna* was one of the two top biomass producers at Wodonga, in the top eleven eucalypt performers at Wagga Wagga, and one of the 2 species selected for planting on the delta site of Werribee. In New Zealand land treatment sites, *E. saligna* has shown mixed results. In the Horotiu meatworks effluent trial, it was among the 3 best performing species with *E. nitens* and *E. botryoides*. At Whakarewarewa, a municipal effluent trial, *E. saligna* and all other species trialed were far out-performed by *E. nitens*. The literature values of *E. saligna* dry matter production rates in New Zealand range from range from 7 to 16 Mg ha^{-1} yr^{-1}.

Thus the three species chosen for this experiment have all shown good potential for biomass production in land treatment sites.

2.1.6 Importance of dry matter production

Harvest of the above-ground biomass of a SRF crop presents a marketable product to improve the economics of the land treatment site. For fuelwood production the trees' above ground dry matter production is of greatest importance. Although all above ground biomass (including the leaves) can be burned, it is likely the larger stems will be used for fuelwood. However, removal of whole trees from the site, rather than just the

stem wood component, is beneficial in a land treatment system to provide nutrient stripping. Removal of all the above-ground biomass takes nutrient out of the land treatment system, increasing the sustainability of high nutrient especially nitrogen inputs to the system. Estimates of above-ground biomass and their nitrogen content are required to assess the nitrogen removed from the site at harvest. Total biomass production, including root mass, is essential for calculation of the nitrogen budget of the system.

Nitrogen distribution in biomass is known to differ with size components and location within the tree (i.e. root, stem or branch) (Young and Carpenter, 1975; Ericsson, 1995). Thus the mass produced in each of the component size classes needs to be known in order to calculate nitrogen uptake.

This chapter sets out to determine the dry matter production of the trees during the experiment. Nitrogen accumulation in the biomass will be discussed in Chapter 4. Biomass production assessment using the component size classes as described by Young and Carpenter (1975) was seen to be an appropriate method.

2.1.7 Root production

Here, both the root mass and root distribution were measured. Root mass is important for the calculation of the nitrogen budgets, whereas root distribution provides information on uptake of water and nitrogen throughout the soil profile. To estimate root mass, complete excavation of the roots from the soil is essential, as it provides measurement of the few very large roots, which in perennial species account for much of the total root biomass (Mackie-Dawson and Atkinson, 1991). However, the loss of fine roots during excavation (Atkinson, 1985), means that excavation methods cannot

be used to estimate fine root production. Fine roots (under 2 mm in diameter) are of particular interest as they represent the major absorbing surface (Dickmann et al., 1996). Root length is a better indicator of nutrient absorbing capacity than mass as it is more sensitive to soil factors (Atkinson and Chauhan, 1987). Fine root mass and root length density are most frequently measured following extraction of soil cores (Mackie-Dawson and Atkinson, 1991). In tree species with relatively low root length densities and substantial variation, large numbers of core samples may be needed to accurately compare root density at different depths (Atkinson, 1985). In this experiment root mass was estimated using both root excavation and root coring methods.

2.1.8 Above ground biomass production

To evaluate biomass production during the course of the experiment required a non-destructive method of measuring biomass production. Leaf area provides an indication of the tree growth, as light interception and utilisation are central to growth. Thus the relationship between the leaf area of a plantation and site attributes is critical to determine growth (Battaglia et al., 1998). In this experiment leaf area measurements were used as a guide to the biomass production of the trees through to the final harvest.

2.2 Methods

2.2.1 Experimental design

Twelve, in-ground lysimeters containing 3 replicates of 3 tree species (*Eucalyptus saligna, E. nitens*, and *Salix kinuyanagi*), plus 1 bare-soil treatment, were established in a field at Aokautere, near Palmerston North, New Zealand. The lysimeters (1.8 m diameter, 1.0 m deep) were filled with Manawatu fine sandy loam to a depth of 0.65 m (weathered, Fluvial, recent, Hewitt, 1993) on top of 0.35 m of river gravel. The lysimeters were repacked to the original bulk density (1.2 Mg m^{-3}) to simulate the local soil profile. Soil properties have been described by Clothier et al. (1977).

A single tree was planted in each lysimeter, in November of 1994. The evergreen Eucalypts were planted as 3-month old seedlings, and the deciduous willows as unrooted cuttings. The control lysimeters contained only soil. Trees of the same species were planted around the lysimeters at approximately 4000 stems ha^{-1} so as to create conditions approximating a small plantation. Surrounding 'guard trees' were not irrigated. All the lysimeters were left for a period of one year before the experiment began. This period allowed the trees to establish themselves and for the soil to settle into the repacked lysimeters. During this establishment phase, the lysimeters received only natural rainfall and no irrigation. Data collection began in December 1995 when the trees were one-year old, and measurements continued through until September 1997. Two weeks prior to data collection all the lysimeters were irrigated to raise the water content of the soil. The above ground biomass of the trees was finally harvested in April 1997. The rotation time of 2.5 years is within the 2-10 year range expected for SRF crops grown for land treatment of dairy-farm effluent (Ranney et al. 1987). During the

course of the experiment lysimeters were maintained weed free via hand weeding. The *E. saligna* trees were sprayed once to control *Ophelimus eucalypti* (gall wasp).

Rainfall during the 620 day experiment totaled 1830 mm. Effluent irrigation was applied over two irrigation periods using sprinkler irrigation (Figure 2.1). Hydraulic loading during the application periods was 21.5 mm $week^{-1}$. In the first irrigation period a total of 618 mm was applied from December 5, 1995 to June 16, 1996. In the second irrigation period, 670 mm was applied from September 16, 1996 to April 14, 1997. Further details on effluent application and hydraulic loading can be found in Sections 3.2.2, 3.3.1 and 3.3.2. Nitrogen loading during the first irrigation period totaled 70 g N $lysimeter^{-1}$ (equivalent to 279 kg N ha^{-1}). Nitrogen loading during the second irrigation period totaled 148 g N $lysimeter^{-1}$ (equivalent to 592 kg N ha^{-1}). Thus total application during the experiment totaled to 871 kg N ha^{-1}. Further information on nutrient loading is available in section 4.3.1.

Figure 2.1 An *E. saligna* lysimeter showing the irrigation sprinklers. The lysimeters were 1.0 m deep and 1.8 m in diameter.

2.2.2 Above ground biomass production

2.2.2.1 Leaf area

Leaf area was measured during the course of the experiment via leaf counts and leaf sample collection. Leaf counts were carried out in January, October and December of 1996, and in February of 1997 summing the total number of leaves on each tree. Leaf samples were collected randomly at the time of these counts. The leaf area of the sample leaves was measured using a Li-3100 area meter (Licor-inc, Lincoln, NE). The total leaf area of the tree was then estimated from the total leaf count multiplied by the average measured leaf area. Total leaf area was measured at the time of harvest by stripping all leaves from the trees. The leaf area of all leaves was then measured with the leaf area meter.

2.2.2.2 Total above-ground biomass production

Guard trees surrounding the lysimeters were all felled on 21 April 1997. Trees were cut approximately 0.1 m above the ground using a chainsaw. Figure 2.2 shows the size of the trees in the lysimeters on the day of harvest. The lysimeter trees were felled over a period of 17 days (Table 2.1), due to the labor intensive data collection requirements at harvest. Harvested biomass from each tree was separated into 4 groups for determination of dry matter production and nitrogen content, following the methods of Young and Carpenter (1975). The woody-stem biomass was separated into 3 size groups based on diameters being; < 6.4 mm, 6.4-25 mm, and > 25 mm. Leaves were the fourth category of above-ground biomass. Biomass in each component category was oven dried at 70°C until no weight change was observed after a 24 hour period.

Table 2.1 Harvest dates of the nine trees in the experiment.

Replicate	Lysimeter number	Species	Date of harvest
1	2	*E. nitens*	1 May 1997
1	3	*S. kimuyanagi*	8 May 1997
1	4	*E. saligna*	29 April 1997
2	6	*E. nitens*	24 April 1997
2	7	*S. kimuyanagi*	24 April 1997
2	8	*E. saligna*	22 April 1997
3	10	*E. nitens*	2 May 1997
3	11	*S. kimuyanagi*	4 May 1997
3	12	*E. saligna*	4 May 1997

Figure 2.2 The four treatments, photographed following the felling and removal of the guard trees. Stumps of the harvested guard trees can be seen in the foreground. Trees age 2.5 years.

2.2.3 Root production

Root production was monitored in two ways: larger roots by root excavation; fine roots mass and root length density by coring methods. Root mass could only be measured at the cessation of the experiment due to the destructive nature of the sampling methods. Thus root system analysis took place in August and September 1997.

2.2.3.1 Excavation of roots

Replicate 3 of each tree treatment was excavated in September 1997. A digger was used to extract the cut stump (above-ground) and main root system. Soil remaining in each lysimeter was removed manually and spread over an area nearby, then raked to gather roots. Collected roots were washed and weighed. The stump was separated for wet and dry weight determination. Sub-samples were collected from the root system, following the method of Young and Carpenter (1975). These sub samples reflected three size groups based on diameters being <6.4 mm, 6.4-25 mm, and >25 mm. Wet and oven dry weights of each category were measured using sub-samples. The range of dry-matter production for roots was calculated using the average water content of the sub-samples and the total wet weight of roots excavated.

2.2.3.2 Root cores

In August 1997, four soil cores of diameter 46 mm to a depth of 0.6-0.7 m were taken from each lysimeter. The maximum depth was limited to 0.6-0.7 m due to stones preventing the corer getting any deeper. Core samples were taken at 0.2, 0.4, 1.4, and 1.6 m across the 1.8 m diameter of the lysimeter. Cores were also attempted at 0.6 m and 1.2 m however the predominance of large roots prevented these cores from being taken. An engine-powered concrete breaker was used to drive coring tubes into the soil with a tripod and winch being used to extract them (Welbank and Williams, 1968). Each

core was cut into 0.1 m lengths. Roots were extracted from each 0.1 m cylinder of soil using a root washing machine designed by Smucker et al. (1982). Roots greater than 2 mm diameter were removed from the samples (Dickmann et al., 1996; Rytter and Hannson, 1996). Total root length of each sample was measured using an automatic root-length scanner. The roots were oven dried and their mass recorded. The root length density (RLD) was calculated by dividing total root length by the volume of the core sample. The mean and standard deviations from 12 cores were calculated for *E. nitens* and *S. kinuyanagi* and for *E. saligna* using 8 cores. Multiplying the mean root density by depth zone volume provided estimates of fine root production for each of the three treatments.

Figure 2.3 A root core collected from the soil profile of an *E. saligna* lysimeter.

2.3 Results

E. saligna is represented by 2 replicates in the analysis of the lysimeter experiment. The third replicate suffered wind-snap, which removed over half of the biomass of the tree in February 1996. Ironically, the same lysimeter also had a faulty leachate recorder that made water balance calculations unattainable for this lysimeter for most of the experiment.

2.3.1 Leaf area development

In January 1996 the *E. nitens* mean leaf area of 13.5 m^2 was nearly double the size of the *S. kimuyanagi* at 7.7 m^2, with *E. saligna* only 4.8 m^2 (Figure 2.4). No further measurements of leaf area were collected until October 1996. Leaf area development during this stage is discussed further in Section 2.4.1. Both Eucalyptus species showed a decrease in leaf area from December 1996 to the harvest period in April 1997. The decline of mean leaf area of *E. nitens* of over 16 m^2 was greater than the almost 4 m^2 of the *E. saligna*, but neither of these declines was statistically significant (P=0.05). *S. kimuyanagi* shed all leaves by late May and started to produce leaves in early August of 1996. Following a significant increase (P=0.05) in mean leaf area from August (0 m^2) to October (27 m^2) 1996 the willows leaf area then decreased significantly (P=0.05) between December 1996 (29 m^2) and February 1997 (7.7 m^2). This significant reduction in leaf area was visually obvious. Large numbers of dry leaves dropped from the trees around the end of January 1997, following several weeks of warm weather and with no rainfall.

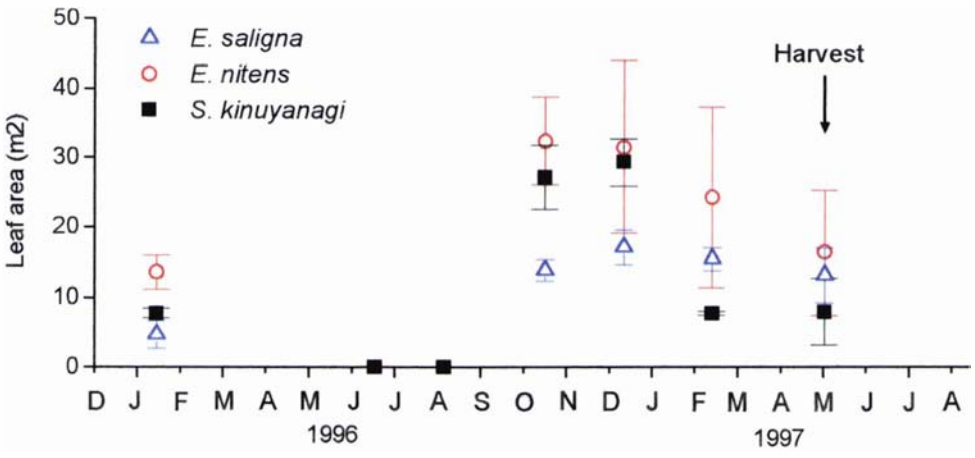

Figure 2.4 Mean leaf area of the three tree species as determined by leaf counts from individual trees. Vertical bars represent the standard deviation.

There was a higher variability of the leaf areas between the replicates of *E. nitens* at the initial stage of the experiment in comparison to the other species. Throughout the experiment this variability increased reflecting one tree growing fast, as a result of some of the guard trees around the lysimeter dying hence giving less competition for light.

2.3.2 Above ground biomass production

Average heights at harvest were 4.6 m, 6.7 m, and 4.3 m for *E. saligna*, *E. nitens*, and *S. kinuyanagi* respectively. The heights were influenced by the formation of double stems near ground level for the three *S. kinuyanagi* trees and one of the *E. nitens*.

The above-ground biomass production of each species is shown in Table 2.2. The mean total annual biomass production during the 2.5 years, based on a stocking density of 4000 stems hectare^{-1} was equivalent to 15.6 Mg ha^{-1} (*E. saligna*), 30.6 Mg ha^{-1} (*E. nitens*) and 21.3 Mg ha^{-1} (*S. kinuyanagi*). *E. nitens* produced significantly more above-ground biomass than *E. saligna* with *S. kinuyanagi* being intermediary. The annual mean woody biomass production in the 2.5 years, based on stocking at 4000 stems ha^{-1} was equivalent to 11.9 Mg ha^{-1} (*E. saligna*), 24.9 Mg ha^{-1} (*E. nitens*) and 19.5 Mg ha^{-1}

(*S. kimuyanagi*). Woody biomass production of *E. nitens* and *S. kimuyanagi* was significantly greater than *E. saligna* (P=0.05). *E. nitens* produced significantly greater mass of large stems than *E. saligna* and *S. kimuyanagi* (P=0.05). Leaf production was variable between the species, with *E. nitens* being the largest producing species. Large stem production comprised 47 % for *E. nitens* in comparison to 37 % for *E. saligna* and 40 % for *S. kimuyanagi*. Distribution of mass between the tree components differed between the species. These results are discussed further in section 2.3.4 where the whole tree biomass production is shown.

All trees were observed to have produced coppice shoots in September 1997, four months after the harvest.

Table 2.2 Mean dry matter yield (kg tree^{-1}) of above-ground components of three tree species harvested at 2.5 years old.

Tree component	*E. saligna*	*E. nitens*	*S. kimuyanagi*
Small stems[1]	1.25	1.79	1.85
Medium stems[1]	1.92	3.32	4.23
Large stems[1]	3.59	8.97	5.30
Bark from large stems[1]	0.68	1.46	0.82
Total stems and bark	7.44	15.54	12.20
Total leaves	2.29	3.60	1.13
Total above ground	**9.73**	**19.14**	**13.33**

[1]Stem size groups refer to diameter of woody biomass (refer to text for details).

2.3.3 Root production

2.3.3.1 Root excavation

Excavation of roots revealed large numbers accumulating around the edge of the lysimeters for all treatments indicating roots were restricted by the lysimeter boundaries. Visual observations confirmed this to be more pronounced in the *E. nitens* treatments. Whilst raking the excavated soil for roots, observations showed the roots were

predominately situated in the soil zone (above the gravel zone) and around the edges of the lysimeter. The *E. nitens* produced the greatest root mass being approximately 1.5 and 2 times greater than the *S. kimuyanagi* and *E. saligna* treatments (Table 2.3).

Table 2.3 Root and stump dry matter production (kg tree^{-1}).

Species	Group	Dry weight
E. saligna	Stump	0.58
	Roots	2.04
E. nitens	Stump	0.84
	Roots	3.79
S. kimuyanagi	Stump	0.44
	Roots	2.57

2.3.3.2 Root cores

All root core sub-samples contained roots. Fine roots of less than 2 mm diameter were present in similar amounts between the treatments, being estimated at 242, 264 and 273 g lysimeter^{-1} for *E. saligna*, *E. nitens* and *S. kimuyanagi*. Fine root length density (RLD) and root mass were close to uniform with depth throughout the *E. saligna* lysimeters (Figure 2.5). *E. saligna* showed slightly greater amounts of roots at the top of the lysimeter and above the gravel layer. *E. nitens* showed a similar pattern to the *E. saligna*. However for *E. nitens* the increase in the abundance of roots in the surface 0.1 m of soil and just above the gravel layer was more pronounced (Figure 2.5). The *S. kimuyanagi* fine root system is concentrated in the surface layer and decreases sharply into the 0.1-0.2 m layer and declines further with depth (Figure 2.5).

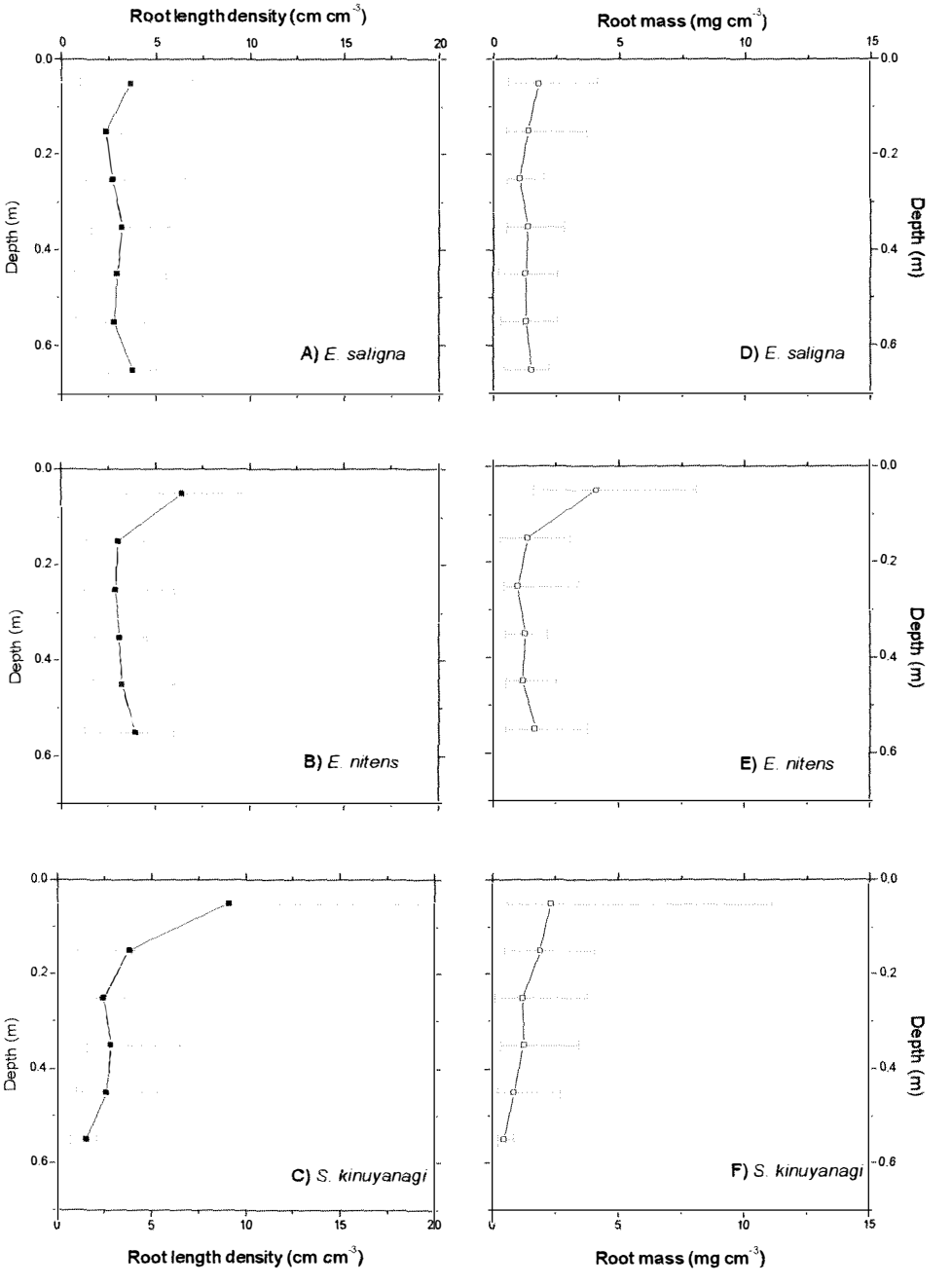

Figure 2.5 Root length density and root mass of fine roots as estimated by soil coring. Means and confidence intervals for each treatment are presented.

2.3.4 Biomass allocation

Both above and below ground biomass production are combined in Table 2.4. The main difference in the biomass production between *S. kinuyanagi* and *E. nitens* was the large stem production. Figure 2.6a shows the mass produced for all the subgroups of tree

anatomy. Figure 2.6b shows the percentage allocation of dry matter production in the sub classes of tree anatomy. Root production is consistent at around 17 % for all trees. Large stem percentage ranges from 33-40 % of the total biomass of trees, with medium and small stems making up a further 21-36 % of the tree. Foliage allocation at the time of harvest is much lower for the deciduous willow (7 %) in comparison to the evergreen Eucalypts (15-18 %).

Table 2.4 Total biomass production of the tree components and the whole tree (kg tree^{-1})

Tree component	*E. saligna*	*E. nitens*	*S. kinuyanagi*
Roots[1]	2.28	4.06	2.84
Stems[2]	4.86	11.27	6.57
Branches[3]	3.18	5.11	6.07
Foliage	2.29	3.60	1.13
Total whole tree	**12.61**	**24.04**	**16.61**

[1]Estimated as the mean value of the total roots excavated plus the estimate of fine roots from soil cores
[2]Includes stump and bark components.
[3]The sum of small and medium stems.

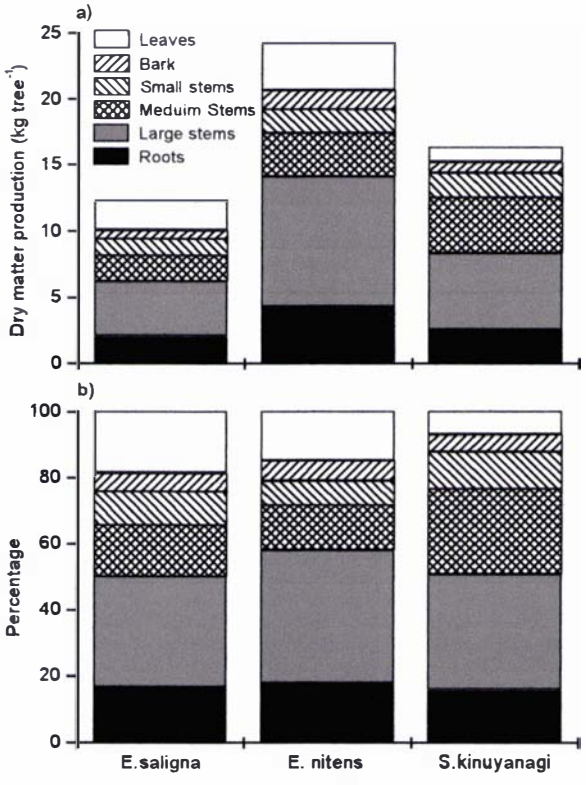

Figure 2.6 Total mass produced by the trees separated into major tree components expressed as a) dry matter production and b) percentage allocation

2.4 Discussion

2.4.1 Leaf area development

Leaf areas of the tree species differed early in the experiment reflecting the different growth rates of the trees in the first year during establishment (November 1994 - November 1995). In this initial stage no irrigation was applied and tree growth may have been limited by water availability during this period. The *E. saligna* and *S. kimuyanagi* showed similar leaf area at this initial stage in the experiment. The greater leaf area of *E. nitens* indicates better growth than the other species during the establishment period. *E. nitens* fast establishment has been observed at other sites (Nicholas, 1997; Senelwa, 1997). Fast establishing species are desirable as these species compete better with weed species in the establishment phase.

Leaf areas were not measured during the period between January 1996 and October 1996. With no information, the leaf area development during this period can only be speculated. The evergreen trees probably increased their leaf area more rapidly in the summer than in the winter to reach the points measured in October 1996. It is expected that the leaf area of the *S. kimuyanagi* increased from January 1996 to May 1996 after which it started to shed it leaves. The *S. kimuyanagi* were noted to have shed all leaves by mid June.

From September 1996 to the harvest in April 1997, the leaf area of *E. saligna* did not vary significantly over time (P=0.05). Although this was not significant, the *E. saligna* leaf area may have been adjusting according to the seasonal conditions. That is the leaf area of *E. saligna* may have increased from spring through to summer and declined

slightly in autumn. The leaf area of the *E. saligna* was greatest in December 1996 at 17 m^2 and decreased to be to 15 m^2 in February 1997 and 13 m^2 in April 1997.

From September 1996 to the harvest in April 1997, *S. kimuyanagi* leaf area showed a decline. With a significant drop in leaf area between December 1996 and January 1997 likely to have been in response to water availability. Despite weekly irrigations of 21 mm on top of rainfall inputs during the summer period, the soil water content in willow treatments was continuously much below (10.5 m^3 m^{-3}) 'field capacity' (approximately 40 m^3 m^{-3}) (Figure 3.7). A shedding of dried leaves was visually observed for the *S. kimuyanagi* in January 1997. This indicates that the willow trees had dried the soil close to permanent wilting point. Permanent wilting point is the point when plants can no longer recover overnight from wilting during the day (Sasse and Sands, 1996).

E. nitens showed a downward trend in leaf area from September 1996 to April 1997, however there was no significant change in leaf area during this period (P=0.05). Despite weekly irrigations of 21 mm on top of rainfall inputs during the summer the soil water content of *E. nitens* treatments was continually low (8.8 m^3 m^{-3}) (Figure 3.6). The period of low water availability was longer for *E. nitens* in comparison to *S. kimuyanagi*. However no dramatic leaf drop was observed for the *E. nitens* trees.

The different responses of the *S. kimuyanagi* and *E. nitens* indicate differing tolerances to water stress. Inputs from rainfall and effluent were similar for both species and the soil water availability was greater for the *S. kimuyanagi* going into the summer period. *S. kimuyanagi* trees nearby the experiment also showed some shedding of leaves due to water stress indicating that this was a natural response of these trees to the hot dry

conditions of the time. Eucalypt species have been observed to vary leaf area in response to water stress (Sharma, 1984). The differing responses of the tree species to water stress are further discussed with the water balance of the site, in Chapter 3.

2.4.2 Above ground biomass production

There are several sources of error associated with scaling up biomass production estimates from small plots. The lysimeters provide a sealed environment from which all inputs and outputs can be monitored. In doing so, they limited the distribution of the roots and limited competition from roots of neighboring trees for resources. Weeds were also controlled limiting their competition for resources with these trees. Biomass production per hectare was estimated from the three lysimeter-grown trees. These should however be treated with caution. Total annual production estimates of the above-ground biomass over the 2.5 years, based on stocking at 4000 stems ha^{-1} was equivalent to 15.6 Mg ha^{-1}, 30.6 Mg ha^{-1}, and 21.3 Mg ha^{-1} for *E. saligna*, *E. nitens*, and *S. kimuyanagi* respectively. The measured biomass production for all species fall within the ranges measured previously for the individual species in New Zealand.

The measured biomass production for the *E. saligna* is close to the highest value recorded for this species in New Zealand (16 Mg ha^{-1} yr^{-1}) (Frederick et al., 1985). However those trees were grown at 829 stems ha^{-1} and Frederick et al. (1985) suggested that closer spacing could increase production. In the current study, biomass production of *E. saligna* was however thought to be limited due to infestation by *Ophelimus eucalypti*, a Eulophid wasp which causes galls to form on the leaves by ovipositing in the leaf (Senelwa, 1997). It can lead to the complete defoliation and the death of well established trees after 4-5 years in areas where the pest population is high. The infestation reduces the photosynthetic surfaces of the leaves, and hence the

photosynthetic capacity. Therefore the metabolic activities of the plant are restricted as is growth. The effect of the pest in this experiment was expected to be similar to that described by Senelwa (1997). Senelwa (1997) concluded that although the biology, ecology and effect of the pest on the growth of trees is inconclusive, the infestation was believed to have resulted in the generally low yields of *E. saligna* in comparison to other species in the study. It was difficult to ascertain whether the chemical treatment used in this experiment impacted on the gall wasp significantly.

At the start of the experiment the leaf areas of the *E. nitens* trees were nearly double and triple that of the *S. kimuyanagi* and *E. saligna* trees respectively. At the harvest, biomass production showed a similar trend. It is likely the growth of both *S. kimuyanagi* and *E. nitens* was limited by water stress during the experiment and better irrigation management may have enhanced the growth of these trees. The limiting factor to *E. saligna* was suspected to be the gall wasp.

2.4.3 Root production

The sealed lysimeters restricted the trees from spreading roots to depth or horizontally through the soil. The presence of large numbers of roots circling the edge of the lysimeter was thought to be the result of restriction on lateral roots. The concentration of roots in the area may also be due to preferential flow down the edge of the lysimeter of rainfall and effluent. This however is unlikely, as effluent application was applied over the surface with a clear buffer zone toward the edges. Furthermore, the soil surface was banked up toward the perimeter of the lysimeter to prevent edge flow. As expected root mass of the trees was reflective of the above-ground biomass production, with mass of roots being greater for *E. nitens* than *S. kimuyanagi*, with *E. saligna* producing the

least amount of roots. The larger production of roots by *E. nitens* is likely to explain the visual observation of more roots wrapping around the lysimeter edges.

Eucalypt trees have been shown to gather water from depths further than 8 m below the soil surface (Dye, 1996). *E. nitens* is reported to grow best when it can form a relatively deep rooting system with an effective rooting system of at least 0.6 m depth (Miller et al., 1992). Willows too can gather water from depth, but are better known for their mat like root systems in the surface horizon which prove valuable for erosion control (van Kraayenoord et al., 1995). It is likely that the restriction of roots in the lysimeters has restricted the amount of available water to the trees.

Irrigated trees are known to concentrate roots in the surface layer of the soil (Sims et al., 1994; Rytter and Hansson, 1996). This study is consistent with such observations with all species showing a concentration of fine roots in the surface layer of the soil.

Rooting depth is also known to alter seasonally and in response to water and nutrient supply (Dickmann et al., 1996; Rytter and Hansson, 1996). Thus the timing of the root cores and extraction only presents a 'snapshot' of the root system. The root cores were collected in August 1997, four months after the coppicing of the trees in April. Although the silvicultural practice of coppicing is well known, virtually nothing is known about root-system responses to coppicing (Dickmann et al., 1996). Following coppicing of effluent irrigated Eucalypts, a die back of roots was observed by Sims et al. (1994). However, Dickmann et al. (1996) found 2 poplar clones to show substantial fine-root production in the spring immediately following coppicing, with no evidence of

shock induced die-back of roots. The timing of the root sampling thus does raise some questions as to how the results relate to the distributions during the study.

The distribution of fine roots has implications for uptake of water and nutrients. If fine root distributions are taken as indicative of total root distribution during the growth of the trees, the patterns readily fit the water status of the trees. Water inputs were all via surface application. All trees correspondingly showed greater root length densities in the surface layers. The *E. saligna* trees had water available from the soil profile throughout the experiment but all inputs were from the surface layer. Correspondingly the fine roots of *E. saligna* were spread very evenly through the top 0.6 m of the soil profile with some concentration in the surface layer. The *E. nitens* fine root network was concentrated just above the gravel layer and at the top of the soil profile. *S. kimuyanagi* fine root mass and root length densities were far greater in the surface 0.1 m and declined with depth. This indicates high uptake in the surface layers and is consistent with the extensive mat-like root systems of willows previously reported (van Kraayenoord, 1995).

The analysis of root core samples is known to have difficulties with interpretation of the absence of roots in some samples (Mackie-Dawson and Atkinson, 1991). In this experiment, such problems were not encountered as all core samples contained some roots. The presence of roots in all core samples may reflect a spring flush of fine roots following the coppicing in April. An alternative explanation to the presence of roots in each sample is the restriction of the tree root systems by the lysimeter. Thus roots that would in a field situation spread throughout a greater volume of soil were concentrated in a small soil volume in the lysimeter. However, the range of values for the willows in

this experiment to a depth of 0.5 m (41-223 g m^{-3}) compare well with measurements for 5 year old irrigated and fertilised willows in Sweden (83.1-170.2 g m^{-3}) (Rytter and Hansson, 1996).

Despite the problems with root distribution in the lysimeters, this system of experimentation is invaluable because of the control it offers in allowing the detailed measurement of the leachate loadings.

2.4.4 Biomass allocation

As expected the whole tree biomass production followed the same pattern of the above ground biomass. The smaller production of leaves by the *S. kimuyanagi* is likely a result of leaf drop in January 1997 carrying over to the harvest of the trees. This may have consequences for the storage of nitrogen at the harvest time, which will be discussed further in the nitrogen chapter (Section 4.4.1.3)

The allocation of biomass in the trees' above-ground components has implications for the usage of these trees for fuelwood production. Greater proportions of stem wood may be advantageous for fuelwood production due to easier handling and greater amounts of saleable product. The proportion of large stems was lower for *S. kimuyanagi* than the *E. nitens*, partly as a result of the formation of double stems at or near ground level. Formation of double stems only occurred for one *E. nitens* replicate and not at all for the *E. saligna*. Greater proportions of large stem wood may be attained by longer rotation lengths. Following coppicing, however, many sprouts develop from the stump. If fuelwood wood is the desirable product it may be necessary to trim these back to the dominant shoot early in each rotation and/or have a longer rotation time. Before embarking on a SRF system for treating dairy farm effluent, the end use of the final

biomass product must be considered. Silvicultural regimes need to be adjusted accordingly. These aspects of management are beyond the scope of this research.

2.5 Conclusions

The total tree biomass production of the *E. nitens* was significantly greater than *E. saligna*, with *S. kinuyanagi* being intermediary. The woody biomass production, however was significantly greater for both *E. nitens* and *S. kinuyanagi* in comparison to the *E. saligna*. Although scaling up biomass estimates from small plot trials and particularly lysimeters introduces associated errors, the estimates fell within the ranges measured elsewhere in New Zealand. The *E. saligna* was infested by the gall wasp *Ophelimus eucalypti* but the effect of this parasite on growth was difficult to determine and warrants further investigation. If *E. saligna* is to be used for SRF plantations in New Zealand the effect of this pest must be considered. Growth of *E. nitens* and *S. kinuyanagi* was probably limited by water stress. *E. nitens* and *S. kinuyanagi* showed differential responses to low water availability. The *S. kinuyanagi* was observed to shed a significant amount of leaves in response to water stress. Water stress will be further discussed in Chapter 3 with the water balance information for all of the treatments. The measurements of mass presented in this chapter will be used for the calculation of nitrogen storage by the trees in Chapter 4 to enable full nitrogen balances to be calculated.

Chapter 3 Water Balance

3.1 Introduction

Land application of dairy-farm effluent may limit the loss to the environment of this nitrogenous waste. However to be sustainable, it is necessary to determine the fate of the components in the effluent after it has been applied to land. Nitrate leaching to groundwater is an environmental concern because of its potential impact on water quality and even human health. This health risk forms one of the major considerations for a discharge consent as it uses New Zealand drinking water standard that potable water (including potable groundwater) should contain less than 11.3 mg L^{-1} of NO_3^--N (Ministry of Health, 1995). Nitrate is poorly retained by soil and moves freely with soil water. Thus, nitrate leaching occurs if the water inputs (effluent irrigation and precipitation) exceed the soil-water storage and evapotranspiration demand, thereby allowing water that may contain nitrogen to proceed down beyond the rootzone to contaminate the groundwater.

So, understanding water movement forms an integral part of understanding the fate of the nitrogen applied in the liquid effluent. Thus a knowledge of water and nutrient balances of the rootzone is central to the successful design of an environmentally sustainable system of land-treatment (Bond, 1998).

The focus of this project is to develop an understanding of the key processes so that modelling tools can be developed for the prediction of the fate of applied water and nitrogen in SRF land treatment systems. This will enable us to quantify the effects of different loading rates on nitrate leaching thereby aiding the design and management of sustainable land treatment systems. The aim of this chapter is to obtain, the basic water

balance information set. Later this understanding of the water balance will be linked to the dynamics of the nitrogen balance (Chapter 4). This combined information is essential to developing predictive models (Chapter 5).

To resolve the water balance of a rootzone, measurements should include both the inputs and outputs of water plus measures of the soil water storage. Here, the inputs are rainfall and effluent irrigation. The outputs are evapotranspiration (ET), drainage and runoff. The water outputs of leaching and evapotranspiration are of particular interest. Evapotranspiration provides the drive for water uptake into the trees and this strongly influences the extent of drainage to groundwater.

The rates of water use by effluent-irrigated plantations can vary. In coastal New South Wales, Dunin and Aston (1984) reported a maximum water-use of 7 mm day^{-1} in summer for a native Eucalypt forest with non-limiting water availability. Myers et al. (1996) measured maximum daily water use rates of 8.0 mm day^{-1} for 3 year old *E. grandis* trial plots irrigated with effluent at Wagga Wagga, NSW, Australia. From these studies it appears maximum transpiration rates for the Eucalypts will be around 7-8 mm day^{-1}. Tungcul et al. (1996) reported values ranging from 3.8-9.7 mm day^{-1} for *Salix* species receiving effluent application at Aokautere, New Zealand, on a cloud free day in summer. Although, Tungcul et al. (1996) used small plots and thus may have had difficulty defining the ground area to calculate the water use, these results provide an indication of water use of Salix in New Zealand.

Previous studies of effluent-irrigated SRF species have tended just to calculate the drainage or they have simply been unable to quantify it. Myers et al. (1996) carried out

water balance studies of *E. grandis* and *Pinus radiata* irrigated with municipal effluent at Wagga Wagga. In their field-based study, they calculated both ET and drainage. Drainage was however, independently verified via a chloride mass balance. Tungcul et al. (1996) irrigated dairy-farm effluent onto *E. nitens* and 9 species of *Salix* at Aokautere, New Zealand and although they predicted that nitrate leaching was occurring, they were unable to quantify this.

Complete records of both the water and nitrogen balances for land-treatment systems of SRF, under New Zealand conditions, have not been made. This is the goal of this experiment. A lysimeter experiment, aimed to measure water and nitrogen balances of three SRF species was set up. A bare-soil control received the same rate of application of dairy-shed effluent. The objective of this chapter is to report the measurements and to give a general interpretation of the water balance obtained from this lysimeter experiment.

3.2 Methods

3.2.1 Experimental design

The overall experimental design is discussed in Section 2.2.1. Here the methodologies utilised in relation to aspects of water balance are more fully discussed. The lysimeter facility (Figure 3.1) was established to enable measurement of all the components of rootzone water balance. Water inputs are via rainfall and effluent irrigation, water outputs are via ET and drainage. The soil provides a storage buffer.

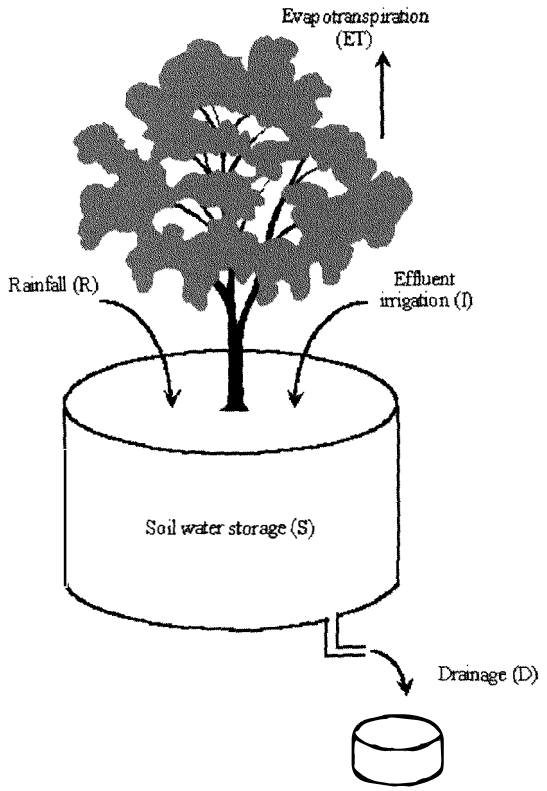

Figure 3.1 Water inputs and outputs of the lysimeter.

3.2.2 Effluent irrigation

Dairy-farm effluent (secondary-pond treated) was applied at weekly intervals during 2 irrigation periods. The first irrigation period was from 5 December 1995 to 16 June 1996, a 29 week period. The second irrigation period was 31 weeks from 16 September 1996 to April 14 1997.

The application rate of effluent was decided to be 21 mm water per week to replace the water use by trees. This rate was based on the assumption that during the application period, on average, the input of rainfall is 2 mm day^{-1}, whereas potential ET is 5 mm day^{-1}. The expected water deficit would therefore be

$$\text{Deficit} = \text{ET} - \text{Rainfall} \qquad [3.1]$$

$$\text{or,} \quad 3 \text{ mm day}^{-1} = 5 \text{ mm day}^{-1} - 2 \text{ mm day}^{-1} \qquad [3.2]$$

Thus the weekly application rate was fixed at 21 mm per week.

Effluent delivery was controlled individually for each lysimeter by a timing device linked to a pump and solenoid valves. An annular pattern of application over the soil surface was achieved through 3 centrally located spray nozzles each irrigating 120 degrees of the lysimeter. A buffer zone of 0.1 m was maintained around the edge to try and avoid preferential flow down the sides of the lysimeters. Effluent was applied over a 3-hour period each week during the application periods. To ensure that the application rate did not exceed the soils infiltration rate, the timing device would turn the pump on and off to deliver pulses of effluent in a 5 minute cycle. The cycle consisted of turning the pump on for thirty seconds of application, which was followed by 4.5 minutes with the pump turned off. Immediately following the complete application of effluent the irrigation system was cleansed by pumping bore water through the lines for 2 cycles of the timer. This prevented stale effluent being applied the following week. Also, this flush cleansed the pipes to reduce the chance of a build up of algae in the pipes.

3.2.3 Rainfall

Rainfall at the site was logged by a rainguage from May 1996. Data from a nearby Automatic Weather Station, 4 km from the site, was used to fill in the missing rainfall values of periods not measured at the field site at Aokautere.

3.2.3.1 Effective rainfall

To estimate accurately the water balance of the rootzone the amount of rainfall reaching the soil surface must be known. Of the total rainfall (R_t) falling above the forest canopy a proportion will fall onto leaf and stem surfaces and be directly evaporated. This loss is termed the intercepted rainfall. The remainder of the rainfall reaches the soil surface either by falling through the canopy (throughfall) or by flow down the stem (stemflow). Thus the effective rainfall (R_e) is the amount of rainfall that reaches the soil surface during a rainfall event via either stemflow and/or throughfall. Although the hourly rainfall was recorded using a weather station, the effective rainfall reaching the lysimeter surface was not measured during this experiment. Thus appropriate values from the literature were sought for use in calculation of the water balance.

Stemflow

The overall contribution of stemflow to the water balance was low. Lloyd et al. (1988) measured a stem flow rate of 3.6 % of total rainfall for a tropical forest and they concluded that although variability of stemflow is very high, its overall contribution to the water balance is very low. Helvey and Patric (1965) reviewed 13 studies of stemflow and they found a range of 0.5 to 8.7 % of total rainfall to be stemflow in deciduous hardwood forests of the eastern United States. Feller (1981) reviewed 10 studies of stemflow in Eucalypt forests. The values ranged from 0.5 to 7.5 % of total rainfall. Stemflows of mature Eucalypt and *Pinus radiata* trees have been measured at

1.9 and 3.6 % of total rainfall, respectively (Smith, 1974). Myers and Talsma (1992) showed the relative dominance of interception relative to stemflow in the water balance of forests. The relationship they found between rainfall event size and net interception for 10–14 year old pines showed that as the rainfall event size increased from 3 to 50 mm, total rainfall intercepted decreased from 60 % to 10 %, whereas the stemflow increased from 0 % to 6 %. Thus for this experiment stemflow was not included because its contribution was probably minor relative to other components of the water budget. For this experiment effective rainfall is assumed equal to the throughfall, which is calculated as the difference between the total rainfall and interception.

Rainfall Interception

The amount of rainfall not reaching the soil surface is termed intercepted rainfall. In this experiment the intercepted rainfall is equal to the total rainfall less the throughfall. Both tree architecture and environmental influences can be important in determining interception loss (Aston, 1979; Dunin and Aston, 1984; Teklehaimanot and Jarvis 1991). Canopy water storage increases with greater foliage mass. Thus interception rates are related to leaf area (Aston, 1979; Myers, 1992). For forests, the relationships between annual effective rainfall and annual total rainfall have been found to be remarkably constant (Helvey and Patric, 1965; Zinke, 1967; Rutter et al., 1971; Smith, 1974). A review of 16 interception studies for *Eucalyptus* species revealed throughfall values ranged from 68-95 % of the incident precipitation (Feller, 1981). The conclusion of this review by Feller (1981) was that rainfall interception is usually of the order of 10-25 % of total rainfall for Eucalypt forests. Dunin and Aston (1984) found rainfall interception to vary between 10-15 % of total rainfall in a 10 year old regenerating Eucalypt forest. Helvey and Patric (1965) reviewed rainfall interception by deciduous

hardwoods of the eastern United States. In the 19 studies they listed throughfall was found to vary from 68-96 % of total rainfall, and they concluded that, on average rainfall interception was around 10 % of total rainfall (Helvey and Patric, 1965).

Such regressions to gain an interception percentage take no account of the rainfall intensity and duration nor the intervals between the storms (Jackson, 1975). To address such factors, more complex models of the rainfall interception process have been developed (Rutter et al., 1971; Rutter, 1975; Rutter and Morton, 1977; Gash, 1979; Aston, 1979; Lloyd et al., 1988; Asdak et al., 1998). These models require many input measurements, some of which were not obtained in this experiment. Regressions, however, will on average be useful estimators of rainfall interception. The use of weekly time-steps for calculation of effective rainfall may limit some of the influence of rainfall intensity, duration and intervals between storms.

During the lysimeter experiment, the trees were growing, and so it is likely that the rates of throughfall were decreasing as the trees gained more leaves. Thus the proportion of rainfall reaching the soil may have been greater in the first season, as the trees were smaller and had not reached canopy closure. The leaf area of the willows is also dynamic, with the shedding of leaves in the winter months. Values from the literature consistently suggest 10-25 % of annual rainfall will be intercepted.

In the experiment, rainfall interception for the evergreen trees was assumed to be 10-15 % of total rainfall. Values of 10 % were used up until canopy closure in November 1996. Post canopy-closure, the Eucalypt rainfall interception was assumed to be 15 %.

The *S. kinuyanagi* (deciduous) treatments were assumed to have rainfall interception of 10 % of total rainfall until June 1996. During the period of leaf shedding in the winter months the rainfall interception was assumed to be 3 % based on the interception of some rainfall by the remaining trunks and branches. At canopy closure in October 1996, rainfall interception was assumed to have increased to 15 %. Following the leaf drop in January 1997 (Figure 2.4), rainfall interception was considered to decrease to 10 %. All treatments received 100 % of total rainfall following the harvest of the trees in April 1997.

3.2.4 Soil water storage

Soil water storage was measured using Time Domain Reflectometry (TDR) (Topp, 1982). TDR probes were installed at 5 depths down the soil profile. Changes in soil water storage could also be monitored by the weight change of the four weighed lysimeters (Edwards, 1986). The TDR probes were installed in different configurations in the weighed lysimeters and static lysimeters (Figure 3.2). The weighed lysimeters were inaccessible from the side, to a depth of 700 mm. The weighed lysimeters had 4 TDR probes inserted vertically from the soil surface down to depths of 100, 250 and 500 mm (Figure 3.2a). The fourth vertical probe, a replicate, was placed at a depth of 250 mm. A fifth probe was inserted horizontally at a depth of 750 mm. The eight static lysimeters had two TDR probes inserted vertically to a depth of 100 mm and another to 250 mm. The remaining three probes were inserted horizontally at depths 250, 500 and 750 mm beneath the soil surface (Figure 3.2b). The lysimeter soil water storage was calculated from these TDR data which were collected at least 5 times per week.

Soil water content and bulk electrical conductivity data for the individual probes were collected and analysed using software developed by Dr S Green (HortResearch

Palmerston North, NZ). Lysimeter soil water content was calculated from the measured TDR data using the following procedures.

The total depth of water stored in each weighed lysimeter, W (mm), was calculated as

$$W = \int_{0}^{1000} \theta(z)\,dz \qquad [3.3]$$

This was then calculated from the TDR data as

$$W = \theta_1 \Delta Z_1 + \theta_2 \Delta Z_2 \qquad [3.4]$$

where θ_1 is the average water content of the upper 500 mm layer of soil ($\Delta Z_1 = 500$ mm) and θ_2 is the average water content of the lower 500 mm layer of soil ($\Delta Z_2 = 500$ mm). Here, θ_1 is measured from the 0–500 mm depth TDR probe (V 0-500) and θ_2 is measured from the 750 mm horizontal probe (H 750).

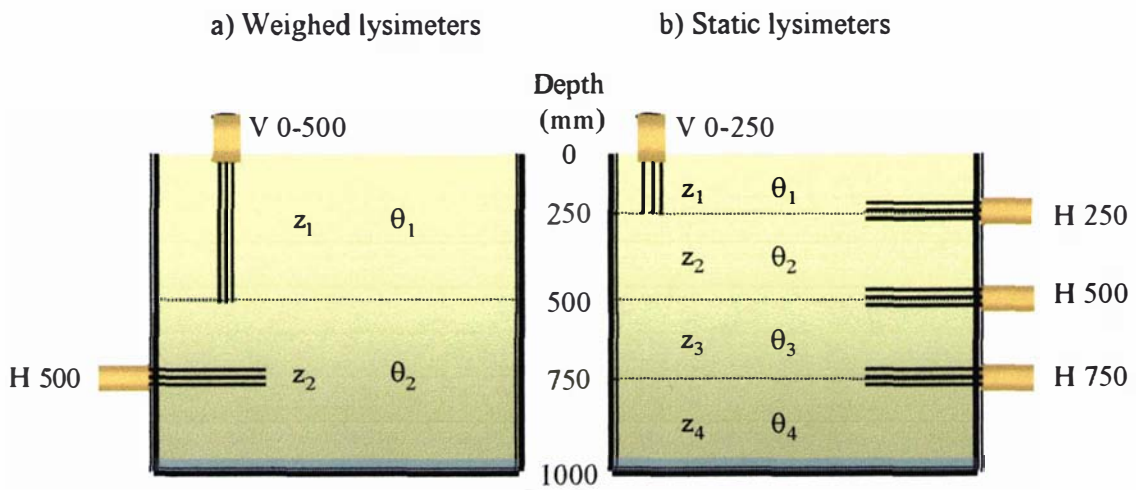

Figure 3.2 Lysimeters showing TDR instrumentation and the layers of soil used for calculation of total lysimeter soil water content (see text for details). Probes are labeled by their orientation (V for vertical, H for horizontal) and their depth beneath the soil surface (mm) a) The layout of a weighed lysimeter b) The layout of a static lysimeter.

The total depth of water stored in each of the static lysimeters, W (mm), was calculated by Eq. [3.3], which for the static lysimeters is equivalent to

$$W = \theta_1 \Delta Z_1 + \theta_2 \Delta Z_2 + \theta_3 \Delta Z_3 + \theta_4 \Delta Z_4 \qquad [3.5]$$

Here the ΔZ values are all 250 mm, and the depths are shown in Figure 3.2. The θ values for the depths as defined in Figure 3.2, and these were calculated from the TDR data from the following relationships

$$\theta_1 = \theta_{V\,0\text{-}250} \qquad [3.6]$$

$$\theta_2 = (\theta_{H\,250} + \theta_{H\,500}) / 2 \qquad [3.7]$$

$$\theta_3 = (\theta_{H\,500} + \theta_{H\,750}) / 2 \qquad [3.8]$$

$$\theta_4 = \theta_{H\,750} \qquad [3.9]$$

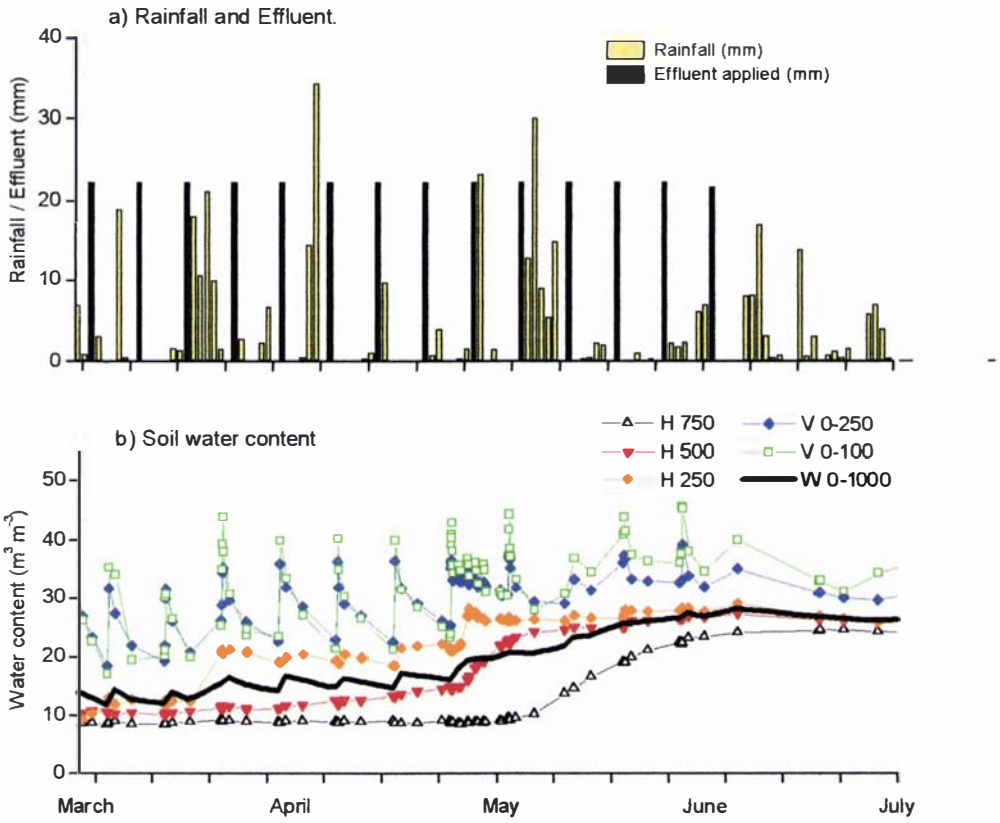

Figure 3.3 a) Water inputs from rainfall and effluent b) Soil water content data from TDR measurements of an *E. nitens* lysimeter. Probes are labeled by their orientation (V for vertical, H for horizontal) and their depth beneath the soil surface (mm). The total water store (W) for the lysimeter is also shown in relation to the separate probe measurements.

As an example of the TDR data and total water storage (W) results for an *E. nitens* treatment over the autumn period are shown in Figure 3.3. The graphs show the water contents measured at different depths in the soil profile in relation to effluent application and rainfall.

3.2.5 Drainage

The packing of river gravel into the base of the lysimeters enabled free drainage to occur beneath the soil profile. Each lysimeter had a single drainage pipe at the base. During the period December 1995 until October 1996, drainage from the soil column was collected via a hose running from the drainage outlet into sealed buckets beneath the lysimeter. The volume of this leachate was measured at least 3 times per week. After rainfall and effluent application additional measurements of drainage were taken. During these initial stages of the project, some drainage could not be recorded due to insufficient capacity of the collection buckets following effluent application and/or rainfall events. The collection capacity of the drainage container was then enlarged leading into the first autumn of the experiment. However, some drainage events exceeded this capacity in the winter. In October 1996, tipping-bucket flow meters were installed for the measurement of leachate flow from the drainage outlet. Drainage flow was then recorded hourly by a CR-10 data logger. Storage of leachate in the buckets continued so as to enable sample collection for chemical analysis.

3.2.6 Evaporation and Evapotranspiration

Because of the critical nature of drainage to this study, a lot of effort went into its measurement. On the other hand, evapotranspiration was a process that is reasonably understood so that it could be inferred as the residual of the water balance.

Evapotranspiration (ET) is the sum of evaporation losses from the soil plus transpiration losses. ET (mm) was calculated from the water balance equation.

$$ET = I + R_e - D - \Delta W \qquad [3.10]$$

Where I (mm) is effluent irrigation, R_e (mm) is effective rainfall, D (mm) is the drainage leachate and ΔW (mm) is the change in soil water storage. Surface run off was not included in this equation as the lysimeter edging prevented any surface flow of water. Unfortunately the drainage record was incomplete for some periods of the experiment. During these periods there are 2 unknowns in the water balance equation, ET and drainage. However, ET estimates were obtained for the periods in which all other measurements were made. These ET values enabled the prediction of the drainage volumes for periods in which they were not measured.

The record of leachate from bare-soil lysimeters was incomplete for most of the first season, due to overflow of the leachate collection buckets, so Eq. [3.10] was unable to be used for the calculation of evaporation. Bare-soil evaporation follows a consistent pattern. Bare-soil evaporation, E (mm), has been shown to follow the short-term relationship

$$E = c\ t^{-0.5} \qquad [3.11]$$

for Manawatu fine sandy loam in summer. Kerr (1974) found c = 2.75 mm day^{-1}. Here, t is the number of days since a rainfall, or irrigation event exceeding 3 mm. Either the rate is controlled by the soil (E) or by the atmosphere's ability to evaporate the water,

namely the maximum soil evaporation rate (E_s). If E exceeds E_s for any given day, then E is set to equal E_s. In this experiment an E_s of 2.5 mm day^{-1} provided a better fit with the measured values of bare-soil evaporation.

Bare-soil evaporation is variable throughout the seasons, in this experiment this is simulated by replacing c from Eq. [3.11] with the potential evaporation rate (E_p) as determined via the Penman equation. Equation [3.12] was used to calculate the bare-soil evaporation for periods where leachate data was not collected. This model is validated and presented in the results, in Section 3.3.3.

$$E = E_p \; t^{-0.5} \qquad [3.12]$$

Incomplete data sets for drainage also occurred for the tree treatments. These were generally less than one or two consecutive days of data. For periods when drainage was known for periods either side, the ET could be calculated. These known ET values were used to calculate the crop coefficient (k_c) (the factor relating tree water use to E_p). Separate crop co-efficients were calculated for each tree as required. The crop coefficients were then used to calculate ET for the periods when drainage was unknown. When drainage data were unavailable, these calculated ET values were used to infer total drainage for that week via Eq. [3.10]. This total weekly-drainage was used to allocate the drainage for those periods of time within each week where data were missing. These data were required for subsequent calculation of the nitrogen quantities leached. Because of this requirement the time of sample collection for nitrogen determination defined the periods for calculation of drainage volume. In cases where

more than one period required determination, the drainage volume was apportioned to the separate events based on the average flow rate.

3.2.7 Data collection frequency

TDR measurements and drainage volumes were always collected on the day of effluent application including before, during and after application. The measurements were repeated the following day and then 2 days later. This was the minimum sampling period for data collection. Following installation of the tipping bucket flow meters, the drainage flow rates were logged hourly. Although the effluent irrigation system was designed to deliver a consistent amount of effluent application, the volume of effluent application was monitored bi-monthly during the irrigation periods.

All water balances calculations were done on a weekly basis. The TDR measurements before irrigation served as the first reference point. The closing value was taken at the end of the week just prior to the next irrigation. Data will be presented as monthly averages and seasonal totals for clarity of presentation and ease of interpretation. The monthly time periods are defined in Table 3.1. Results are presented as mean daily averages provide comparable values between months with a different number of days. Minitab 12.1 (Minitab Inc. 1998) was used for all statistical analysis. ANOVA tests were used to compare the treatments.

3.3 Results and Discussion

3.3.1 Effluent irrigation

During the application period from 5 December 1995 to 16 June 1996 (period 1, 29 weeks), the lysimeters received a total of 618 mm of irrigation. During the longer application period from 16 September 1996 to April 14 1997 (period 2, 31 weeks), the lysimeters received a total of 670 mm of irrigation. The average weekly effluent hydraulic loading rate during the application periods was 21.5 mm $week^{-1}$, which includes the bore water used to flush the pipes following each application. There were no significant differences in the amount of effluent received between the treatments (P=0.05). Effluent application periods and rates applied are shown for each month in Figure 3.4a, and for each season in Table 3.1.

3.3.2 Rainfall

The cumulative rainfall during the 620 day experiment was 1830 mm. The monthly distribution is shown in Figure 3.4a and the seasonal summary is shown in Table 3.1. Other than the regular effluent applications, the only other irrigation was some 35 mm, that was applied to all lysimeters on August 6 1997 to simulate a heavy rainfall event just prior to the end of the experiment. In this analysis, the irrigation event is included in the rainfall for August 1997.

Table 3.1 Seasonal water balance (mm) of the 4 treatments. Except for tree water use, the conversion from volume (L) to depth equivalent (mm) is simply based on the area of the lysimeter. Tree water use is converted from volume (L) to equivalent depth (mm) based on a stocking density of 4000 stems ha^{-1}. Numbers in italics contain some calculated values (see text for details).

	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Total
Starting	1/1/96	6/3/96	5/6/96	27/8/96	26/11/96	25/2/97	27/5/97	
Finishing	5/3/96	4/6/96	26/8/96	25/11/96	24/2/97	26/5/97	11/8/97	
No. Days	63	91	83	91	91	91	80	590
Bare-soil								
R_e[1]	155.7	317.7	263.8	271.5	207.7	208.6	204.3	1629.3
I[1]	181.1	261.6	60.8	197.1	260.2	158.5	0.0	1119.3
ΔW[1]	17.5	24.1	0.3	0.3	-3.5	-5.5	24.6	57.8
D[1]	*184.7*	*437.6*	*282.7*	*320.2*	262.9	*269.7*	*136.6*	*1894.3*
ET[1]	*134.6*	*117.6*	*41.6*	*148.1*	208.5	*102.9*	*43.1*	*796.4*
E. saligna								
R_e[1]	144.4	294.9	237.4	240.2	176.5	187.3	204.4	1485.1
I[1]	166.9	241.1	56.0	181.8	247.4	150.5	0.0	1043.6
ΔW[1]	7.8	36.4	-0.5	-6	-161.7	55.2	99.2	30.3
D[1]	*29.1*	*180.2*	*130.8*	*64.8*	6.7	2.3	*49.2*	*463.1*
ET[1]	*274.4*	*319.4*	*163.1*	*363.2*	578.9	280.3	*56.0*	*2035.3*
E. nitens								
R_e[1]	144.4	294.9	237.4	240.2	176.5	187.2	202.3	1482.9
I[1]	185.2	267.5	62.2	202.3	262.1	158.8	0.0	1138.1
ΔW[1]	-138.5	126.9	10.2	-131.5	-37.5	77.0	121.7	28.3
D[1]	0.3	28.2	77.4	0.6	0.0	0.0	*48.8*	*155.3*
ET[1]	467.8	407.3	212.0	573.4	476.1	269.0	*31.8*	*2437.4*
S. kinuyanagi								
R_e[1]	144.4	294.9	255.9	253.5	178.9	190.6	203.8	1522.0
I[1]	171.0	246.9	60.3	208.1	248.0	152.7	0.0	1087.0
ΔW[1]	-119.0	141.6	50.8	-180.5	44.0	72.8	75.0	84.7
D[1]	0.8	10.3	*204.1*	*122.0*	0.0	65.8	*88.9*	*491.9*
ET[1]	433.6	389.9	*61.3*	*520.1*	382.9	204.7	*39.9*	*2032.4*

[1] R_e= rainfall, I = irrigation, ΔW= change in soil water storage, D = drainage and ET=evapotranspiration.

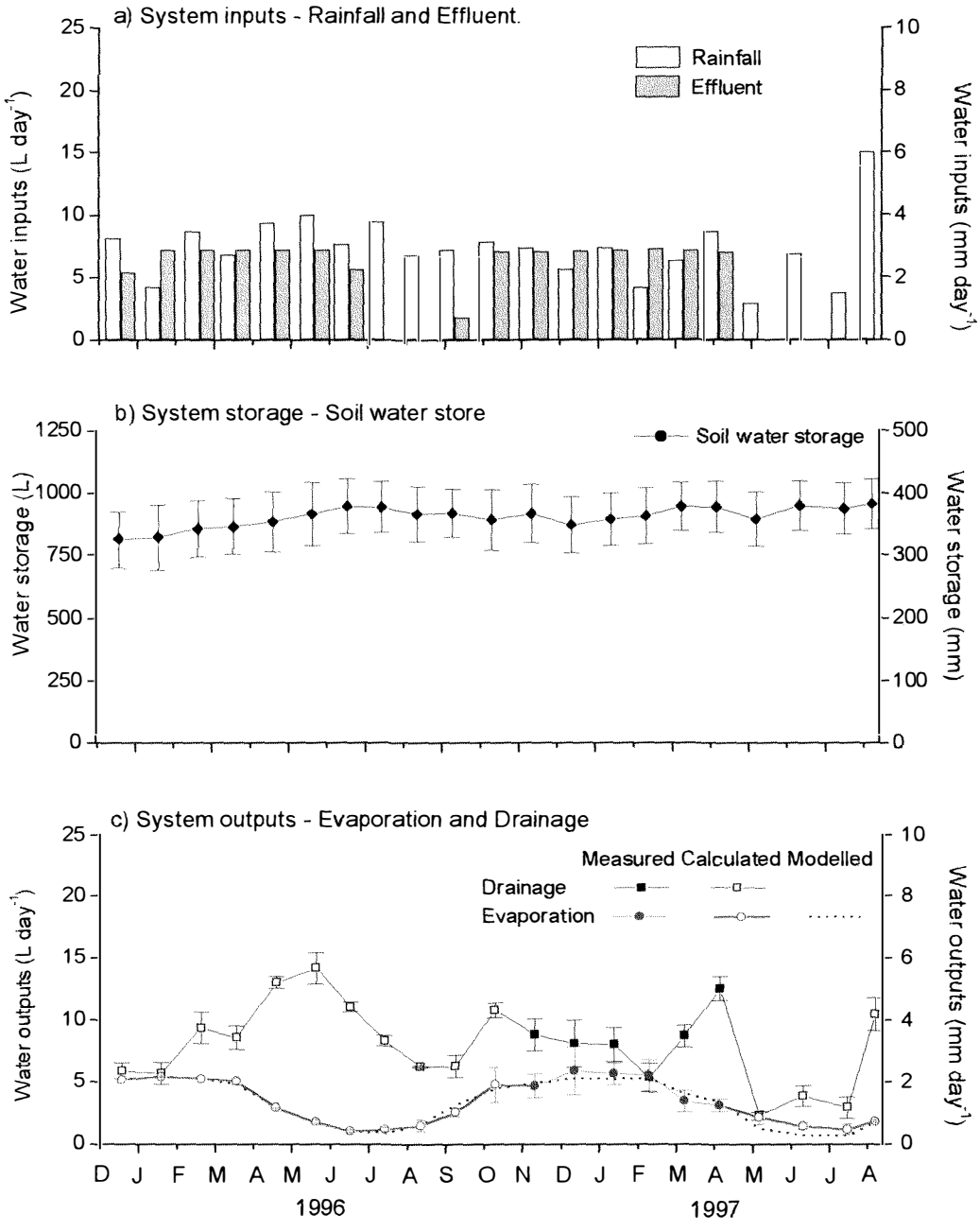

Figure 3.4 Bare-soil monthly water balance a) System inputs - total rainfall and effluent applied. b) System storage - mean values of soil water storage of the 3 bare-soil lysimeters and the standard deviation. c) System outputs - mean values of drainage and evaporation of the 3 bare-soil lysimeters and the standard deviation. The dashed line represents the modelled data for bare soil evaporation. See text for details.

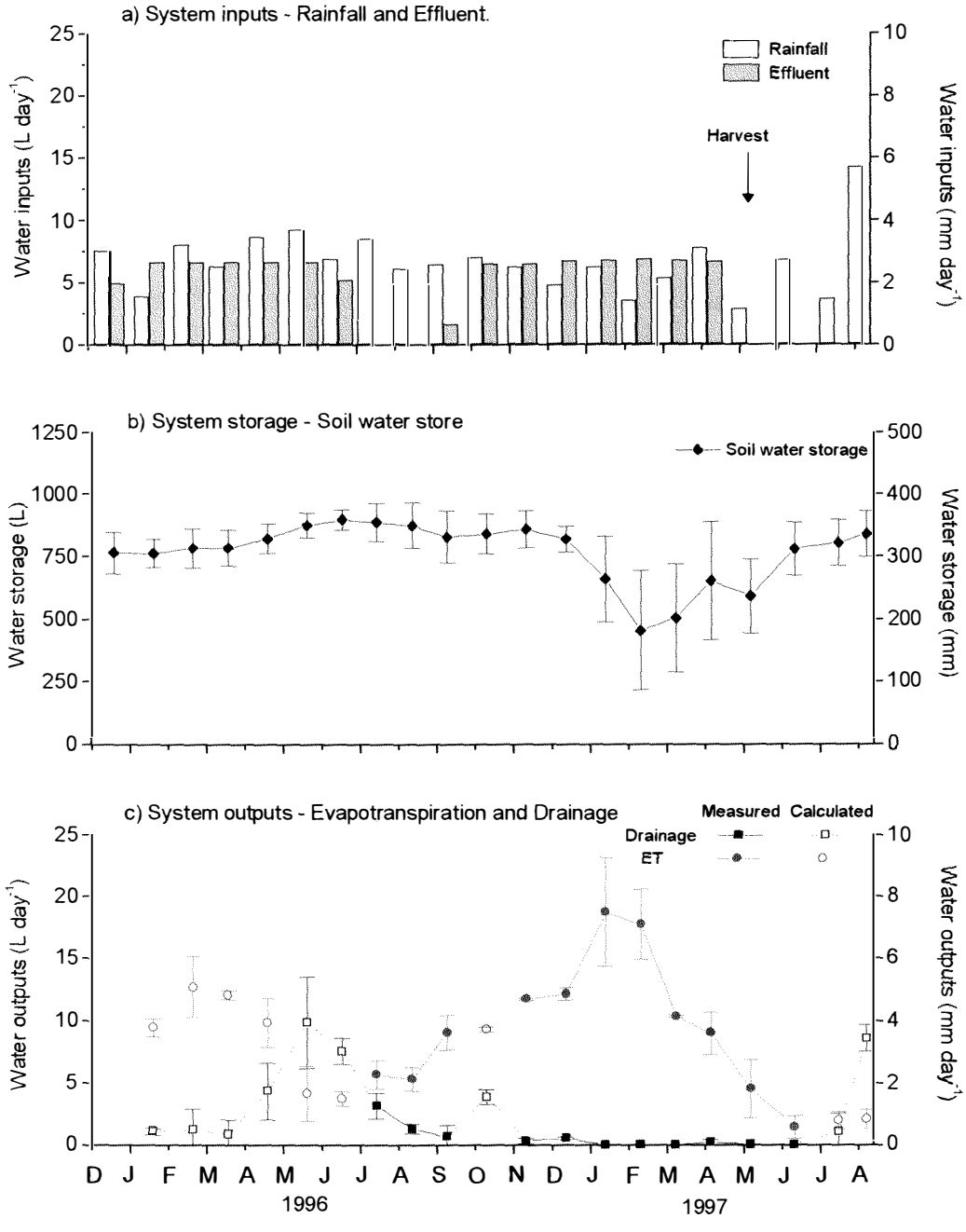

Figure 3.5 *E. saligna* (evergreen) monthly water balance a) System inputs - total rainfall and effluent applied. b) System storage - mean values of soil water storage of the 2 *E. saligna* lysimeters and the standard deviation. c) System outputs - mean values of drainage and evapotranspiration of the two *E. saligna* lysimeters and the standard deviation. See text for details.

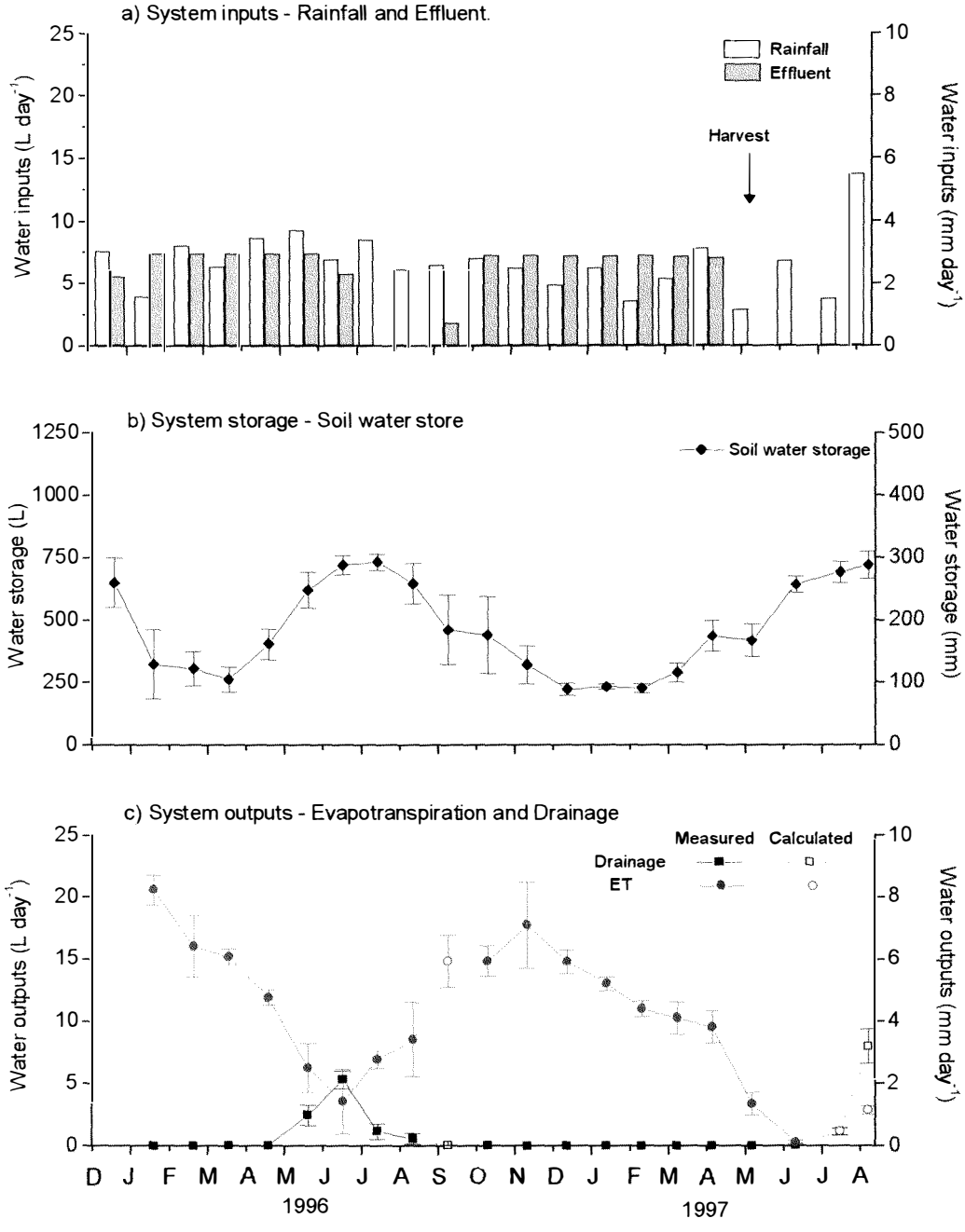

Figure 3.6 *Eucalyptus nitens* (evergreen) monthly water balance a) System inputs - total rainfall and effluent applied. b) System storage- values of soil water storage of the three *E. nitens* lysimeters and the standard deviation. c) System outputs - mean values of drainage and evapotranspiration of the 3 *E. nitens* lysimeters and the standard deviation. See text for details.

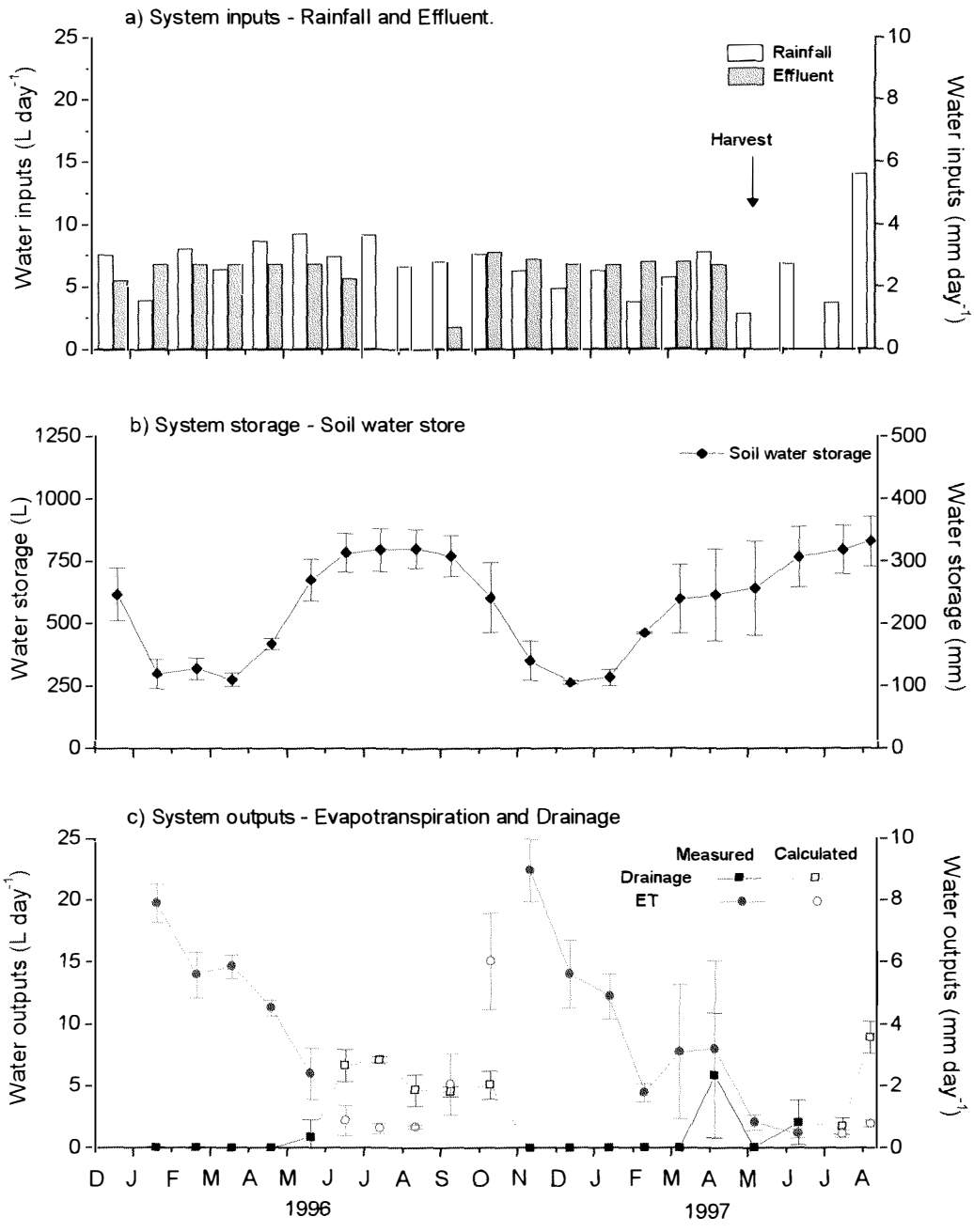

Figure 3.7 *S. kinuyanagi* (deciduous) monthly water balance a) System inputs - total rainfall and effluent applied. b) System storage - mean values of soil water storage of the three *S. kinuyanagi* lysimeters and the standard deviation. c) System outputs - mean values of drainage and evapotranspiration of the three *S. kinuyanagi* lysimeters and the standard deviation. See text for details.

3.3.3 Water balance

The water balance of the bare-soil lysimeters is shown in Figure 3.4 and Table 3.1. Weekly bare-soil evaporation was calculated using Eq. [3.10], when drainage volumes were known. Periods with a complete data set are represented in Figure 3.4c as filled circles. The bare-soil evaporation values calculated via the model (Eq. [3.12]) are shown in Figure 3.4c (dashed line) to be good agreement with the measured values (Eq. [3.10]) (solid line). So there is confidence in using the model evaporation data to calculate the missing drainage values.

For those weeks with unknown drainage, the model Eq. [3.12] was used to estimate weekly evaporation. These modelled values were combined with the measured values to provide a complete record of evaporation on a monthly basis. These combined data are termed calculated evaporation, as shown with a solid line and open circles in Figure 3.4c. Where drainage data were unavailable, the calculated evaporation values were used to estimate total drainage for each week using Eq. [3.10]. This total weekly drainage was used to determine the drainage for the periods of time within each week where data were missing. Calculated drainage values are shown as open squares in Figure 3.4c.

The bare-soil lysimeters provide base information on the role of the soil in land treatment system. The bare-soil water storage was consistent throughout the experiment ranging between 325 mm lysimeter^{-1} and a 'field capacity' of around 380 mm lysimeter^{-1} during the experiment. This is expected because the overall bare-soil evaporation is small, due to the rapid drop off in evaporation following wetting (see Eq. [3.12]). The water storage is lower in summer, as greater evaporation removed more soil

water from the soil surface. However, evaporative demand throughout the experiment was generally lower than the water inputs, hence the consistently high values of water storage. Because of the imbalance between water inputs and evaporation, drainage was continuous throughout the experiment and ranged from 0.9 mm day^{-1} to 5.2 mm day^{-1} and averaging 3.2 mm day^{-1}. Because of the conservative nature of evaporation, rainfall and effluent application had the greatest influence on the amount of drainage. For bare-soil, the amount of drainage effectively was the sum of water inputs minus evaporation with only minor adjustments for water-storage changes.

The trees added another level of complexity to the system through their greater and seasonally variable transpiration. The water use of the trees had a large impact on the systems water balance. As expected evapotranspiration of the trees was greater than evaporative demand by bare-soil. The highest average monthly evaporation for bare-soil was 5.5 L day^{-1} (or 2.2 mm day^{-1}). In the lysimeters, the *E. saligna*, *E. nitens*, and *S. kimuyanagi* trees recorded highest monthly ET rates of 18.8, 20.5 and 22.5 L day^{-1} (or 7.5, 8.2 and 9.0 mm day^{-1}) respectively.

E. saligna trees with smaller leaf areas (Figure 2.4), had ET rates lower than the water inputs for the first 11 months of the experiment (Figure 3.5, Table 3.1). During this period, *E. saligna* and bare-soil treatments therefore had similar patterns of ET, drainage and water storage. Although transpiration maintained the water storage and the drainage from the *E. saligna* soil profiles lower than the bare-soil treatments. However, as *E. saligna* trees grew, transpiration subsequently increased. In the second summer, the tree water use increased sufficiently, due to a larger leaf area, to be greater than the water inputs. Increased water use resulted in the water storage declining and drainage

completely stopping. In autumn, *E. saligna* water use lowered with the change in potential evaporation. In autumn, drainage of *E. saligna* treatments remained low as the balance of water inputs over ET was met by soil-water-storage recharge. Following the harvest in April-May 1997 water use was limited to evaporation, consequently the drainage increased.

The high ET rates (Table 3.1) of *E. nitens* (Figure 3.6) and *S. kimuyanagi* (Figure 3.7) trees in the summer periods were possibly due to their leaf areas (Figure 2.4). This resulted in the water storage decline and negligible drainage. In March 1996 water storage went as low as 104 and 110 mm for *E. nitens* and *S. kimuyanagi* respectively. The water balances of these species in winter were quite different. The deciduous *S. kimuyanagi* shed it leaves while the evergreen *E. nitens* trees maintained a leaf cover. During the winter, the *S. kimuyanagi* treatments had the greater drainage of 204 mm compared to the *E. nitens* treatments drainage of 74 mm. Without leaves the *S. kimuyanagi* ET was much lower than *E. nitens* treatments, thus a larger proportion of water inputs drained. The throughfall of rain would have been greater for the *S. kimuyanagi* without the canopy of leaves, which Eucalypts had maintained. Increased throughfall added to the hydraulic loading and drainage volume from the root zone of the *S. kimuyanagi* treatments.

By early summer, December-96, the water storage of the *E. nitens* and *S. kimuyanagi* trees again declined rapidly to be at 88 mm and 110 mm respectively. Tree water use at this time was limited by the inputs of water. The *S. kimuyanagi* treatments were observed to drop a significant number of leaves in January 1997, this was attributed to water stress. From this period once again the *E. nitens* and *S. kimuyanagi* water balances

differed. *S. kimuyanagi* ET declined due to its smaller leaf area. The water inputs were then greater than the ET, resulting in an increase in water storage in the soil. In contrast *E. nitens*, which had maintained leaf area during this period, continued to use all of the water inputs to meet ET demands. Thus *E. nitens* maintained water storage at a low level throughout the summer. In autumn, potential ET lowered and the water inputs stayed high. *S. kimuyanagi* began to drain in April, following recharge of the soil profile (Figure 3.7). The *E. nitens* however did not drain during autumn as the surplus of water inputs over ET went into recharge of the soil profile. The first drainage from the *E. nitens* treatments in 1997 occurred in July (Figure 3.6c).

3.3.4 Tree water use

Tree water use of the *E. nitens* and *S. kimuyanagi* was likely limited by water availability from January to April of 1996 and November 1996 to February 1996. At peak however, the *E. saligna*, *E. nitens*, *S. kimuyanagi* trees recorded highest monthly ET rates of 18.8, 20.5 and 22.5 L day^{-1} (or 7.5, 8.2 and 9.0 mm day^{-1}) respectively. These values are slightly higher than the range found by other Australian and New Zealand studies (Dunin and Aston, 1984; Sharma, 1984; Myers et al., 1996; Tungcul et al., 1996). In this experiment, summer time effluent-irrigation was applied to small plots in an otherwise dry landscape, thereby there may have been an 'oasis effect' as was found by Myers and Talsma (1992) and Myers et al. (1996). It is also likely the higher water use rates at peak in comparison to other studies, are a consequence of the area used to calculate tree water use. To calculate water use in mm, the number of liters a tree used was divided by a ground area of 2.5 m^2. This was based on a stocking rate of 4000 stems ha^{-1} and also coincided with the 2.5 m^2 surface area of the lysimeter. Annual ET rates from the *E. nitens* and *S. kimuyanagi* trees are probably lower than would have been attained had water been in adequate supply.

3.3.5 Water stress

The *E. nitens* and *S. kimuyanagi* trees were likely water stressed in periods where soil water storage was less than 125 mm in the lysimeter. Water stress in trees can develop in the short term because the rate of water uptake from soil is slower than the rate of loss by transpiration. In the longer term, over days and weeks, cumulative losses by transpiration can reduce the availability of soil water for trees (Slatyer, 1967). Permanent wilting point occurs when plants can no longer recover overnight from wilting during the day (Sasse and Sands, 1996). So severe was water stress in this experiment, that after December-96 rapid decline in leaf area is observed for the *S. kimuyanagi* (Figure 2.4). This premature leaf abscission indicates that the trees had dried the soil close to permanent wilting point. Indeed the rate of irrigation during this period could (and should) have been increased.

Initially, the willow leaf area increased rapidly in the spring of 1996 and soil water storage decline was subsequently fast. A shedding of leaves by *S. kimuyanagi* trees was first observed in January 1997 and leaf area measurements showed a significant drop between December 1996 and January 1997. The leaf area of *E. nitens* also trended downward from December 1996 to April 1997 but not dramatically and there was no significant difference in leaf area for either species during this period (P=0.05). The decline of soil water storage began earlier but was slower for *E. nitens* than *S. kimuyanagi.* The *S. kimuyanagi* and *E. nitens* have shown differing responses to the water stress. The implication of the early leaf abscission of the willows was the advanced recharge of soil water storage and onset of drainage earlier than the *E. nitens* treatments.

High rates of water use played a part in species selection for this experiment. However, it was not expected that water levels would reach the stress points. Dairy-shed effluent treatment using SRF trees should seek to exploit, rather than hinder tree water use. Limitation to root expansion, because of the lysimeter size may also have contributed to the observed water stress. There are several factors that may have contributed to the different response of *E. nitens* to water stress in comparison to *S. kimuyanagi*. Eucalypts are known to vary leaf area within a season in relation to water stress (Sharma, 1984). Eucalyptus species also have a strong stomatal response to increasing vapor pressure deficit, which can have the effect of lowering water use rates in dry conditions (Leuning, 1990; Sun and Dickinson, 1993; Myers et al., 1998). Following water stress conditions and rewatering, *E. nitens* stomatal conductance has been shown to stay low for up to 3 weeks (White et al., 1999). The effect of the water stress for SRF land treatment management is discussed further in section 3.3.7.

3.3.6 Drainage

The total drainage volume for the experiment, including after the harvest, showed some variation between the treatments. Bare-soil drainage of 1894 mm was much greater than the tree treatments over the course of the experiment. In comparison, the deciduous *S. kimuyanagi* drained 492 mm, whereas the evergreen *E. saligna* and *E. nitens* treatments drained 463 and 155 mm during the experiment. The drainage of the *S. kimuyanagi* treatments was greater than *E. saligna*, this difference reflects a difference in the water inputs in winter and spring of 1996 (Table 3.1). The cumulative values of ET of the *E. saligna* and *S. kimuyanagi* were similar (Table 3.1).

Differing water use of the treatments greatly affected the volume and timing of drainage events, as discussed above. The quantity of leachate going to groundwater does have

some consequences for aquifer management, however it is the quality that is of most concern. The nitrate loading to the aquifers is discussed in Chapter 4.

3.3.7 Management strategy

Drainage of the soil profile was greatest between May and June 1996 for all treatments. This leaching and the water shortage of the trees could potentially have been minimised by careful management of the irrigation events. At the onset of this experiment it was decided to irrigate once per week at 21 mm $week^{-1}$. In hindsight, tailored irrigation of larger amounts of effluent to the trees in the dry seasons to match crop demand for water would have been preferable. More frequent irrigation in the summer is another consideration to ensure the trees always have water to transpire, so as to maximise uptake by trees. These options of greater irrigation earlier in the season could also reduce the requirement for hydraulic loading in the wetter seasons, thereby reducing the leaching.

In hindsight, alternate management strategies may have resulted in better management of the water regime. Some of these will be addressed with the computer model in Chapter 5. The model provides a quicker and less costly assessment of these proposed strategies than the use of experiments.

3.4 Conclusions

The bare-soil treatments had high amounts of drainage throughout the course of the experiment. Rainfall and effluent inputs had a considerable influence on the amount of leachate from the soil zone. The modified model of Kerr (1974) predicted bare-soil evaporation, and was in good agreement with field measurements.

The addition of trees to the system introduced the component of transpiration, which greatly effected the water balance. *E. saligna*, initially with small leaf area, behaved at first similar to the bare-soil treatment. Then ET demand in the tree was lower than water inputs, resulting in drainage for a considerable part of the experiment, just like the bare soil. Once the leaf area increased the ET demand was greater than water inputs and drainage ceased.

Both *E. nitens* and *S. kimuyanagi* had high ET demand during the first irrigation season which lowered the water storage close to stress point. In the second irrigation season both species again used water at a rate which lowered the soil water content to the stress point. The *E. nitens* showed no significant drop in leaf area however the *S. kimuyanagi* showed significant loss in leaf area. This dramatic physiological change in willow was attributed to water stress. Water use of the trees is likely to have been greater had there been a greater supply of water at the times of water stress.

Maximum water use estimates for the trees are slightly higher than those reported elsewhere. This is attributed to increased transpiration through an 'oasis effect' and difficulties in ascertaining the ground area to which the tree water use relates.

Water use of these trees differed over the winter. The deciduous willows shed leaves in winter decreasing both ET and rainfall interception, resulting in greater winter drainage from the deciduous trees than the evergreen trees.

The leaching period of the *E. nitens* was shorter than that of the *S. kirnuyanagi* and rates of leaching were lower. Both these treatments leached for shorter periods than *E. saligna*. Leaching of the bare-soil treatment was consistently high throughout the experiment.

This chapter has highlighted the differing water balances of the treatments. Water use was found to have a large impact on drainage volume and timing. In Chapter 4, the implications of differing water use on the nitrogen balances of the system will be investigated. As a part of this the leachate nitrogen concentrations and loading moving to the aquifer is assessed. The water balance study has identified the importance of irrigation management on water balances. Chapter 5 sets out to use a model to question the impacts of such management practices on water and nitrogen budgets in land treatment systems.

Chapter 4 Nitrogen Balance

4.1 Introduction

The experimental focus of this project is to obtain measurements of water and nitrogen balances in an SRF dairy-shed effluent-treatment system. These will be used to develop an understanding of the key processes so that modelling tools can be used for the prediction of the fate of applied water and nitrogen in SRF land treatment systems. This second step of modelling will enable us to estimate the effects of different loading rates on nitrate leaching, thereby aiding the design and management of sustainable land treatment systems. In the previous chapter the measured water balance of the system has been discussed. The aim of this complementary chapter is to obtain the basic nitrogen balance information, which is essential to develop predictive models (Chapter 5).

To resolve the nitrogen balance of a rootzone, measurements need to include both the inputs and outputs of nitrogen, plus measures of the changes in the soil store of nitrogen. Here, the inputs are from effluent application and mineralisation. The outputs are plant uptake and leaching. Plant uptake is therefore linked intimately to any leaching to groundwater. What is not taken up by the plant, or lost in the soil through immobilisation, denitrification and volatilization is available to be drawn downwards by any percolating water.

Biomass accumulation of nitrogen is an integral part of a nitrogen budget of the system. Harvesting and removal of above-ground biomass takes nitrogen away from the site potentially enhancing the sustainability of the high nitrogen inputs into the system. Thus for this land-treatment research, the plant uptake must be assessed as both the above and below ground accumulation of nitrogen. The nitrogen distribution in biomass is known

to differ between roots and shoots, and within the various size classes of these biomass components (Young and Carpenter, 1975; Ericsson, 1995). Thus the nitrogen content in each of the size classes needs to be assessed.

Previous studies of dairy-shed effluent irrigation to SRF species have identified the need for further investigation of nitrate leaching (Tungcul et al., 1996; Lu, 1997). Tungcul et al. (1996) irrigated dairy-farm effluent onto *E. nitens*, and 9 species of *Salix* at Aokautere, New Zealand. They predicted nitrate leaching was occurring, however were unable to quantify this. Lu (1997) identified that as application rates of dairy-shed effluent to willows increased (from 6 to 30 mm $week^{-1}$) nutrient leaching also increased. Other studies irrigating with municipal effluent to trees have also identified the potential for nitrate leaching. For example, in the Rotorua Land Treatment System (RLTS), an effluent irrigated pine forest, high nitrate fluxes in seepage waters have been observed (Tomer et al., 1997). Experiments at Wagga Wagga, Australia, have revealed elevated nitrate leaching under both pines and Eucalypts that were irrigated with municipal effluent (Myers et al., 1997). Nitrate leaching is thus a major consideration for the sustainability of land treatment systems. Prior to advocating SRF for treatment of dairy-shed effluent, an investigation of nitrate leaching would be merited.

The goal of this current work is to understand better the key processes that determine the fate of nitrogen applied as dairy-farm effluent to SRF species. To achieve this, a lysimeter experiment was set up to quantify nitrogen balances of 3 SRF species. A bare soil control received the same amount of effluent application so that the efficiency of the SRF system could be assessed.

In this experiment, determination of the nitrogen budget for each treatment was sought through measurement of

1) The inputs of nitrogen from the effluent applied
2) The changes in soil nitrogen storage
3) The nitrogen accumulation in the biomass of the trees
4) The amount of nitrogen leaching from the root zone of the 4 treatments.

4.2 Methods

4.2.1 Experiment design

The overall experimental design is discussed in Section 2.2.1. Here the methodologies utilised in relation to aspects of nitrogen balance are more fully discussed. The lysimeter facility (Figure 4.1) enabled measurement of all inputs and most of the outputs of nitrogen from the rootzone. The nitrogen inputs are from effluent irrigation. The measured outputs of nitrogen were plant uptake and nitrogen leaching. The soil nitrogen storage provides a buffer from which these inputs and outputs take place. Volatilization and denitrification are other possible output pathways from the system. Volatilization and denitrification were not measured in this experiment. The implications of this omission will be discussed further in Sections 4.4.5 and 4.4.6.

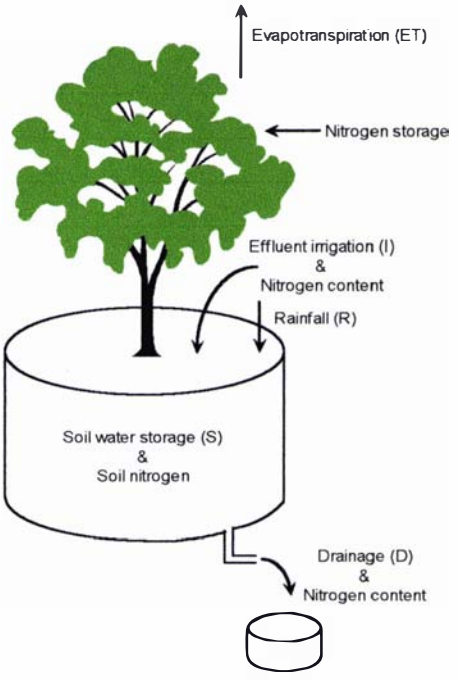

Figure 4.1 Inputs and outputs of water and nitrogen measured in the lysimeter treatments.

4.2.2 Effluent source and application

The effluent used in this project was sourced from Massey University's Number 4 Dairy Farm. The effluent was trucked from the secondary pond on the farm to the experimental site on each day of effluent application. Effluent samples were collected prior to each effluent application and they were analysed for mineral nitrogen content. A filter in the irrigation system was used to remove the sludge, which contains the organic nitrogen, from the effluent. Effluent application methods are discussed fully in Section 3.2.2.

4.2.3 Soil sampling techniques

To establish the initial soil-nitrogen concentrations soil was sampled on 5 December 1995 prior to effluent application. Five samples of approximately 20 grams each were collected from the soil surface (0-0.1 m) of each lysimeter. These were then analysed for total nitrogen and KCl extractable nitrogen. The analytical methods are discussed in detail below (Section 4.2.6).

To determine the soil nitrogen concentrations at the end of the experiment, soil samples were collected in August 1997. Two soil cores, with a diameter of 46 mm, were taken to a depth of 0.6-0.7 m from each lysimeter. The maximum depth reflects the depth at which stones prevented the corer from going any deeper. The two cores were taken at 0.2 and 0.4 m across the 0.9 m radius of the lysimeter edge. Cores were attempted at 0.6 m across the diameter however the predominance of large roots prevented the extraction. An engine-powered concrete breaker was used to drive the coring tubes into the soil and a tripod and winch system was used to extract them (Welbank and Williams, 1968). Each core was cut into 0.1 m lengths. The 0.1 m cylinders of soil were then sieved to 2 mm to remove any stones and roots. The soil samples were all analysed

for KCl extractable mineral nitrogen. Total Kjeldahl nitrogen contents were determined for surface samples of each core and for all depths for one core of one replicate.

4.2.4 Biomass nitrogen content

The nitrogen content of all plant samples was analysed using the total Kjeldahl method. The details of this method are described in Section 4.2.6. The sub-sampling techniques used to obtain the representative biomass samples for nitrogen analysis are described in the following sections.

4.2.4.1 Plant nitrogen content - above ground

The accumulation of nitrogen in the above-ground biomass during the experiment was measured at the time of harvest, which was between 24 April-8 May 1997. Table 2.1 shows the dates of harvest for the individual trees. The above-ground biomass of the trees was separated into 4 groups following the method of Young and Carpenter (1975). The woody-stem biomass was separated into 3 diameter groups that include; < 6.4 mm, 6.4-25 mm, and > 25 mm. Leaves were the fourth category of above-ground biomass. The mass produced in these categories is presented and discussed in Chapter 2. Nitrogen content of all these categories was analysed separately.

4.2.4.2 Plant nitrogen content - below ground

Root mass was measured at the end of the experiment in August-September 1997. Root mass was estimated by both root coring and total excavation. The methods and results of root mass estimation are presented and discussed fully in Chapter 2. Nitrogen concentration of the roots was also determined for the size classes as used by Young and Carpenter (1975). Root biomass was sampled in 3 size groups based on diameters being; <6.4 mm, 0.64-2.5 mm, and >25 mm. All root samples were collected from the excavated root systems.

4.2.5 Nitrogen leaching

During the experiment, samples of drainage water were collected at least 3 times per week. Collection of the samples was, however, more intensive during the effluent irrigation seasons and following any large rainfall events. These drainage water samples were analysed for nitrogen content via methods described in section 4.2.6.2 below. The amount of nitrogen leached between these collection times was calculated as the drainage volume between collection times multiplied by the measured concentration of the sample. Drainage volumes were determined by the methods described in Section 3.2.5.

4.2.6 Determination of nitrogen content

4.2.6.1 Organic nitrogen

Nitrogen contents of soil and plant samples were determined using Kjeldahl digestion. Kjeldahl digestion is a general method for total nitrogen measurement in organic material such as soils, plant material and animal manure's (Nelson and Sommers, 1980; Bremner and Mulvaney, 1982; Mahimairaja et al., 1990). Kjeldahl-N includes mostly the organic and ammoniacal-N (Bremner and Mulvaney, 1982; Mahimairaja et al., 1990). During the Kjeldahl digestion nitrogen is recovered in ammoniacal form. This form of nitrogen was measured using an autoanalyser by following Berthelot's indophenol blue reaction method (Markus et al., 1985).

4.2.6.2 Inorganic nitrogen

Inorganic forms of nitrogen in soil can be measured using an extraction with 2*M* KCl solution (Bremner and Keeney, 1966; Bremner and Mulvaney, 1982; Mahimairaja et al., 1990). The inorganic forms of nitrogen in water-based liquid samples, which included effluent and drainage samples, did not require KCl extraction. These could be measured

directly. Inorganic forms of nitrogen in KCl extracts and water-based samples were measured by following the nitroprusside method for NH_4-N (Weatherburn, 1967). The diazotization coupling reaction (Griess-Ilosvay reaction) method was used for NO_2-N and NO_3-N determination (Bremner and Mulvaney, 1982).

4.3 Results

4.3.1 Effluent nitrogen loading

The volumes of effluent applied to each lysimeter varied due to the irrigation system. This resulted in small differences in nitrogen loading to each treatment, but these differences were not significant (P=0.05). Average loading rates are presented in Table 4.1, whereas the loading rates to the separate treatments is shown later in Section 4.3.5. The concentration of nitrogen in effluent ranged from 33 to 149 mg N L^{-1} with a mean concentration of 77 mg N L^{-1} (Figure 4.2). Most of the nitrogen in the effluent (>98 %) was present in ammoniacal form. Of the irrigation periods average nitrogen concentration in the first season (50 mg N L^{-1}) was about half that of the second (102 mg N L^{-1}), while the hydraulic loading was similar for the two seasons (Figure 4.2a). As a result, the effluent nitrogen loading was doubled in the second irrigation season compared to the first season (Table 4.1). The total amount of nitrogen applied, based on the full 620 days of the experiment, equates to a loading rate of 513 kg N ha^{-1} $year^{-1}$.

Table 4.1 Nitrogen loading from effluent application for the lysimeters in the separate seasons and for whole experiment.

	Season 1	Season 2	Total
Number of weeks	29	31	60
Cumulative total (g N $lysimeter^{-1}$)	70	148	218
Cumulative total (kg N ha^{-1})	279	592	871

4.3.2 Soil nitrogen content

4.3.2.1 Organic Nitrogen

All calculations of nitrogen storage and addition use a soil depth of 0.65 m and a bulk density of 1.26 Mg m^{-3}. Changes in total nitrogen content of the lysimeter soil during the course of the experiment are shown in Figure 4.3. Total nitrogen concentrations at the start of the experiment were measured from surface-soil samples only. The initial distribution of nitrogen was assumed to be uniform throughout the profile. The amount of nitrogen applied is small (0.105 mg N g^{-1} soil) relative to the total nitrogen storage in the soil (1.567 mg N g^{-1} soil). Nitrogen in the effluent was in mineral form predominately as NH_4^+-N.

There was no significant difference between the amount of total nitrogen in surface soil sampled, at the beginning and termination of the experiment (P=0.05). This is to be expected as the amount of nitrogen added through effluent irrigation, and the amount of nitrogen lost through leaching and plant uptake are negligible, at least when compared to the amount of total nitrogen already present in the soil.

Depthwise distributions of total nitrogen at the end of the experiment are shown in Figure 4.4. Comparisons of the samples from the top 0.1 m of the soil indicate that changes in total nitrogen were small. Although total soil nitrogen distribution in the beginning was assumed to be uniform, the depthwise-distribution of total nitrogen at the end of the experiment showed total nitrogen to be non-uniformly distributed. It is unlikely these distributions are a result of the inputs, given the small amount of nitrogen applied relative to the total nitrogen storage.

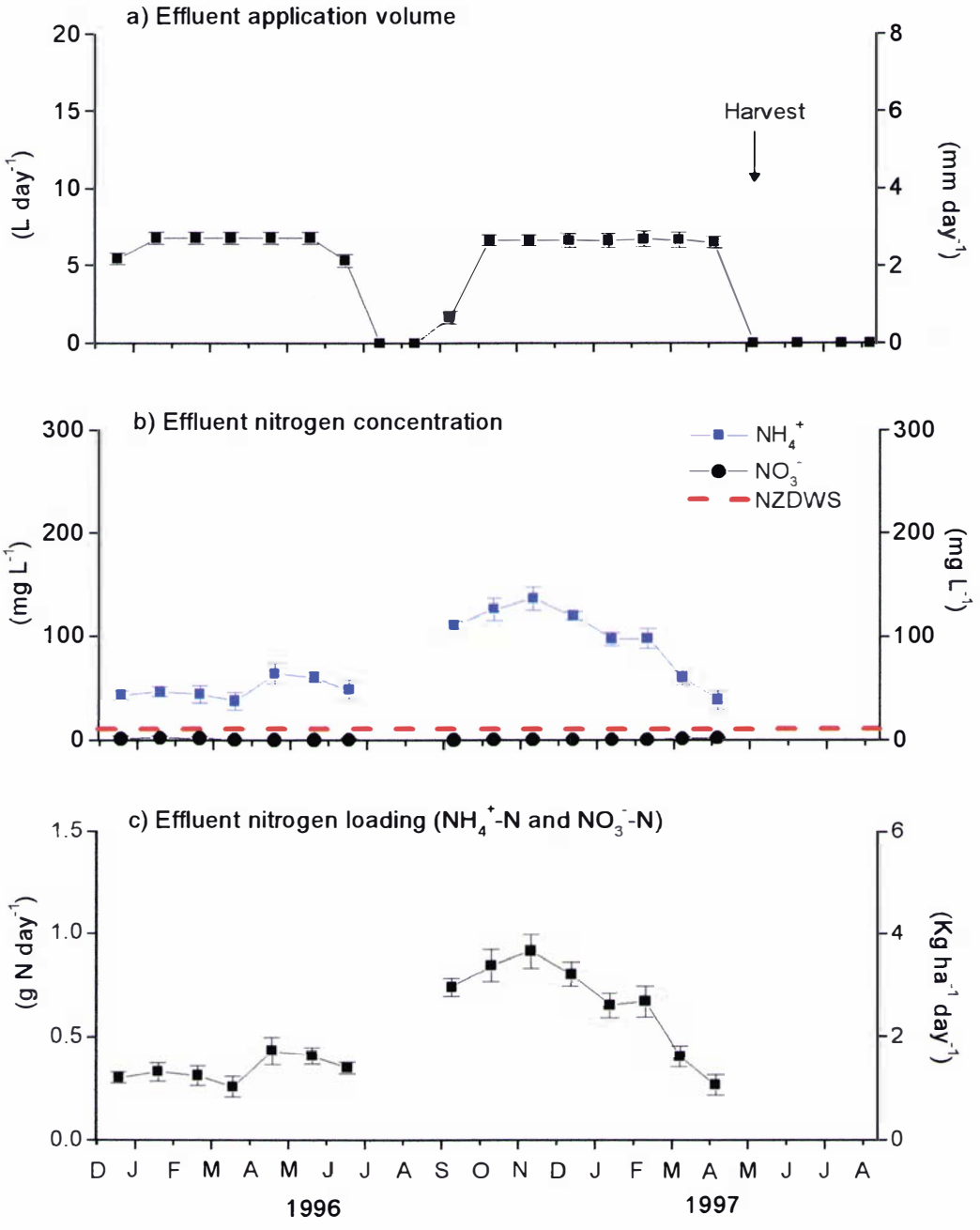

Figure 4.2 Effluent application to the lysimeters a) Volume of effluent applied given as the mean daily average and barely visible standard deviation b) Nitrogen concentration of the effluent presented as mean monthly average and standard deviation also showing the NZ drinking water standard (NZDWS) c) Nitrogen additions from the effluent shown as mean daily averages and standard deviation.

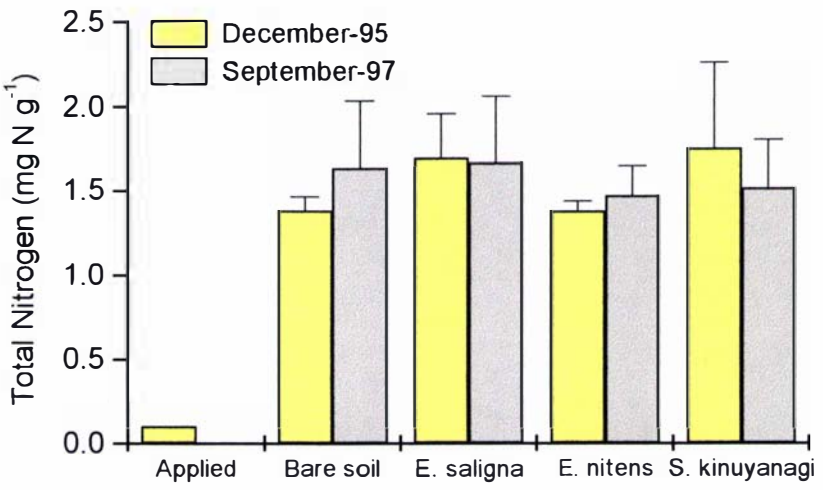

Figure 4.3 Total nitrogen content of the soil before and after the experiment. Applied nitrogen was in mineral form.

4.3.2.2 Inorganic nitrogen

Changes in nitrate and ammonium content of the soil during the course of the experiment are shown in Figure 4.5a and Figure 4.5b. Concentrations at the start of the experiment were measured only from surface samples. The distribution of inorganic nitrogen in the beginning was assumed to be uniform throughout the profile.

Nitrogen in the effluent was predominately in ammoniacal form. The concentrations of ammonium and nitrate in the soil at end of the experiment were not significantly different from the initial concentrations (P=0.05). At both times ammonium concentrations are 5 to 10 times lower than the nitrate concentrations in the soil. The concentration of ammonium added through effluent irrigation was 40-90 times that found in the soil at any time. There was no significant increase in the ammonium concentration in the soil during the course of the experiment. This suggests ammonium was continuously being nitrified to nitrate in the soil. Figure 4.5b shows that this

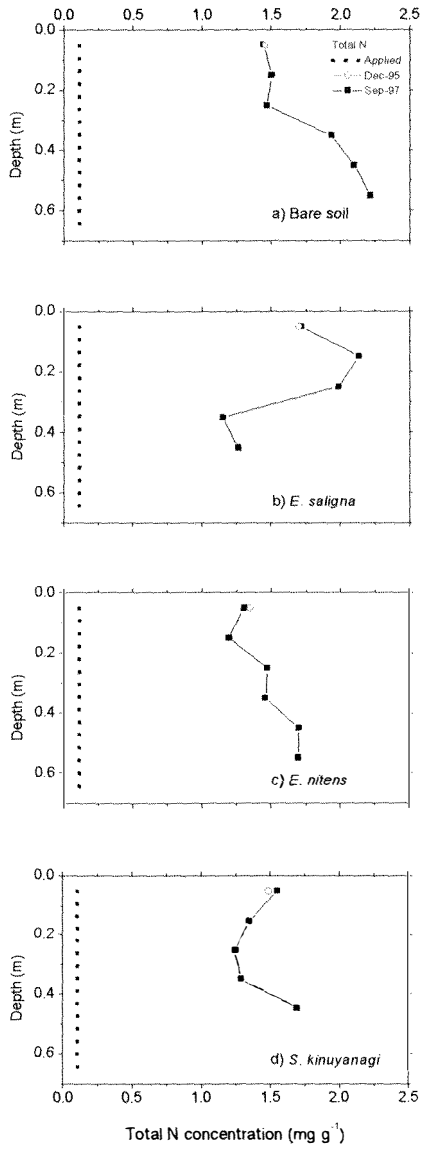

Figure 4.4 The depthwise distribution of total soil nitrogen in the lysimeters at the beginning and end of the experiment. The total amount of effluent nitrogen applied is shown for comparison, as an average value for the whole lysimeter.

transformation was not reflected as an increase in the nitrate concentration of the soil.

The depthwise distribution of nitrate and ammonium at the end of the experiment are shown in Figure 4.6a and Figure 4.6b respectively. The ammonium concentrations in the top 0.1 m of the soil at the start and end of the experiment were very similar for all

treatments. The nitrate contents of the surface soil were generally lower at the end of the experiment for all treatments although this was not significant (P=0.05).

The concentrations of ammonium were evenly distributed with depth for all treatments. Nitrate concentrations were found to be highly variable at all depths across all treatments, however the following generalisations can be made. Nitrate shows higher concentrations with depth for the bare soil and the Eucalypt treatments. The *S. kinuyanagi* treatments seem to have a decreasing nitrate concentration with depth.

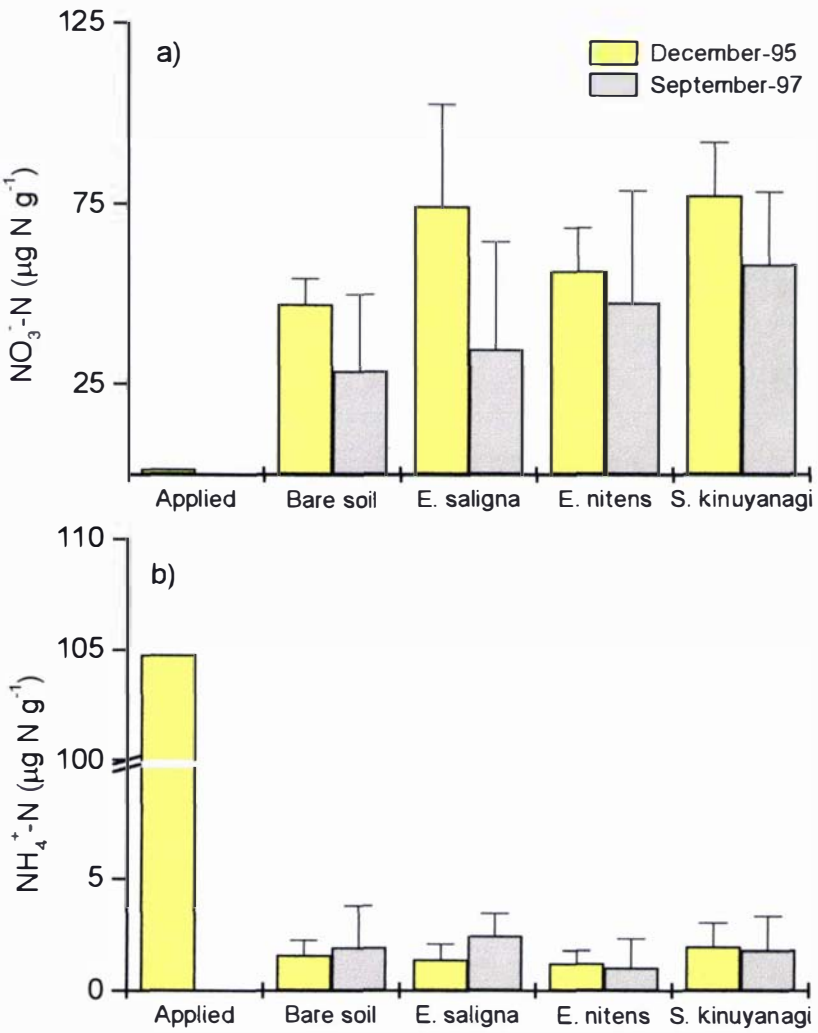

Figure 4.5 Mineral nitrogen stored in the soil before and after the experiment a) NO_3^--N and b) NH_4^+-N. Note the different scales for the two graphs.

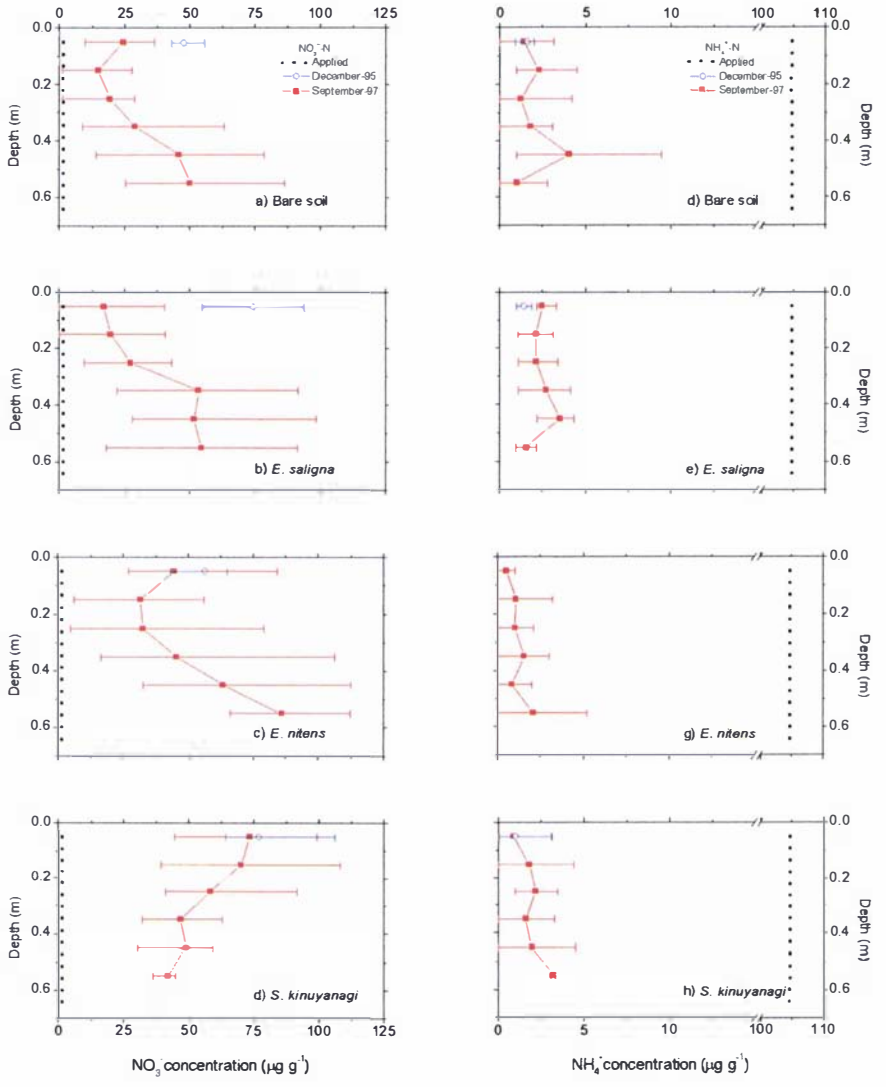

Figure 4.6 Mineral NO_3^--N and NH_4^+-N depthwise distribution at the end of the experiment shown as means and standard deviations. Note the different scales for the two graphs. Application of mineral nitrogen is shown as an average value for the whole lysimeter.

4.3.3 Biomass nitrogen content

The nitrogen concentration of the various biomass components is presented in Table 4.2. Nitrogen concentrations of the biomass differ between the size groups, however some generalisations can be made. Within each tree species, leaves had the highest nitrogen concentration and stem nitrogen concentrations increased as diameters

decreased. Relative to small stems the bark nitrogen concentrations are similar, and the root nitrogen concentrations are higher. Species differences were also notable. Eucalypt stems and bark had a small range of nitrogen concentrations (2.4-6.0 mg N g^{-1}). Whereas, the Salix stems exhibited a larger range with small-stem and bark nitrogen concentrations being 2.5 times greater than those of the Eucalypts.

Here the focus is on nitrogen accumulation in plants (Figure 4.7, Table 4.2). This accumulation is clearly related to the biomass production of the trees, which is presented in Chapter 2. The nitrogen accumulation data presented here corresponds to the tree growth from November 1994, when the trees were planted, to April 1997, when the trees were harvested. The nitrogen accumulation at the start of the experiment, December 1995, was unable to be determined due to the destructive nature of sampling although it is assumed to be small as the trees were less than 1.5 m high by December 1995.

The biomass of *E. nitens* treatments contained about 172 g N $tree^{-1}$ (Table 4.2). It was not possible to quantify the amount of nitrogen taken from native soil source and effluent source. However if it is assumed, all nitrogen was sourced from the effluent, this equates to about 79 % of the nitrogen applied. *S. kimuyanagi* treatments stored 164 g N $tree^{-1}$ (75 %), while *E. saligna* stored 98 g N $tree^{-1}$ (44 %). Nitrogen accumulation was not significantly different between tree species because of the variation between the trees within the treatments ($P=0.05$).

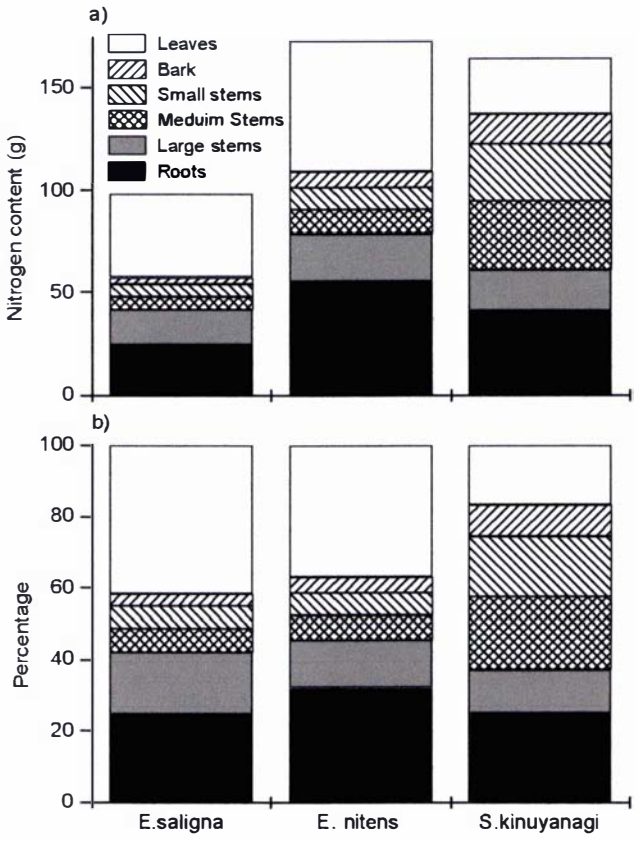

Figure 4.7 The nitrogen content of the biomass, by mass in each size group (a) and by the percentage within each biomass category (b) the size group of large stems includes the stump.

Table 4.2 Tree component nitrogen concentration and uptake.

Tree component	*E. saligna*		*E. nitens*		*S. kinuyanagi*	
	Conc. (mg N g^{-1})	N content (g)	Conc. (mg N g^{-1})	N content (g)	Conc. (mg N g^{-1})	N content (g)
Above ground						
Large stems	4.6	14.4	2.4	21.7	3.5	18.7
Medium stems	4.1	6.5	3.6	11.8	8.0	33.8
Small stems	5.7	6.2	6.0	10.7	15.0	27.7
Bark	5.5	3.2	5.2	7.6	17.6	14.4
Leaves	21.8	40.9	17.8	64.1	24.6	27.7
Total		71.2		115.8		122.3
Root system						
Stump	4.6	2.7	2.4	2.0	3.5	1.6
Roots	11.8	24.1	14.5	55.0	15.8	40.6
Total		26.8		57.0		42.2
Whole tree		**98**		**172.9**		**164.5**

The above-ground biomass represents a source of nitrogen that can be removed from the land treatment system. The above-ground biomass of *E. nitens* treatments contained some 116 g N $tree^{-1}$ (Table 4.2). This equates to about 53 % of the nitrogen applied. *S. kimuyanagi* treatments stored 122 g N $tree^{-1}$ (56 %), while *E. saligna* stored 71 g N $tree^{-1}$ (33 %). Nitrogen accumulation in above-ground biomass did not significantly differ between tree species this again being due to the large variation between the trees within the treatments ($P=0.05$).

Here the total nitrogen accumulation and percentage distribution of nitrogen within the trees is presented in Figure 4.7a and Figure 4.7b. Although the *E. nitens* and *S. kimuyanagi* took up similar amounts of nitrogen, the amount stored to different components varied between the tree species (Figure 4.7b). The Eucalypt trees are very similar in the nitrogen storage proportions with the largest storage being in leaves at around 40 %. The leaf storage in the Salix was almost half that at around 20 %. Thus the Salix trees have a larger percentage of nitrogen stored in the woody biomass compared to both Eucalypts. The proportion of nitrogen stored in large stems is similar for the three species. The Salix treatments have greater nitrogen-storage in the medium and smaller stems.

If a plantation density of 4000 stems ha^{-1} is assumed then the total nitrogen accumulation in all tree components, after 2.5 years of growth would range from 392 kg N ha^{-1} for *E. saligna* to 691 kg N ha^{-1} for *E. nitens*, with 658 kg N ha^{-1} for *S. kimuyanagi*. When considered in terms of annual removable nitrogen in above-ground biomass (ignoring litter fall) the nitrogen accumulation rates were 114 kg N ha^{-1} for *E. saligna*, 186 kg N ha^{-1} for *E. nitens* and 196 kg N ha^{-1} for *S. kimuyanagi*. These trees

were grown in lysimeters so any extrapolation of uptake rates to field plantations should be treated with caution.

4.3.4 Nitrogen leaching

The amount of nitrogen leached is inherently related to the soil water balance through the drainage volume. Details on the soil water budgets of this experiment are discussed in Chapter 3. The total nitrogen leached is the product of the leachate volume and the nitrogen concentration in leachate.

Ammonium concentrations in the leachate were negligible for all treatments. The rapid nitrification of ammonium to nitrate in the soil, combined with ammonium being strongly retained to cation exchange sites make it unlikely to be liable for leaching. Nitrate was found in significant quantities in the leachate. The drainage volume, nitrate content and amount of nitrate leached are shown separately for bare soil, *E. saligna*, *E. nitens* and *S. kimuyanagi* treatments in Figure 4.8, Figure 4.9, Figure 4.10, and Figure 4.11. Daily average values are presented for ease of comparison of months with differing number of days. It is important to note that at the finer scale of weeks and days, the values display much more variability. The results presented here start from January 1996, as drainage volume was not quantified for December 1995.

The nitrate concentrations in the leachate varied within the treatments throughout the experiment. Bare-soil leachate concentrations (Figure 4.8b) in January 1996 were initially just over the NZ drinking water standard (NZDWS) of 11.3 mg NO_3^--N. Leachate concentrations had risen to 4 times that limit by May. These then lowered a little when effluent irrigation ceased in winter and rainfall flushed the system. However once again the level increased to remain over 6 times the NZDWS during the second

irrigation season. The nitrate concentration in the leachate from the *E. saligna* (Figure 4.9b) treatments followed a similar pattern to the bare soil. However nitrate concentrations of the *E. saligna* treatments were lower than those under the bare soil. Following the harvest, nitrate concentrations for the two Eucalypt species reached peaks of 10-14 times the NZDWS. The *E. nitens* treatments leached significant quantities of nitrogen only during the first winter, and again after harvest (Figure 4.10). In the first winter, the *E. nitens* treatments leachate-nitrogen-concentrations were 2-3 times the NZDWS and following the harvest the concentrations were as high as 14 times the NZDWS. The *S. kimuyanagi* leachate concentrations remained close to the NZDWS until autumn of 1997. Concentrations were then consistently 6-7 times the NZDWS.

Quantities of nitrogen leached varied greatly between the treatments. The leachate of bare-soil treatments totaled about 207 g N lysimeter^{-1} (Table 4.2). This equates to about 92 % of the nitrogen applied. The *E. saligna* treatments leached 95 g N lysimeter^{-1} (44 %), while *E. nitens* leached 23 g N lysimeter^{-1} (11 %) and *S. kimuyanagi* leached 42 g N lysimeter^{-1} (19 %). Not surprisingly, the bare-soil nitrate leaching was significantly greater than all tree species ($P=0.05$). Nitrate leaching was not significantly different between the tree species because of the large variation in leaching within the treatments ($P=0.05$).

The total amounts of nitrogen leached (Table 4.3) represent a loading to the underlying groundwater of 512 kg N ha^{-1} year^{-1} for bare soil, 261 kg N ha^{-1} year^{-1} for *E. saligna*, 72 kg N ha^{-1} year^{-1} for the *E. nitens* and 114 kg N ha^{-1} year^{-1} for *S. kimuyanagi*. However it is pertinent to note that for the *E. nitens* and *S. kimuyanagi* treatments the major proportion of this leaching occurred after the trees had been harvested.

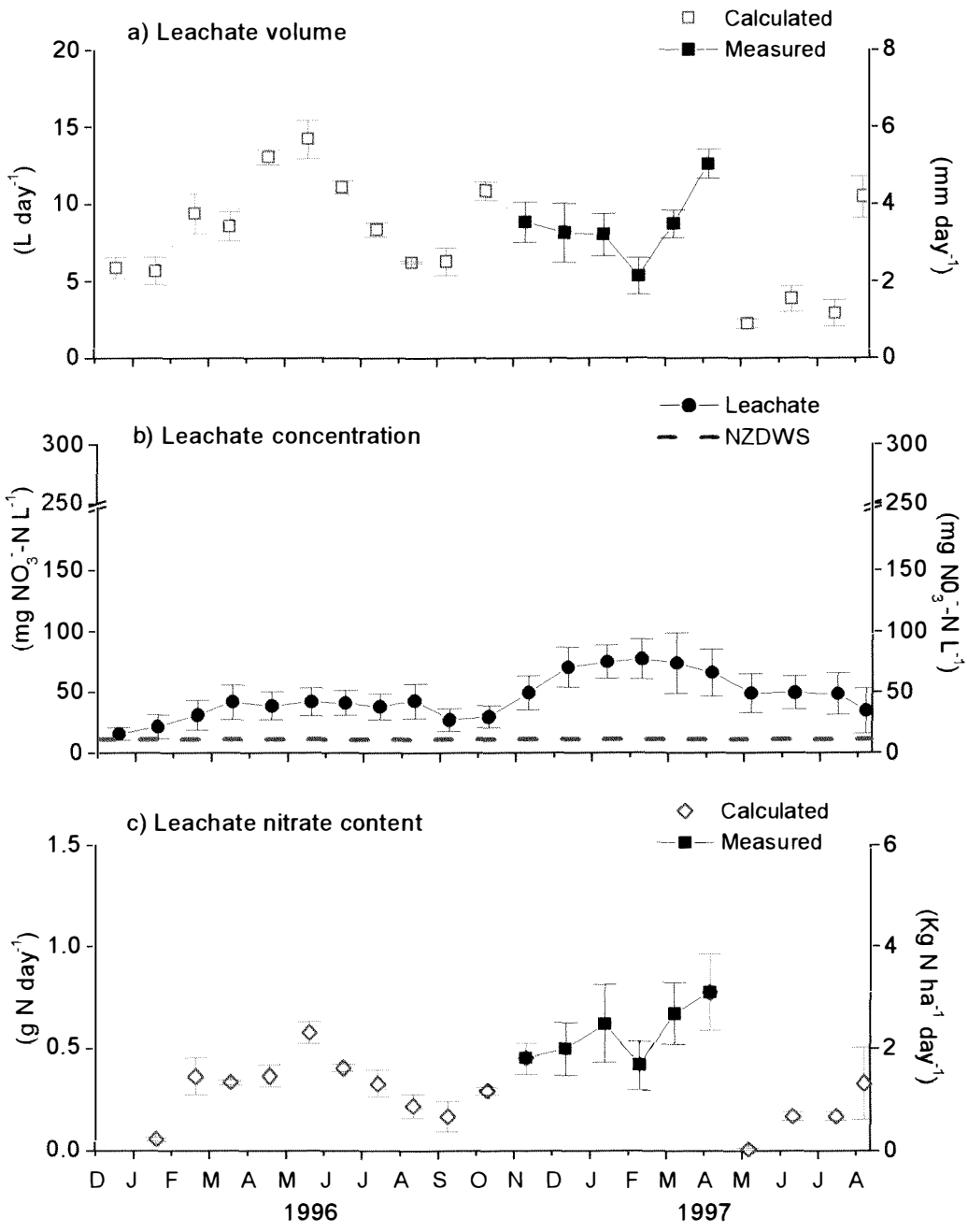

Figure 4.8 The bare-soil drainage volume (mean daily average and standard deviation) (a) and the leachate nitrogen concentration (monthly average and standard deviation) (b). Also shown is the New Zealand drinking water standard (NZDWS). In (c) is quantity of nitrogen leached (mean daily average and standard deviation).

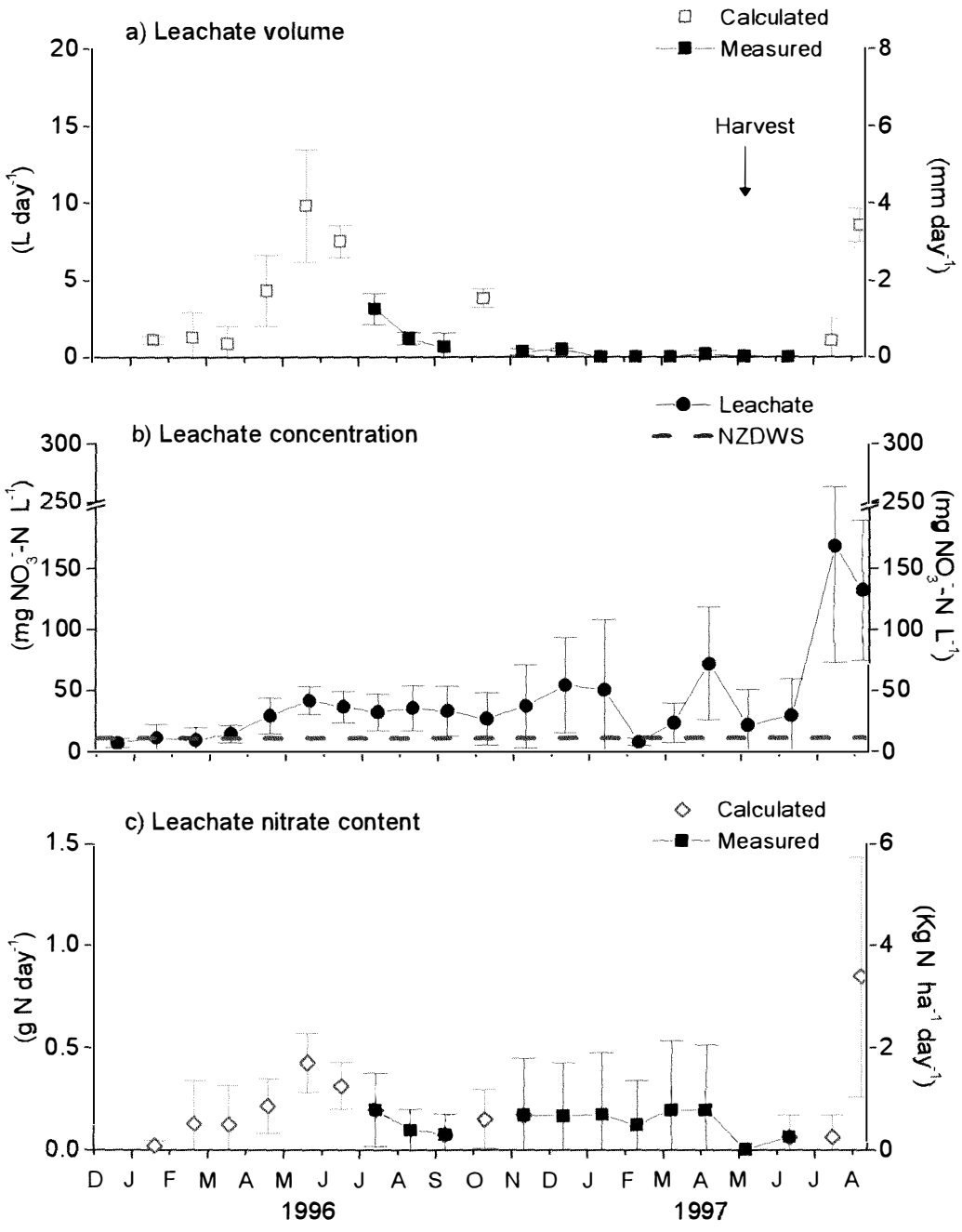

Figure 4.9 The *E. saligna* drainage volume (mean daily average and standard deviation) (a) and the leachate nitrogen concentration (monthly average and standard deviation) (b). Also shown is the New Zealand drinking water standard (NZDWS). In (c) is quantity of nitrogen leached (mean daily average and standard deviation).

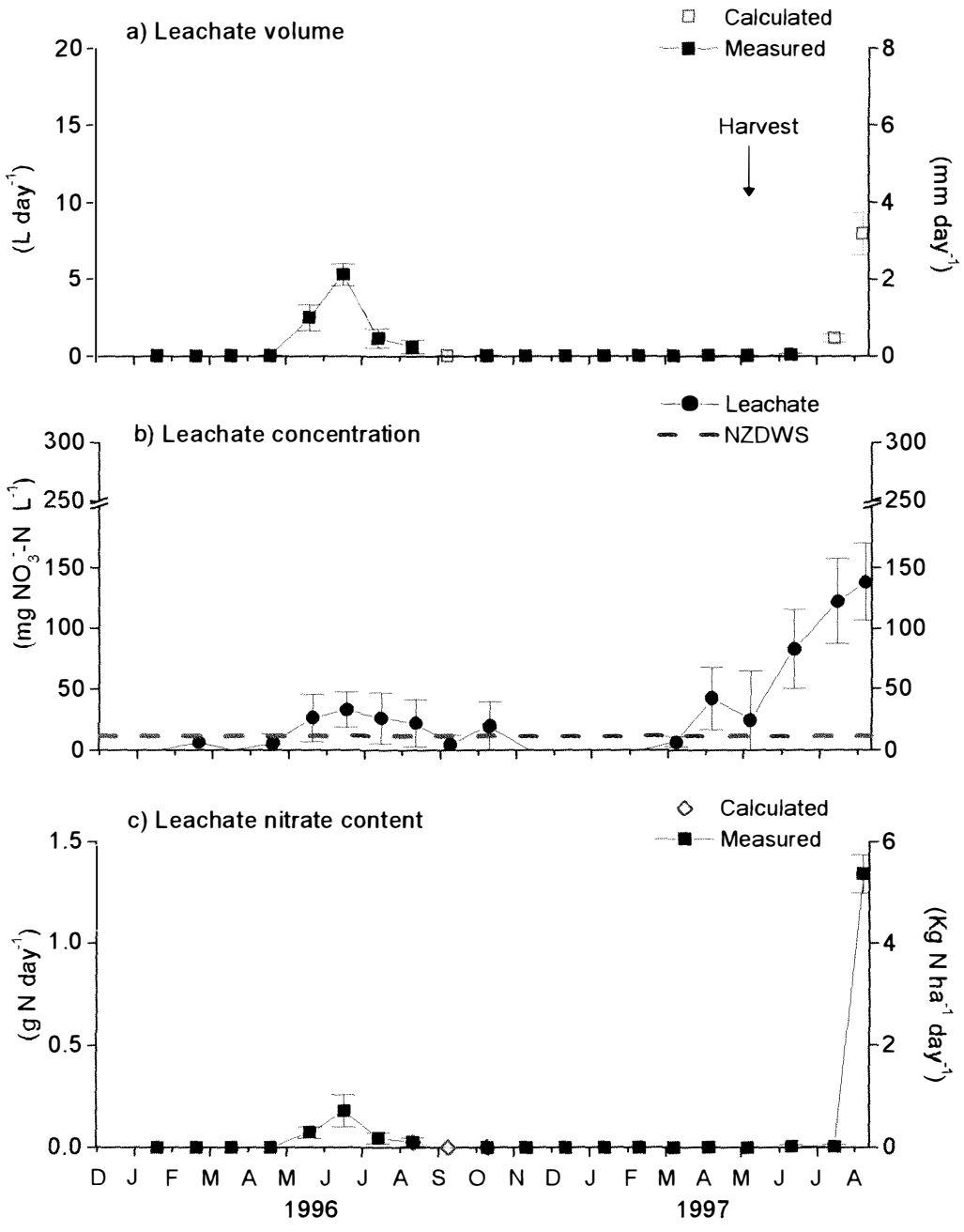

Figure 4.10 The *E. nitens* drainage volume (mean daily average and standard deviation) (a) and the leachate nitrogen concentration (monthly average and standard deviation) (b). Also shown is the New Zealand drinking water standard (NZDWS). In (c) is quantity of nitrogen leached (mean daily average and standard deviation).

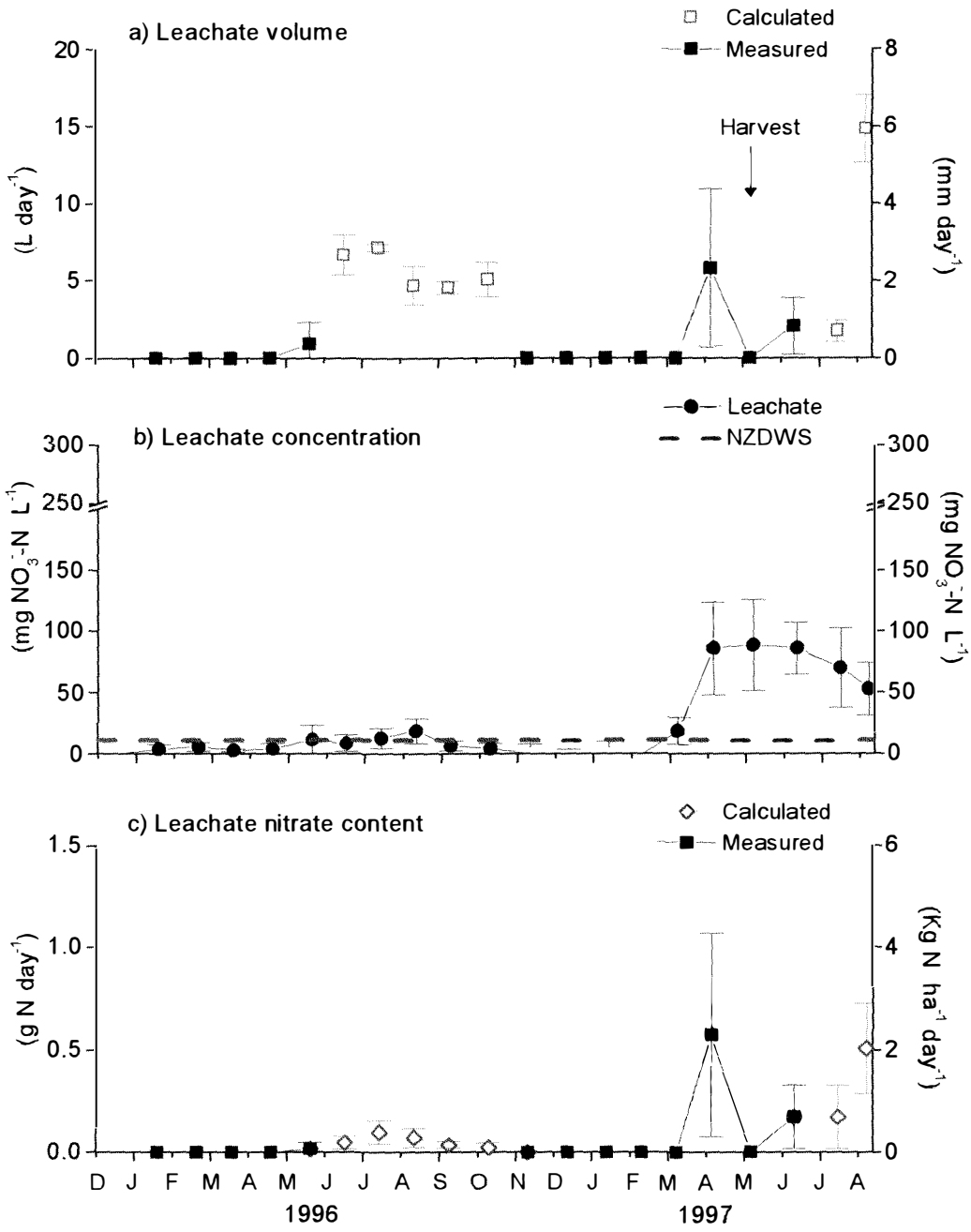

Figure 4.11 The *S. kinuyanagi* drainage volume (mean daily average and standard deviation) (a) and the leachate nitrogen concentration (monthly average and standard deviation) (b). Also shown is the New Zealand drinking water standard (NZDWS). In (c) is quantity of nitrogen leached (mean daily average and standard deviation).

Table 4.3 The seasonal nitrate leaching during the experiment. Expressed as drainage volume, D (mm), leachate concentration, C (mg N L^{-1}) and quantity of nitrogen leached, L (g N). Nitrate leaching was measured for individual events, each with a specific volume and nitrogen concentration. Numbers in italics contain some values of calculated drainage volume (see text for details).

	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Total
Starting	1/1/96	6/3/96	5/6/96	27/8/96	26/11/96	25/2/97	27/5/97	
Finishing	5/3/96	4/6/96	26/8/96	25/11/96	24/2/97	26/5/97	11/8/97	
No. Days	63	91	83	91	91	91	80	590[1]
Bare Soil								
D	*184.7*	*437.6*	*282.7*	*320.2*	262.9	*269.7*	*136.6*	*1894.3*
C	25.7	40.5	40.3	34.9	73.7	61.9	46.9	46.2
L	*12.2*	*38.5*	*26.3*	*27.6*	46.4	*40.8*	*15.2*	*207*
E. saligna								
D	*29.1*	*180.2*	*130.8*	*64.8*	6.7	2.3	*49.2*	*463.1*
C	10.5	28.5	34.8	32	38.8	37.5	103.0	40.7
L	*4.4*	*23.2*	*16.9*	*12.5*	14.3	11.0	*12.9*	*95.2*
E. nitens								
D	0.3	28.2	77.4	0.6	0.0	0.02	*48.8*	*155.3*
C	6.5	10.1	27.0	8.8	0.0	24.5	106.9	26.3
L	0.0	2.0	6.8	0.0	0.0	0.0	*13.8*	*22.6*
S. kimuyanagi								
D	0.8	10.3	*204.1*	*122.0*	0.0	65.8	*88.9*	*491.9*
C	4.5	5.9	13.3	3.7	0	66.4	74.9	*24.1*
L	0.0	0.5	*6.0*	*1.8*	0	16.1	*17.2*	*41.6*

[1]The full length of the experiment was 620 days so leaching from December 1995 is not included.

4.3.5 Nitrogen balance

The nitrogen balance of the four treatments differed (Table 4.4). Effluent volumes applied varied for the lysimeters, resulting in differing nitrogen loading to each treatment. Nitrogen loading from effluent application, however, did not significantly differ for the treatments (P=0.05). The total inputs for the 4 treatments tally to almost similar values. Whereas the contribution from various components of the total outputs differ between the treatments. The nitrogen outputs of bare soil are predominately through leaching. *E. saligna* outputs of nitrogen are evenly split between leaching and plant uptake. The nitrogen outputs of *E. nitens* and *S. kimuyanagi* are mostly by plant

uptake. Leaching makes up 12 and 20 % of nitrogen outputs for the *E. nitens* and *S. kinuyanagi* treatments, respectively.

Table 4.4 Nitrogen balance of the four treatments. All units are in g N lysimeter^{-1}.

Process	Bare Soil	*E. saligna*	*E. nitens*	*S. kinuyanagi*
Inputs				
Effluent inputs	219	212	223	216
Storage change				
NO_3^--N[a]	-39	-81	-18	-39
NH_4^+-N[a]	1	2	0	0
Total change	-38	-79	-18	-39
Outputs				
Plant uptake[b]				
Above ground	0	71	116	122
Below ground	0	27	57	42
Leaching[c]	207	95	23	42
Total outputs[b,c]	207	193	196	206
Net[d]	**-26**	**-60**	**9**	**-29**

[a]Assume depth =0.65 m.
[b]Includes uptake in the year of establishment.
[c]Excludes leaching during December 1995.
[d]Balance excludes mineralisation, volatilisation and denitrification

The nitrogen balance has been resolved from the input and output measurements made in this experiment. The processes of volatilization and denitrification were not measured in this experiment. The implication of this on the nitrogen balances will be discussed further in later Sections 4.4.5 and 4.4.6. The nitrogen balance assumes a soil depth of 0.65 m, for the calculation of changes in the mineral nitrogen concentration of the soil. Also, the nitrogen balance does not include leachate outputs for December 1995. The plant nitrogen uptake includes the period November 1994 to December 1995. That the mass balance of nitrogen does not tally to zero, will be discussed further in Section 4.4.

4.4 Discussion

This study has been successful in determining the nitrogen balance for the four treatments receiving dairy-shed effluent. The nitrogen balance was calculated as the difference between inputs, (effluent application) and outputs (plant uptake and leaching) and adjusted for changes in soil storage. Complete tally of the mass balance of nitrogen however was not achieved. Outputs and storage changes from *E. nitens* were 4 % less than inputs. However for bare soil, *S. kimuyanagi* and *E. saligna* treatments the balance exceeded the inputs by 12, 13 and 30 %. Some of the assumptions involved in the measurements and subsequent calculations of various components of the balance may contribute to this. Factors not measured in the balance, including volatilization and denitrification, may also play a role in not achieving the closure of mass balance. The implication of these on mass balance is discussed below.

4.4.1 Effluent application

In this study, dairy shed effluent contained on average 77 mg N L^{-1}. Nitrogen was predominately (98%) present in the ammoniacal form. This is similar to the value of 82 mg NH_4^+-N L^{-1} measured by Hickey et al. (1989) in a survey of dairy-effluent ponds in the Manawatu and Southland provinces of New Zealand. Effluent nitrogen concentrations ranged from 3-13 times greater than the NZDWS, and thus would require either extra treatment prior to discharge or large amounts of dilution in the water ways. Effluent concentration varied temporally, which contributed to larger nitrogen loading in the second irrigation season. Estimation of the inputs of nitrogen from effluent will have only small associated errors. So it is not thought that the estimation of nitrogen inputs presents a large error source for the nitrogen balance.

4.4.2 Changes in soil nitrogen status

Depthwise sampling of the soil in the lysimeters at the initiation of the effluent irrigation was not possible, because of the influence this would have had on the drainage characteristics of the lysimeters. The soil nutrient content was therefore assumed to be uniform throughout the soil profile. The soil in the lysimeters was repacked in 1994 and left for one year to allow for the trees to establish and the soil to settle. It is likely that the initial soil nitrogen distribution was indeed stratified through the packing of the lysimeters and following the year of unknown leaching and plant uptake. The influence of the initial profiles of soil nitrogen, on the nitrogen balance remain unquantifiable.

Changes in soil nitrogen were measured by two methods, one method analysed for total nitrogen and the other for mineral nitrogen. The nitrogen balance used mineral nitrogen to indicate the changes in the soil nitrogen status of the lysimeters. The amount of nitrogen applied in the effluent (0.10 mg g^{-1} soil) was very small compared to the total nitrogen content of the soil (approx. 1.5 mg g^{-1} soil). Thus the inference of temporal changes in soil nitrogen using total organic nitrogen content would involve measurement of a small change in a large number and therefore would have large associated errors. Mineral nitrogen content on the other hand (approximately 0.07 mg g^{-1} soil) provides a more accurate measured value of soil nitrogen, and easily changes relative to the inputs. Also, effluent applications were in mineral form. Mineral nitrogen content however, is known to be more seasonally variable. Thus sampling before and after the experiment presents only a 'snapshot' in the changes of soil mineral-nitrogen. There could well be no change in total mineral nitrogen over the 620 days of the experiment, but large changes may have occurred within that time period.

Changes in mineral nitrogen show a decrease in the amount of nitrate at the end of the experiment (Figure 4.5). Given the seasonal changes in mineral nitrogen it is difficult to ascertain the importance of this. The large volume of irrigation water applied prior to the end of the experiment flushed much nitrate from the profiles of all lysimeters (Figures 4.8, 4.9, 4.10 and 4.11). All lysimeters were leaching nitrate at the end of the experiment in August 1997. Furthermore, the soil cores were obtained two weeks after the end of the experiment. This late sampling creates a time period in which further leaching may well have occurred, especially with the absence of any plant uptake. It is likely that the estimation of soil nitrogen changes in this experiment has somewhat larger errors, relative to other components of nitrogen balance.

Nevertheless the observed lowering of soil mineral nitrogen levels during this experiment is consistent with measurements at other effluent irrigated sites. Polglase et al. (1995) and Falkiner and Smith (1997) both measured a decrease in soil nitrogen under effluent irrigated pines and Eucalypts at Wagga Wagga in Australia. They both concluded that losses of mineral nitrogen could be attributed to accelerated decomposition caused by wetting and drying cycles associated with irrigation resulting in the immobilisation of mineral nitrogen to organic nitrogen. This may be partly the cause of the loss of mineral nitrogen in this experiment. However it is likely the large flushing of nitrogen at the end of the experiment contributed to lowering soil mineral nitrogen content.

4.4.3 Plant nitrogen storage

The amount of nitrogen stored in the plant tissues at the start of the experiment, December 1995, was unknown. In the nitrogen balance some plant nitrogen accumulated during the establishment period, November 1994-December 1995, was

included in the balance. Thus storage for the balance period which runs from December 1995-August 1997 is over estimated because of inclusion of plant nitrogen accumulated during the establishment phase. This inclusion of plant uptake during the establishment period likely contributes to not achieving total mass balance. However, nitrogen uptake during this phase was likely small as the trees were less than 1.5 m high at the start of the experiment. If tree uptake is estimated as being distributed evenly over the 2.5 years of the experiment, the uptake in this establishment phase would be around 40 % that of the total uptake. However, it is assumed that uptake was lower than this in the first year in the absence of any irrigation or fertilisation, thus it is likely lower than 30 %. In any case, the removal of plant uptake during this establishment phase from the nitrogen balance would lower the amount of nitrogen outputs for the tree treatments.

Assuming all the plant nitrogen is derived from the effluent irrigation, the trees in this study stored 44-79 % of nitrogen applied, thereby preventing some nitrogen leaching into water ways. The potential to remove nitrogen from a land treatment SRF system is realised through regular removal of the above-ground biomass. In the present study, the above ground portions contained 33-56 % of nitrogen applied. Hopmans et al. (1990) measured nitrogen uptake rates in above-ground biomass of a number of species (*E. saligna, E. grandis, E. camadulensis, Populus deltoides, P. deltoides* x *P. nigra, Casuarina cunninghamiana* and *Pinus radiata*). These trees were receiving municipal effluent at Wodonga, Australia. Effluent application added the equivalent of 400 kg N ha^{-1} year over a 44-month period. Hopmans et al. (1990) found no significant difference in nitrogen uptake among the tree species. The nitrogen uptake averaged only 19 %, with a maximum nitrogen uptake of 28 % relative to the inputs of nitrogen from effluent. The trees in the Wodonga experiment appear to be less efficient at taking up

nitrogen than were trees in this experiment. One possible explanation for this is that larger quantities of water were applied at Wodonga, possibly increasing nitrogen leaching thereby concomitantly decreasing tree uptake. The percentage uptake figures presented for both studies assume all nitrogen is sourced from effluent irrigated nitrogen. It is likely however that some nitrogen is sourced from the soil. In the case of the lysimeter experiment all uptake in the establishment phase of the experiment is from soil nitrogen.

Nitrogen removal from the system in this experiment was 113-195 kg N ha^{-1} yr^{-1}. It is likely that the rates of nitrogen storage measured in this study would have been greater, had tree growth been further optimised. As outlined in the earlier chapters of this thesis the growth of the larger *E. nitens* and *S. kimuyanagi* trees were likely lowered by water stress during the summer periods. Any increased biomass production would have been expected to provide greater nitrogen uptake. It is noted that the uptake rate of the *S. kimuyanagi* may have been higher, had they maintained leaves throughout the second irrigation season. The leaf nitrogen concentration of the *S. kimuyanagi* was high. Thus a greater mass of leaves harvested, may have increased nitrogen storage. Nevertheless, it is likely the higher concentrations of nitrogen in the small stems and bark of the *S. kimuyanagi* trees reflects translocation of nitrogen to these components prior to the leaf drop.

Litter fall nitrogen was not estimated during the experiment, plant uptake and nitrogen immobilisation may thus have been greater than the storage rates estimated here. Litter fall need not be restricted to the leaf fraction, for bark and reproductive material were also shed during this experiment.

The leaf nitrogen content is a substantial part of the nitrogen balance. Thus harvesting of the trees should aim to remove leaves. The leaf litter return of deciduous trees in the annual drop could be minimised by such management strategies as coppicing. On dairy farms the use of the small branches and leaves as stock fodder presents an opportunity to remove nitrogen from the site at the end of the irrigation season.

4.4.4 Leachate nitrogen losses

Not all the nitrogen applied was taken up by the trees. Some nitrate did pass beyond the rootzone where it would be expected to continue its passage downwards to contaminate groundwater. The quantity of drainage going to groundwater does have some consequences for aquifer management in terms of the potential rise in the groundwater table. However, it is the quality of the leachate that is of most concern. The quality of leachate is judged here in terms of the nitrate concentration and total nitrogen loading. The predominance of nitrate in leachate indicates microbiological activity is rapidly converting ammonium applied in effluent to nitrate. Nitrate is only weakly adsorbed by soil, and therefore it moves freely in draining water.

Leaching during the first month of the experiment was unknown and was not included in the mass balance calculations. During this period leaching was observed from all lysimeters. Leachate volumes for some periods were calculated via a water balance in the absence of direct measurements. This estimation and is implications are fully discussed in Chapter 3. It is not foreseen that the errors associated with these calculations are large, in comparison to other components of the nitrogen balance.

The differential water use of the treatments had implications for the concentration and quantity of nitrogen leaching beyond the root zone. Consequently the following discussion of nitrogen leaching does repeat some aspects of the water balance discussion.

Bare-soil evaporation rarely exceeded the water inputs from rainfall and effluent application. So drainage volume was consistently high. Bare-soil leachate nitrogen concentrations were lower than the effluent nitrogen concentrations as a result of dilution by rainfall and some small retention/immobilisation by the soil. Nevertheless the concentrations leached closely reflected the concentrations of the effluent being applied, being higher in the second irrigation period than in the first. Concentrations were also lower in periods when no effluent was applied. Overall, the bare-soil leachate nitrate concentrations and nitrogen content remained high throughout the experiment, with 95 % of applied nitrogen being leached.

Trees provide further complexity to the hydrologic cycle due to their greater evapotranspiration (ET). When ET exceeds water inputs for sufficiently long periods, drainage ceases. Plants also provide an extra sink for nitrogen in the system.

The ET of the *E. saligna* trees was lower than the inputs from effluent irrigation and rainfall for the first 11 months of the experiment. This was initially due to the small leaf area. In winter it was due to low potential evapotranspiration. Drainage volumes of *E. saligna* followed a similar pattern to that of bare soil with the changes in drainage volume coinciding with water inputs. The ET of *E. saligna* was greater than bare-soil evaporation, thus the drainage volume was less than that of bare soil. Nitrogen uptake

by the *E. saligna* trees reduced the nitrogen concentrations and quantity of nitrogen in the leachate in comparison to the bare-soil treatments. Growth of the *E. saligna* trees, combined with increased potential evapotranspiration in spring, increased ET above the rate of water inputs in October of 1997. Consequently drainage stopped. Thus applied nitrogen had no mechanism to travel to the ground water until after water inputs again exceeded ET. The greater leaf area and tree size thus play an important role in controlling the loss of nitrogen by leaching. Greater leaf area increases water use of the trees. Greater tree size increases the amount of nitrogen stored in trees.

The *E. nitens* and *S. kimuyanagi* treatments had higher leaf area than the *E. saligna* trees at the start of the experiment (Figure 2.4). During the first summer, January-February 1996, the ET of *E. nitens* and *S. kimuyanagi* treatments was greater than water inputs and hence these trees leached minimal amounts. During the autumn, winter and spring of 1996 the *S. kimuyanagi* and *E. nitens* leached a similar amount of nitrogen. However the *S. kimuyanagi* trees maintained much lower nitrogen concentrations in the leachate. These lower concentrations were a result of greater hydraulic loading to the ground water, as described in Section 3.3.3. Lower concentrations were a result of greater dilution from the increased drainage volume of the *S. kimuyanagi* treatments. The difference in concentrations was most pronounced in the winter period. The *S. kimuyanagi* leachate concentrations on average were just above the NZDWS during winter of 1996 (Table 4.3). In comparison, the Eucalypt treatments were on average 2-3 times greater than the NZDWS during winter of 1996.

Following the winter period of 1996, ET by the 3 tree species exceeded water inputs. Different rates of water use continued to influence both quantity and quality of the

leachate. Water use by the *E. saligna* trees fell below the water inputs on several occasions through the spring-summer period. This resulted in a small amount of nitrogen being leached, at a range of concentrations. No drainage occurred under the *S. kinuyanagi* and *E. nitens* treatments during the summer of 1997. During this period the *S. kinuyanagi* rapidly exhausted the available water supply, exhibiting a premature shedding of many leaves (Figure 3.7, Figure 2.4). When irrigation and rainfall inputs increased again the water content of the soil profile rapidly increased. Leaching was measured again for *S. kinuyanagi* in autumn of 1997. Interestingly, the *E. nitens* did not leach in this autumn period, as the rainfall and irrigation inputs were not greater than the soil-water storage capacity and the ET demand.

The greater tree size of the *S. kinuyanagi* and *E. nitens* trees resulted in a greater amount of nitrogen storage in the trees at harvest (Section 4.3.3). This also accounted for lower amounts of leaching up until the harvest.

The harvest of the trees, in April-May 1997, returned all treatments to a bare-soil scenario. Following harvest, there was uncharacteristically low rainfall. When the rains arrived, and the irrigation of water was applied, high drainage volumes for all treatments were observed. The concentrations of nitrogen in the leachate were high for all tree treatments, ranging up to 6 times the NZDWS for the Salix, and to 9 times the limit for the Eucalypts. It is suspected that high rates of mineralisation of leaf litter nitrogen, and soil organic nitrogen, may have contributed to this, especially during the warm winter, following harvest. Higher leachate nitrogen concentrations following the harvest could also be linked to higher concentrations of ammonium in the effluent during the second season.

Leachate quality is likely to be a critical factor in determining the sustainability of SRF systems for land treatment. Literature values of nitrogen leaching from SRF land treatment sites are scarce. However, comparisons with pasture-based land treatment systems are possible. Di et al. (1998a, 1998b) applied dairy-shed effluent to pasture grown in lysimeters. The application rate was 400 kg N ha^{-1} $year^{-1}$. In their 2-year experiment they measured annual leaching losses of just 8-25 kg N ha^{-1}. Silva et al. (1999) monitored pasture grown in lysimeters receiving varying rates of dairy-shed effluent. The treatments received 0, 200 and 400 kg N ha^{-1} $year^{-1}$ and leached 3.2, 6.3 and 10 kg N ha^{-1} $year^{-1}$ (Silva et al. 1999). Carey et al. (1997) applied pig slurry to pasture grown in lysimeters. Their study applied slurry at 200 and 400 kg N ha^{-1} $year^{-1}$, and recorded leaching of 15 and 58 kg N ha^{-1} $year^{-1}$.

The application rate in the present study was 513 kg ha^{-1} $year^{-1}$ and leaching of nitrogen was lowest for *E. nitens* (72 kg N $year^{-1}$) with 50 percent more leaching occurring for the *S. kinuyanagi* (114 kg N $year^{-1}$). Given the increased loading rate of both nitrogen and water in the present study, these higher values of leaching are expected. The high nitrogen application rate is not seen as sustainable for SRF land treatment systems due to the high concentrations and quantities of nitrogen leaching to the ground water. Latter-on, the computer model will be used to assess the impact of lowering the annual nitrogen loading rate on the long-term sustainability of land treatment systems (Chapter 5).

It is possible that in the present study, the amount of leaching could have been further minimised through more optimal scheduling of the effluent irrigation. Effluent irrigation in this experiment was always applied weekly, at the same rate, throughout the

irrigation periods. Matching irrigation more efficiently with plant water and nitrogen demands, may result in greater uptake of water and nitrogen. This should then lower the amount available for leaching. The development of the computer model, in the next chapter, will enable such management options to be investigated.

A large proportion of the nitrogen leaching for *E. nitens* and *S. kimuyanagi* occurred following coppicing. Furthermore, leaching after the termination of the experiment in August 1997 would be expected to remain high until the trees grow back. The stumps of the trees were already observed to have coppice shoots by September 1997. The large proportion of leaching during the coppicing phase must be addressed if SRF systems are to be used for land treatment of dairy shed effluent. There is a need to investigate the effect of management practices on leaching during the coppicing phase.

Factors that may impact on the leaching during the coppicing phase include harvest frequency i.e. number of years between harvest and harvest timing within the year. Other factors include the effluent loading rate, and harvest date in relation to cessation of effluent application. The development of the computer model may allow these aspects of management to be investigated.

4.4.5 Volatilization

It is probable that volatilization contributes to the outputs of the nitrogen balance of the systems. Volatilization was not measured during the experiment. However, spray irrigation with aerosol production combined with the high ammonium content and alkaline characteristics of dairy-shed effluent would suggest that NH_3 volatilization could be an important pathway for nitrogen loss from the system. Ammonia volatilization losses ranging from 10-99 % of the applied nitrogen from the surface

application of wastes have been measured. Such losses depend on the edaphic and environmental factors (Beauchamp et al., 1982; Schilke-Gartley and Sims, 1993).

Smith et al. (1996) found ammonia volatilization to occur mainly in the first 24 hours following an effluent-irrigation event. This is when 'free' effluent water would be evaporating from the soil and plant surfaces. Smith et al. (1996) suggest that in the absence of NH_3 flux measurements, estimation of the NH_3 losses from bare soil can be made by multiplying the effluents ammoniacal-nitrogen content by the evaporation on the day of application. Here, their approach is utilised to estimate the magnitude of volatilization losses for this experiment. Their study of bare soil evaporation was done at temperatures above 27°C. To adjust for temperature differences in New Zealand, here it is assumed that half the amount of daily evaporation should be utilised for estimation of volatilization. In this study, effluent was applied on 55 days at an average concentration of 77 mg N L^{-1}. The maximum evaporation rate has been established to be about 2.5 mm day^{-1} (Section 3.2.6). Assuming the maximum evaporation rate on each effluent application day, and an average concentration of the effluent, the estimated volatilization losses are 5.3 g N. This equates to less than 3 % of the nitrogen inputs. So overall it is concluded that the contribution of volatilization to nitrogen outputs of the system are small.

This approach to calculation of ammonium volatilization will also be considered in the modelling chapter. Of the four treatments in this experiment, it is likely that NH_3 losses were higher for the bare-soil treatments. The irrigation of effluent under the forest canopy may have lowered volatilization rates due to less evaporative loss, lower soil surface temperature and lower wind-speed. Volatilization loss, being an output in the

nitrogen balance, would mean that its inclusion is likely to push the balance of this experiment more negative.

4.4.6 Denitrification

Denitrification losses in this study were likely to be small. Although denitrification was not measured in this experiment, the conditions which promote denitrification have received much investigation (Firestone, 1982; Myrold and Tiedje, 1985; Sextone et al., 1985; Ruz-Jerez et al., 1994; Monnett et al., 1995). The primary condition required for denitrification is extended periods of the soil moisture content above field capacity (Ruz-Jerez et al., 1994). The influence of increased soil moisture on denitrification is due to a reduction in oxygen concentration that is essential for denitrification to occur. In the present lysimeter experiment, prolonged periods of soil moisture content above field capacity were rare, due to the free draining nature of the soil. The annual rate of denitrification in pasture fertilised at 400 kg N ha^{-1} $year^{-1}$ in the Manawatu was measured at 19.3 kg ha^{-1} $year^{-1}$ (Ruz-Jerez et al., 1994). In the lysimeter experiment, the rates of annual denitrification for bare soil were possibly higher than this rate due to greater hydraulic and nitrogen loading. However, even if denitrification of bare soil is assumed to be double that figure, its contribution to the nitrogen balance remains small at around 5 % of the total amount of nitrogen applied.

Soil moisture contents of the tree treatments were generally lower than those of the bare-soil treatments. Thus, it is assumed that denitrification in the tree treatments would have been less than the bare soil treatment. Of the tree species, it is likely that denitrification was a source of nitrogen loss (albeit small) for the *E. saligna* treatments. Indications that denitrification rates will be low for the trees are consistent with results

from the Rotorua Land Treatment System (RLTS). In the RLTS the denitrification rate was determined to be < 3 kg N ha^{-1} year (Barton et al., 1998).

4.5 Conclusions

In this study, trees have been shown to improve effluent treatment because high evapotranspiration rates reduced the volume of leachate passing beyond the root zone. Further, uptake of nitrogen by the trees reduced the quantities of nitrogen available for leaching. In this study, both *E. nitens* and *S. kimuyanagi* were more effective than the other 2 treatments evaluated for land treatment. The low nitrogen concentration in the leachate under the *S. kimuyanagi* is the key criterion, which determines the suitability of this tree species for land treatment of effluent. The low total loading of nitrogen to the groundwater of the *E. nitens* treatments is the key criterion in determining *E. nitens'* suitability. Both of these species showed high levels of biomass production and nutrient accumulation.

However nitrogen concentrations in the leachate of all treatments were greater than the New Zealand drinking water standard during certain periods of the experiment. The leachate nitrogen concentrations might have been reduced if the amount of nitrogen applied in the effluent was reduced. Several key aspects of management practices thus require further investigation. To explore these experimentally would be time consuming and costly. The computer modelling section now sets out to provide a platform for investigating the optimisation of land treatment system design and management.

Chapter 5 Modelling

5.1 Introduction

Land treatment systems must adhere to the requirements of the Resource Management Act (1991). The prime sustainability concern for land treatment of dairy-shed effluent is whether the nitrogen content of the applied effluent will increase the leaching of nitrogen to groundwater. Thus, there is a requirement for knowledge of the environmental impact of management practices and design decisions. This understanding can be enhanced through controlled experiments and apt computer modelling.

Experiments can, with a large degree of effort and commitment, provide information on the impact of certain management decisions under a given set of conditions. Extrapolating this knowledge to other situations is possible through the development of theoretically sound computer models. Such models aim to synthesise our understanding and allow us to generate predictions that can demonstrate the effects of a change in management practice. Such models can then be used as decision support tools.

The focus of this research project through the field experiment was to understand better the key processes of water and nitrogen movement in SRF land treatment systems. These investigations were aimed toward developing modelling tools for predicting the fate of applied water and nitrogen in SRF land-treatment systems. The modelling results may enable farmers and regulatory authorities to assess the potential performance of effluent land-treatment-systems, and to develop sustainable strategies.

5.2 Model selection

A wide range of modelling approaches have been developed to describe soil-water-and chemical movement in soils. These models range in complexity from simple analytical models e.g. LPM (Van Genuchten, 1981, Ling and El-Kadi, 1998) to fully numerical models e.g. WAVE (Vanclooster et al., 1994).

The fundamental question remains as to the degree of model complexity required to simulate agrochemical movement to groundwater. According to Hutson and Wagenet (1993), the most appropriate model depends on the desired outcome.

Numerical models are only appropriate when all the inputs, sinks, sources and transformation processes can be parameterised. Complex numerical models have been developed to simulate the fate of nitrogen in the soil-plant system e.g. WAVE, (Vanclooster et al., 1994). However, their application is often difficult due to large amounts of information being required to run the model. In cases where data are limited, and where there is a need for long-term and multiple simulations, less sophisticated and more-pragmatic, analytical models may be more appropriate (Hutson and Wagenet, 1993; Ling and El Kadi, 1998). For example, to undertake a risk assessment of a land use practice over longer time periods with an incomplete knowledge of soil processes, simpler analytical models may suffice (Green and Clothier, 1999). In any case, a more-complex processed-based model does not always guarantee more realistic results (de Willigen, 1991). The choice of model depends on the purpose of the exercise, and this purpose determines the degree of theoretical rigour and the amount of data required (Hutson and Wagenet, 1993).

For the assessment of land-treatment systems, it is likely that only limited information will be available for the site's soil properties. With this in mind, a simple modelling approach was adopted here to predict nitrate leaching under SRF. The important features of the model are that it attempts to incorporate the long term changes in both the system performance, and the key element of the weather. Ultimately, the model will be used to investigate the effect of changes in management practice on sustainability of the system in terms of nitrogen leaching. The process of the model development and use is shown in Figure 5.1 and is further discussed here.

The model is a lumped parameter model (LPM), similar to that recently developed by Ling and El-Kadi (1998). This LPM model was chosen initially because the predictions from the Ling and El-Kadi (1998) model gave results that were in very good agreement with field measurements of nitrogen movement in the root zone. Indeed their simple model gave even better predictions than did a more-complex water and nitrogen transport model derived from a complete numerical solution of the transport equations.

The LPM model used here was developed jointly with Steve Green and Brent Clothier of HortResearch (Clothier, Green and Roygard, 1999). Steve Green coded the software. Although this model uses a similar structure to Ling and El-Kadi (1998) to determine the fate of nitrogen, the model also, in addition, incorporates crop growth and uptake of nitrogen based on similar principles to King (1993). Nitrogen mineralisation is also included in the model following Johnsson et al. (1987). Thus the model presented here differs from the three references above for it encompasses not only nitrogen transport and fate, but also crop growth, harvest and litter return, and mineralisation. This model also incorporates inputs of effluent to the soil-plant-atmosphere system.

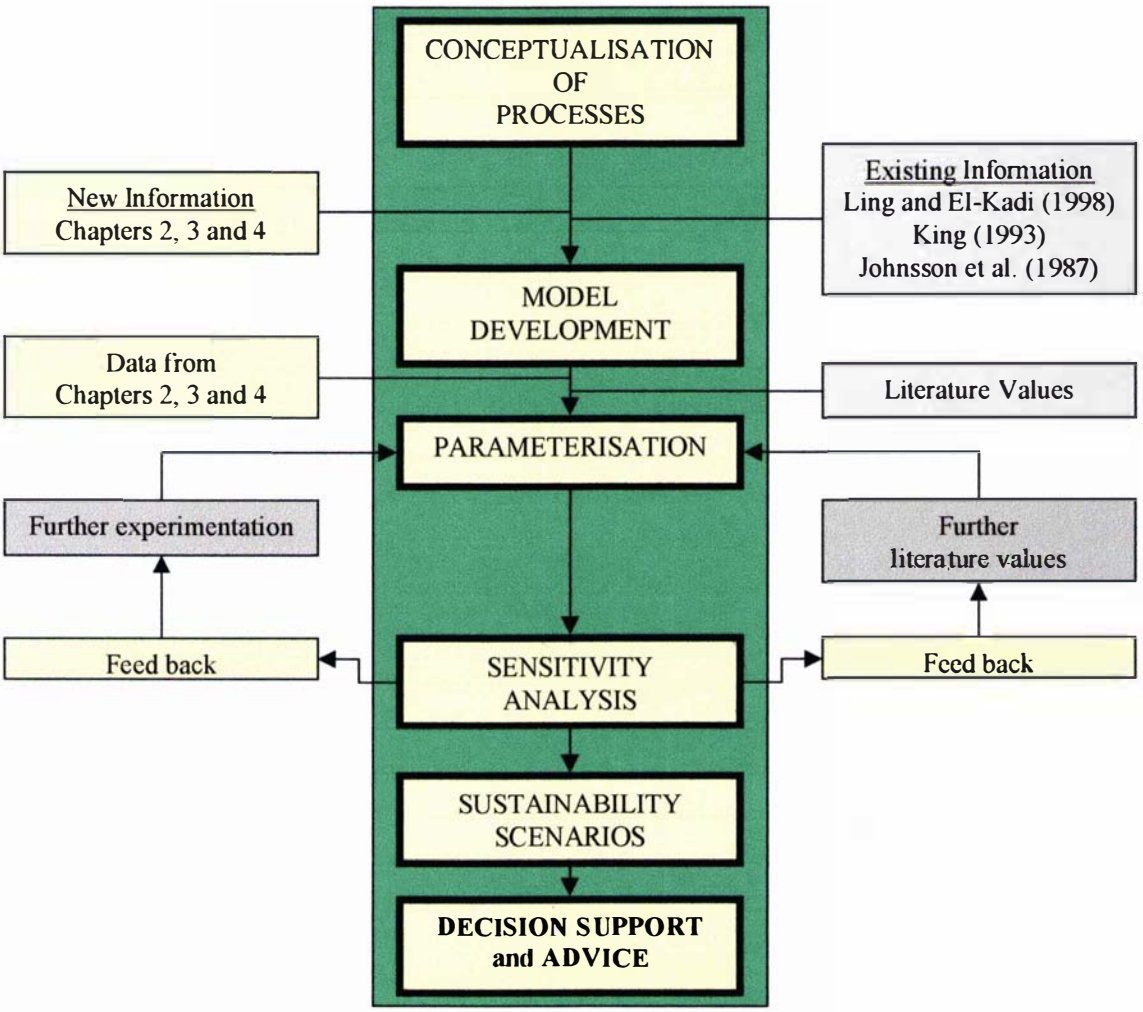

Figure 5.1 Flow chart of the model development for decision support. Lighter coloured boxes indicate the emphasis of this chapter.

Though untested, the LPM model used here has been used previously to investigate the sustainability of land treatment of municipal sewage effluent on Manawatu soils (Clothier, Green, and Roygard, 1999). This LPM model has also been used to study the irrigation requirements of field crops in the Auckland Region (Green and Clothier, 1999). Another application of the model was an investigation of the nitrogen leaching under grazed dairy pasture (Watt et al., 1998). The work reported here, is the first time that the model has been parameterised for a SRF land treatment system for dairy-farm effluent.

This chapter investigates the suitability of the simple LPM model for assessing the fate of water and nitrogen in a land treatment system. Predictions from the model are compared with the measurements recorded during the field experiment. A sensitivity analysis of the input parameters is also carried out to identify which parameters have the largest influence on the model output. The model's application as a decision support tool is then demonstrated through assessment of the sustainable nitrogen loading rate for dairy-shed effluent applied to a SRF plantation in the Manawatu region of New Zealand.

5.3 Modelling Methodology

The lumped parameter model combines the mechanisms of water transport through the root zone, with nitrogen transport and transformation processes occurring in the soil. Nitrogen transformations include natural processes such as mineralisation and denitrification, and volatilization. A full mathematical description of the model is given in Appendix A. Here, a brief description of the model is given along with a discussion of the assumptions, the inputs and the outputs.

The model considers the root zone to be one dimensional, comprising of a uniform soil for which the user provides the hydraulic and chemical properties. The root zone extends to a depth z_R (mm) (Figure 5.2).

Water flow through the soil is modelled according to the balance between inputs and outputs of water within the rootzone (Figure 5.2). The frequency and timing of effluent irrigation are included as input parameters. Effluent irrigation can be as supplied by the user or applied at regular intervals, e.g. once a week, during certain weeks of the year. Plant uptake is dependent on weather, the physiological stage of the plants and the soil

water content. Plants can tolerate a degree of water deficit but their rate of water uptake will decline if the soil dries below a given level. Drainage of water below the root zone is calculated through a simple analytical drainage model (Sisson et al., 1980), that considers the soil's hydraulic properties and the amount of water in the soil profile between z_R and the soil surface.

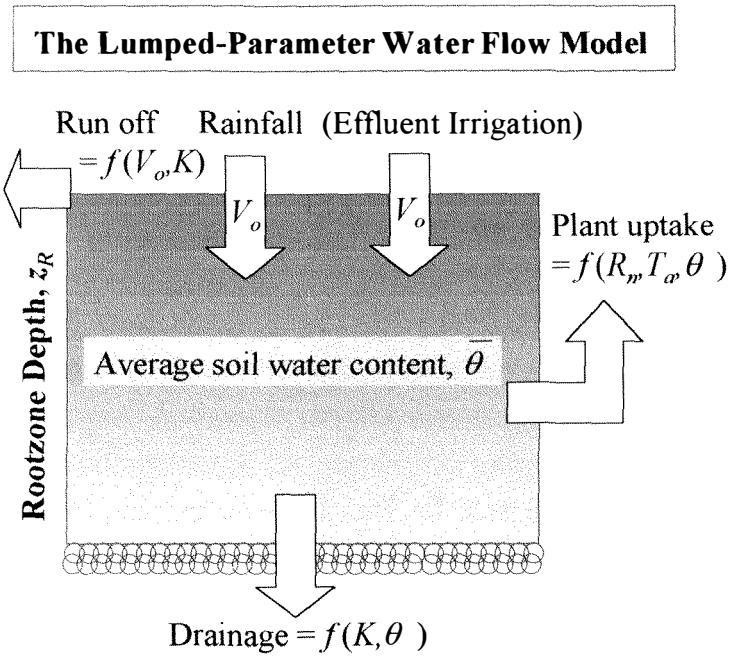

Figure 5.2 The lumped parameter model for water flow. Rain or effluent falls on the soil at a rate V_o, and depending on the soil's hydraulic conductivity K, it may either enter or run off. The soil has an average water content $\bar{\theta}$. Drainage is a function of K and $\bar{\theta}$. Plant uptake of water depends on net radiation R_n, air temperature T_a, and $\bar{\theta}$.

As water moves through the soil it carries along with it any chemicals, such as mineral nitrogen, that are in solution. Thus, drainage water moving beyond the rootzone has the same nitrogen concentration as that predicted for the soil solution on the day of leaching. Therefore, the fate of any surface applied nitrogen will depend on the processes that dictate water movement in the soil. Nitrogen flow through soil is modelled from a consideration of mass balance between the inputs, transformations and outputs of nitrogen (Figure 5.3). Mineralisation is the result of complex nitrogen transformations occurring within the soil. Here, the model considers the soil to have a

pool of organic nitrogen (derived from resident soil biomass and fresh organic matter) and a pool of mineral nitrogen in the form of ammonium and nitrate. First-order kinetics are used to describe the various transformation processes (decomposition of organic mater, mineralisation, and denitrification) with the rate constants being moderated by soil water content and temperature. Nitrogen uptake by the plants is based on plant growth, and the nitrogen content of the plant tissue. Uptake is reduced if the store of soil nitrogen falls below a critical value. Any mineral nitrogen leaving the bottom of the root zone will travel downwards and eventually enter the groundwater as a contaminant. It is therefore important to match the application rates to the assimilation capacity of the soil–plant atmosphere system, in order to limit leaching losses.

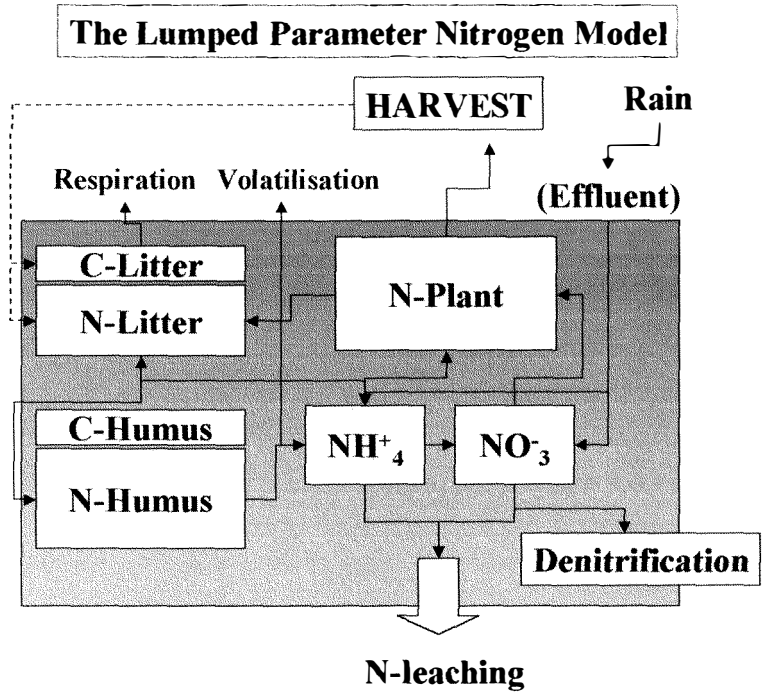

Figure 5.3 Schematic of the Lumped Parameter Model of nitrogen dynamics in the soil. Two linked pools of nitrogen are considered, the mineral pool of ammonium and nitrate, and the organic pool. The organic pool is divided into a fast cycling litter pool, and a more stable humus pool.

5.4 Simulation of the lysimeter experiment

5.4.1 Introduction

In this section, the model predictions of water and nitrogen fate are tested against measurements from the field experiment. Here, only the bare soil and *E. nitens* treatments of the field experiment are modelled. These two treatments represent the greatest and least amount of nitrogen leached. The model was not intended to simulate a deciduous tree as it does not simulate seasonal differences in leaf fall. Thus the *S. kinuyanagi* treatment was not considered for modelling.

5.4.2 Determination of parameters for the LPM model

The lumped parameter model requires over 60 input parameters to describe a given scenario, as can be seen in Table 5.1 and the example parameter files (Appendix B). Certain parameters are clearly defined, e.g. the parameters controlling the rate and timing of effluent application rate. Rainfall inputs and weather conditions are as measured for the lysimeter-study. Rainfall interception has already been accounted for in the input rainfall data. Thus for these comparisons, the effective rainfall coefficient is set to 1.0 times the total input rainfall. Modelling parameters, which describe the soil, the water uptake, the nitrogen uptake and the nitrogen transformation processes occurring in the root zone were either determined from the field study results or taken from literature values. Parameter files used for simulation of the bare soil and *E. nitens* are in Appendix B.1 and Appendix B.2.

The important soil hydraulic properties are maximum water content (θ_s), field capacity (θ_f), and the saturated hydraulic conductivity (K_s). Individual lysimeters exhibited large

Table 5.1 Parameters of the LPM utilised for the simulation of the Bare soil and *E. nitens* treatments of the lysimeter experiment.

Parameter[1]		Unit	Bare soil	*E. nitens*
Water movement				
Maximum water content	θ_s	$m^3\ m^{-3}$	0.45	0.36
Field capacity	θ_f	$m^3\ m^{-3}$	0.40	0.31
Wilting point	θ_w	$m^3\ m^{-3}$	0.08	0.08
Saturated hydraulic conductivity	K_s	$mm\ day^{-1}$	750	750
Beta constant	β		0.0293	0.0293
Soil bulk density	ρ_b	$Mg\ m^{-3}$	1.26	1.26
Root zone depth	z_R	m	1	1
Maximum soil evaporation	E_s	$mm\ day^{-1}$	2.5	2.5
Nitrogen parameters				
Nitrate adsorption	k_{DN}	$L\ kg^{-1}$	0	0
Ammonium adsorption	k_{DA}	$L\ kg^{-1}$	5.0	5.0
Nitrification	k_2	day^{-1}	0.2	0.2
Denitrification	k_3	day^{-1}	0.006	0.006
Denitrification zone below θ_f	δ_D	$m^3\ m^{-3}$	0.06	0.06
Volatilisation (days time evaporation rate)	k_v	days	0.5	0.5
C:N ratio	r_o		20:1	20:1
Critical N content for growth	N_{crit}	$kg\ N\ ha^{-1}$	10	10
Decomposition of litter	k_{lit}	day^{-1}	0.008	0.008
Decomposition of humus	k_{hum}	day^{-1}	$7x10^{-5}$	$7x10^{-5}$
Crop parameters				
Crop factor	k_c		0	1
Drought tolerance	τ		0	0.8
Light utilisation efficiency	ε	$g\ DM\ MJ^{-1}$	0	2.8
Senescence rate of roots	γ_R	day^{-1}	0	0.0055
Senescence rate of stems	γ_W	day^{-1}	0	0.003
Senescence rate of leaves	γ_F	day^{-1}	0	0.0013
Maximum leaf N content	N_f	$mg\ N\ g^{-1}$	0	3.0
Allocation to leaves	A_l		0	0.15
Allocation to stems	A_s		0	0.68
Allocation to roots	A_r		0	0.17
Specific leaf area	σ_f	ha-leaf $kg\text{-}DM^{-1}$	0	$5.4e^{-4}$

[1]See text and Appendix B.1 and Appendix B.2 for further details

variation in these properties, presumably due to differences in repacking and subsequent tree and root growth. The maximum water content was defined as the maximum water content measured via the TDR in the experiment. Field capacity was taken as the water content above which drainage occurred, as measured by the TDR and the drainage

record. For bare soil, θ_s and θ_f were set at 0.45 and 0.40 m^3 m^{-3} respectively, being similar in comparison to other studies of the Manawatu fine sandy loam (Clothier et al., 1977). For *E. nitens* values of θ_s and θ_f were set at 0.36 and 0.31 m^3 m^{-3} respectively (Table 5.1). The difference in these values between the bare soil and the *E. nitens* treatments was assumed to be related to the extent of the gravel layer packed into the lysimeters. For both simulations the wilting point (θ_w) was set at 0.08 m^3 m^{-3} and this was based on the lowest water content ever measured by the TDR in the field experiment. The K_s of the soil was determined to be 750 mm day^{-1} by disk permeameter measurements at the end of the experiment (Roygard and Vogeler, 1999). The β constant defines how quickly the hydraulic conductivity declines as the soil dries (Appendix A, Equation 6). This was taken to be 0.0293 from the soil water retention curve for the Manawatu fine sandy loam (Green and Clothier, 1999). Soil bulk density is set at 1.26 Mg m^{-3} and was based on field measurements.

The important nitrogen transport parameters are the adsorption distribution coefficients, for nitrate (k_{DN}) and ammonium, (k_{DA}). Nitrate moves without retardation through this soil type, thus k_{DN} is zero. Ammonium, which is strongly adsorbed on this soil, has a k_{DA} of 5.0 L kg^{-1} (Clothier et al., 1988). The nitrification rate (k_2) was kept as 0.2 day^{-1} (Johnsson et al., 1987). The denitrification rate (k_3) of 0.006 day^{-1} follows Ling and El-Kadi (1998). These rates were modified by soil water content and soil temperature using the same functions and parameter values of Johnsson et al. (1987). Denitrification was inhibited at a water content, δ_D, below θ_f. Based on the work of Ruz-Jurez et al. (1994), δ_D was estimated as 0.06 m^3 m^{-3}. Ruz-Jurez et al. (1994) showed a rapid lowering of denitrification rate below field capacity for the Manawatu fine sandy loam.

The carbon to nitrogen ratio (r_o) of the soil biomass and the rates of decomposition of soil humus and litter pools, affect mineralisation of nitrogen. The initial value of r_o was set equal to 20 by the following method. The organic matter fraction of the soil is 6 %, of which 50 % was assumed to be carbon. The total organic nitrogen content of the soil was measured to be 0.15 % (Figure 4.3) resulting in a carbon to nitrogen ratio of 20:1. Decomposition rate of soil litter (k_{lit}) was set to 0.0008 day^{-1} being typical of resistant plant material (Dendooven, 1990, as cited in Vanclooster et al., 1994). The rate for resistant plant material was chosen because of the waxy nature of Eucalypt leaves. The decomposition rate for the humus fraction (k_{hum}) was set at 7×10^{-5} day^{-1}, as suggested by Johnsson et al. (1987)

At an effluent-irrigated site in Australia, ammonia losses through volatilization were approximately the effluent nitrogen concentration applied times the evaporation rate on the day of application (Smith et al., 1996). That research also showed increasing temperature enhanced the volatilization rate. Here, the effect of lower temperatures at the experimental site was taken into account in parameter selection. Volatilization is calculated as half the evaporation rate on the day of effluent application (k_v=0.5) times the concentration of nitrogen in the effluent applied.

Soil profile depth (z_R) was set equal to 1 m for both the bare-soil and the *E. nitens* scenarios. To simulate the bare-soil scenario, plant factors were set to zero. Bare-soil evaporation was calculated using Equation 3.12, with the maximum bare-soil evaporation, (E_s) set equal to 2.5 mm day^{-1} for the Manawatu fine sandy loam. Parameters for the growth, water use, and nitrogen uptake by *E. nitens* were defined as follows. Crop coefficients (k_c) were established from the field experiment, simulating

the growth of the large trees in relatively small lysimeters. The k_c values determined were artificially higher than would be expected for a field-grown crop. The lysimeter measurements of crop factor were direct measurements of tree water use in relation to potential evaporation. The use of a drought tolerance factor (τ) by the model meant that these direct measurements would not remove the correct amount water from the soil. Thus in the initial run of the model assumed the drought tolerance factor to be included in the derived crop factors, and set the drought tolerance factor to 1. However, the model predicted the trees to dry the soil to wilting point and then the simulations stopped, as there was no water for the trees to use. Effectively the model predicted the trees to die. To maintain the trees 'alive' in the model, a drought factor of 0.8 was introduced and the crop coefficients were adjusted to provide water use similar to that which was measured.

Nitrogen uptake was determined from biomass production and the nitrogen contents of the plant tissues. The growth of plants in the model was determined through the light intercepted by the plant, and the efficiency of conversion of this light into plant tissue (Appendix A, Equation 11). The light-utilisation-efficiency parameter (ε) is used to reproduce the correct growth rates observed in the lysimeter experiment. The trees storage of nitrogen is not only defined by growth but also relates to the allocation of dry matter to leaves, stems and roots as well as the senescence of each of these categories (Appendix A, Equations 13, 14 and 15). Allocation of dry matter to roots, stems, and leaves as well as the specific leaf area were defined on the basis of the final harvest for *E. nitens*. The senescence rates, fractions of nitrogen recycled before senescence were as used by King (1993). For *E. nitens*, the maximum nitrogen content of the leaves (N_f)

utilised here is 3.0 mg N g^{-1} dry matter and corresponds to the highest concentration measured in the lysimeter study.

The initial conditions in the model were the same as measured in the lysimeter experiment. Initial water contents were established from the mean TDR measurements for the treatments on January 1, 1996. Initial values for the tree dry matter were determined from biomass estimates for December 1995. These assumed that 30 % of the trees dry matter was produced in the establishment phase of the experiment. The initial solute concentrations in the soil were set equal to the leachate concentrations collected in the last drainage events of December 1995.

5.4.3 Simulation results and discussion

The model shows a broad agreement with the water and nitrogen balances of the bare soil and the *E. nitens* for the 1996 period (Figure 5.4, Figure 5.5, Table 5.2). The water balances for these two scenarios show particularly good agreement. This was expected because the crop factors for the *E. nitens* were determined directly from the lysimeter experiment. Some of the bare soil data were also obtained via the same evaporation calculation as used in the lysimeter experiment because of the missing drainage data (Section 3.2.6). This explains a good agreement for the water content data. The drainage from the measurements, and the LPM model, do however provide a test of two mass balance approaches, i.e. the tipping bucket solution used in the field experiment and the analytical solution of Sisson et al. (1980). These two methods show good agreement in drainage volume estimates. For the *E. nitens*, the drainage model (Sisson et al., 1980) predicts drainage to occur somewhat later than it was measured in the field experiment. Nevertheless, on an annual basis the prediction is very close.

Table 5.2 Water and nitrogen balances for 1996 as measured in the experiment and predicted by the LPM model.

	Bare soil		*E. nitens*	
	Experiment	Model	Experiment	Model
Water (mm)				
Rainfall	1087	1087	983	983
Irrigation	817	817	817	817
ET	524	544	1867	1849
Drainage	1339	1299	106	99
Nitrogen (kg N $ha^{-1}yr^{-1}$)				
Effluent inputs	621	621	621	621
Mineralisation	Unknown	12	Unknown	14
Uptake	0	0	273[1]	270
Drainage	488	364	35	44
Volatilization	Unknown	31	Unknown	7
Denitrification	Unknown	32	Unknown	7

[1] Annual average for 2.5 years of tree growth.

Nitrogen concentrations of the leachate from the bare soil show very similar patterns for the measurements and the model (Figure 5.4). Predictions are well within the scatter of measured leachate concentrations, and they show a similar rise in concentration latter in the year. On average, the model predicts lower concentrations in the bare soil leachate over the winter period. The large flux of water through the bare soil treatment provides a multiplier effect on the slight under estimation of concentration in the root zone. This is reflected in the lower annual total amount of leaching predicted by the model. The increase in soil nitrogen concentration in the spring-summer period is due to the higher nitrogen concentrations in the effluent being applied (Figure 4.2). This increase in soil nitrogen concentration is predicted accurately by the model (Figure 5.4). Overall the model seems reasonable at simulating leaching of nitrogen from the bare soil treatment.

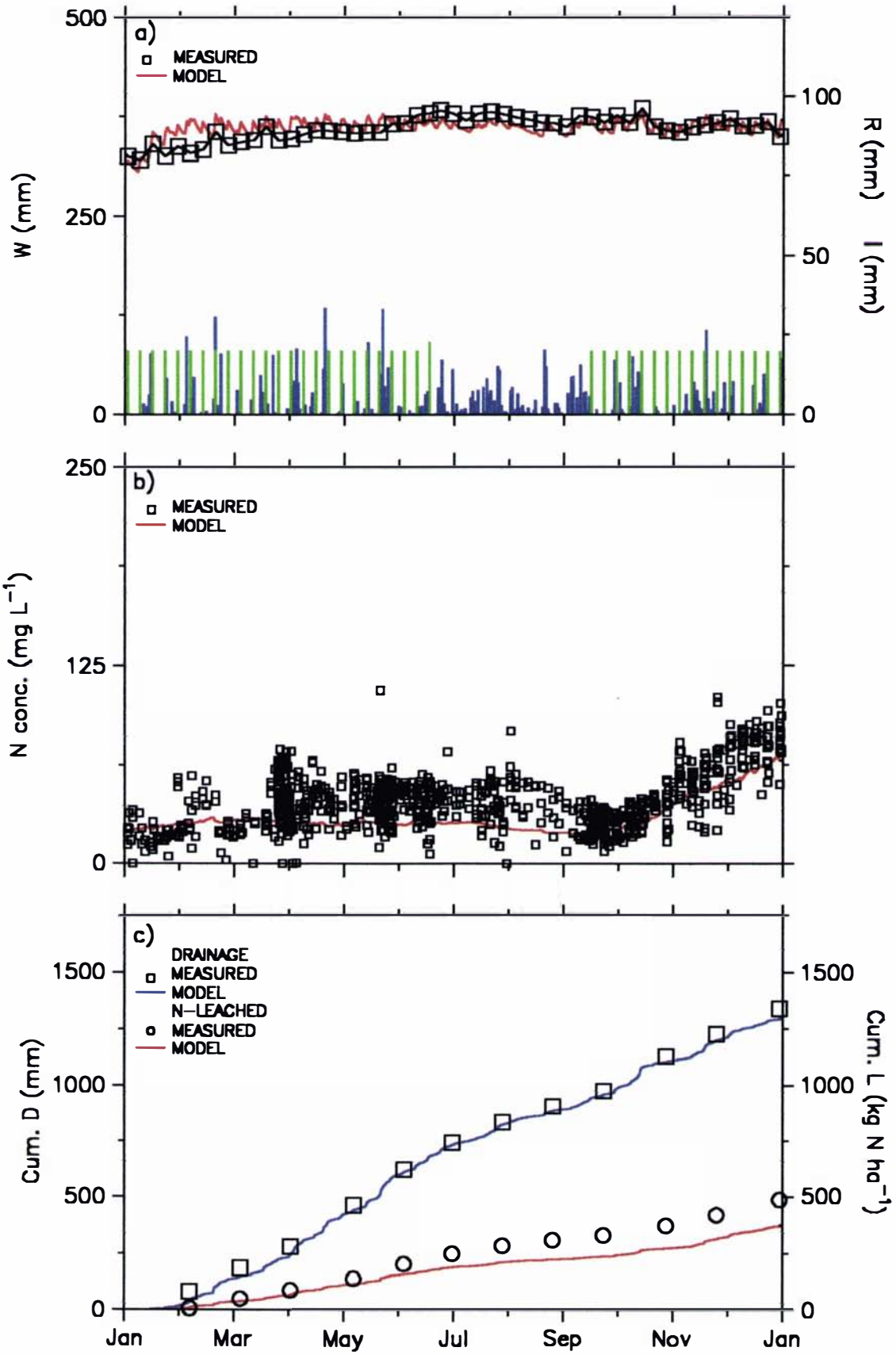

Figure 5.4 Bare soil measured and modelled data a) water content of the soil profile, W, also showing the rainfall (R) and irrigation events (I). b) Nitrate concentrations in the leachate. c) Cumulative drainage, (Cum. D) and cumulative nitrogen leached (Cum. L).

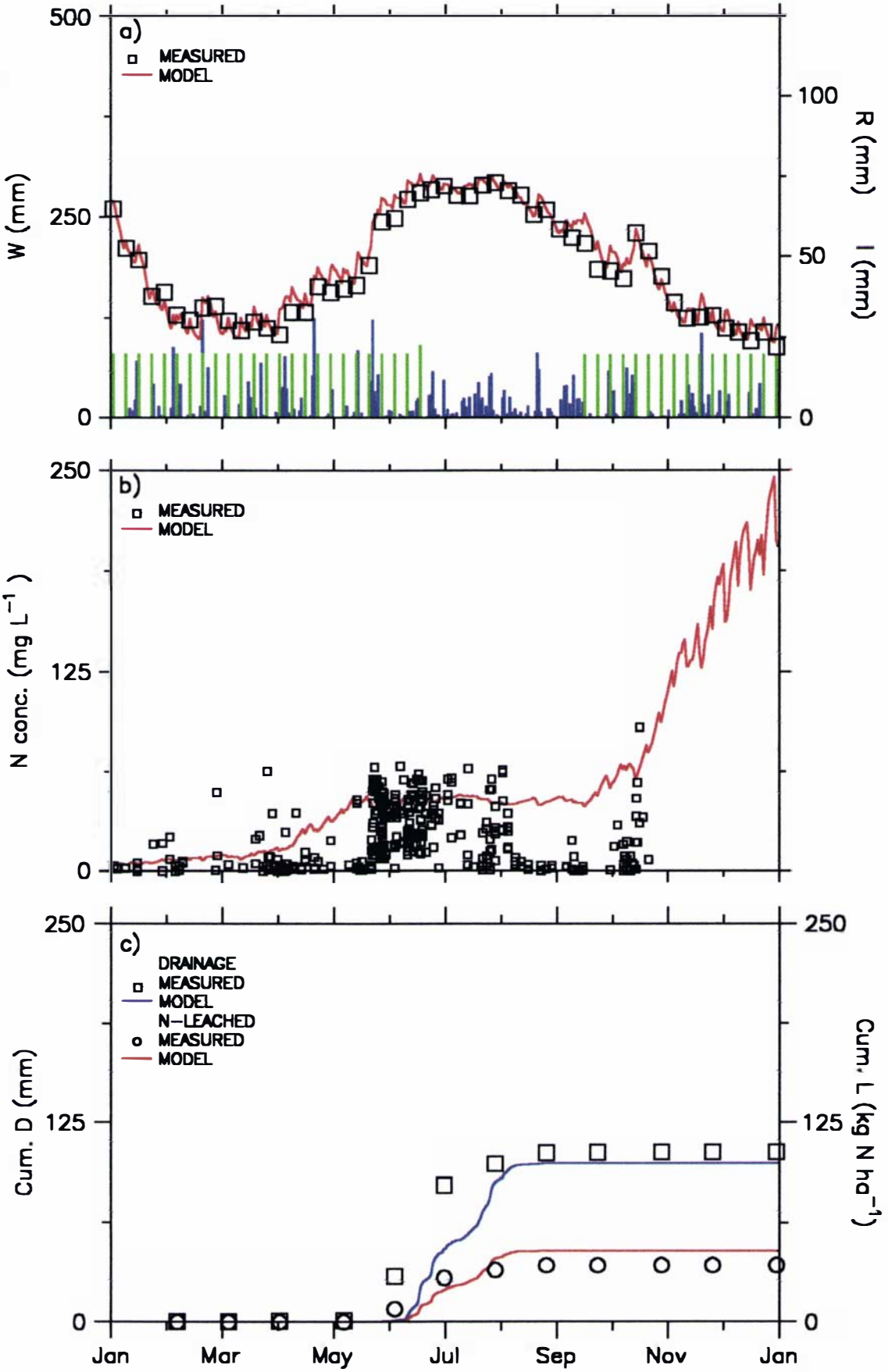

Figure 5.5 *E. nitens* measured and modelled data a) water content of the soil profile, W, also showing the rainfall (R) and irrigation events (I). b) Nitrate concentrations in the leachate. c) Cumulative drainage, (Cum. D) and cumulative nitrogen leached (Cum. L).

Annual nitrogen leaching predicted by the model for *E. nitens* is 44 kg N ha^{-1} yr^{-1} which is close to the measured 35 kg N ha^{-1} yr^{-1}. Given the simplicity of the model, this is seen as a robust prediction. The model predicted soil-nitrogen content of the *E. nitens* also to increase in response to higher effluent-nitrogen concentrations applied in the spring-summer period (Figure 5.5). For *E. nitens* the increase was predicted to be greater than that of the bare soil. This reflects the difference in water status of the two treatments. The bare soil was leaching nitrogen and had a higher water content. Thus the bare soil retained less of the applied nitrogen over the winter than the *E. nitens*. At the same time, whilst the *E. nitens* was not leaching, the trees were experiencing mild water stress and thus the added nitrogen remained in the soil profile, resulting in an increase in nitrogen concentration in soil solution.

Model inputs were chosen to generate the correct growth rate for the tree biomass via the light-use efficiency factor. Tree growth during the year produced the equivalent of 41.8 Mg ha^{-1} biomass. This was slightly higher than the *E. nitens* biomass production in the lysimeter experiment, which averaged 38.6 Mg ha^{-1} yr^{-1} based on a planting density of 4000 stems ha^{-1}. The growth parameters also predicted a nitrogen uptake rate of 270 kg N ha^{-1} yr^{-1} (Table 5.2), being 3 kg less than the uptake rate measured in the lysimeter experiment (273 kg N ha^{-1} yr^{-1}). Without precise information on the allocation of biomass growth and distribution of tissue nitrogen over the 2.5 years of the experiment, the average values are seen as appropriate.

Confidence in the model predictions has been gained from the simulation procedures presented above. It is important to note that with some parameters being deduced from the field experiment, the model results are not totally independent. However the level of

agreement between the model and the measurements is heartening. The model is now used in the next phase of the study, application to a plantation scale scenario of SRF for land treatment of dairy-shed effluent.

5.5 Modelling at the plantation scale

5.5.1 Reference scenario description and parameterisation

The performance of a SRF land treatment system was simulated over a 27 year period using climate data recorded at the MAF station at Levin (1972-1998). Assuming similar average climatic data in the future, the use of this climatic data enables the model to predict the long term fate of dairy-shed effluent applied to an SRF system. The input climate data could easily be modified to incorporate prescribed scenarios of climatic change and investigate the effect of these on sustainability of SRF land treatment systems.

The use of this effluent application of dairy-shed effluent was limited to 200 kg ha^{-1} yr^{-1} in accordance with the Regional Plan of horizons.mw (formerly Manawatu Wanganui Regional Council) (Forsyth, 1996). This scenario serves to define a baseline against which other scenarios will be compared. The parameter file for this scenario is provided in Appendix B.3

For the reference scenario, effluent was applied at weekly intervals over 'summer'. Here 'summer' is defined as the 185 days between the 300th day of year (about 25th October) and the120th day of year (the end of April). This time period avoids application during the wetter months of the year. Nitrogen loadings are calculated based on average nitrogen concentrations reported by Hickey et al. (1989). Their study surveyed characteristics of 11 dairy-farm effluent ponds monthly in Manawatu and Southland.

The average nitrogen concentrations were of 81.7 mg NH_4^+-N L^{-1} and 0.2 mg NO_3^--N L^{-1}. These values agreed well with the values measured for the effluent in the lysimeter study. For the model a loading of 200 kg N ha^{-1} $year^{-1}$ was achieved, using a weekly application of 9.5 mm during the 185 days of 'summer'.

In the lysimeter experiment, tree water-use was calculated on the basis of a plantation density of 4000 stems ha^{-1} reflecting the lysimeter surface area. Estimates of maximum tree water-use were only slightly higher than those in other studies (Section 3.3.4). However the calculated crop coefficients were higher than would be observed in a field environment because the experiment had big trees growing in lysimeters of small surface area. For this reason, literature values of crop coefficients (k_c) were deemed more appropriate. Doorenbos and Pruitt (1977) suggest closed canopy forest k_c values range from 0.9 to 1.1. Conservative values of 1.0 are used here for the reference scenario. The drought tolerance factor, τ=0.5 also reflects a more realistic value (Doorenbos and Pruitt, 1977).

Although the biomass production rates of the lysimeter experiment were in the range estimated by other studies, it was seen as more appropriate to simulate production at a level similar to that measured in plantation scale studies. Thus, to achieve rates of production measured in plantations of *E. nitens* in New Zealand, a light utilisation efficiency, ε, of 1.7, was used.

The practice of tree harvesting was then introduced for these scenarios. The model can harvest or thin whenever the above-ground biomass reaches a certain level. This level was set equal to 70 Mg ha^{-1} to maintain a short (<4 year) rotation period. In theory short

rotations maintain a high nitrogen demand as the plants are continually regenerating the canopy leaves which have the highest amount of nitrogen (Miller, 1984). The model requires a specification of the amount of leaf remaining after a harvest. Here, a value of 1 Mg ha^{-1} was used in order to maintain some leaf material from which the model could regenerate growth of the trees. Harvesting parameters also include the amount of stem remaining following harvest. In this simulation, 90 % of stems were removed leaving some stem to balance with the proportion of leaf remaining after harvest. All harvested biomass was removed from the system, although the model does allow some harvested biomass to be returned as litter to the system.

All other factors were maintained as for the simulation of the *E. nitens* treatment in the lysimeter experiment.

5.5.2 Scenario results and discussion

Annual water and nitrogen budgets simulated for the 27 years are presented in (Table 5.3, Table 5.4, Figure 5.6, Figure 5.7, Figure 5.8). Table 5.3 and Table 5.4 also show means, standard deviation and ranges of the predicted balances. These statistics require consideration in terms of sustainability. The means provide a summary of the data but extreme values may limit the system, as they highlight the potential 'worst' and 'best' case of annual values.

The trees were harvested 7 times during this scenario, and were nearing an eighth harvest in the last year of the simulation. The first harvest of biomass occurred after 2.5 years, following this crop rotations were consistently 3-4 years long. The average biomass production in the simulation was 18.5 Mg ha^{-1} yr^{-1} (Figure 5.7). This production rate fits well with rates of 15 and 20 Mg ha^{-1} yr^{-1}, previously recorded for 4

and 5 year rotations of unfertilised *E. nitens* in New Zealand (Frederick et al., 1984; Madgwick et al., 1981). The production rate however is conservative in comparison to estimates of 32 and 40 Mg ha^{-1} yr^{-1} for 7 year rotations of *E. nitens* at stocking rates of 5,861 and 17,414 stems ha^{-1} respectively (Kincheff and Carter, 1991). The growth rates of the model are seen as appropriate, given the short rotations and extra water and nutrient inputs.

Table 5.3 Model prediction of the water balance of SRF system treating dairy shed effluent (200 kg N ha^{-1} yr^{-1}) for 27 years. Bold numbers indicate maximum, minimum, means or standard deviation for the column. All units are mm yr^{-1}.

Year	Rainfall	Irrigation	ET	Drainage	Run off
1972	898.8	256.5	979.4	227.9	0
1973	777.0	266.0	909.6	175.6	0
1974	666.7	256.5	833.9	95.2	0
1975	853.7	247.0	856.9	255.0	0
1976	862.4	247.0	868.4	208.8	0
1977	**1029.6**	247.0	915.3	342.0	0
1978	865.9	247.0	925.5	183.1	0
1979	708.4	256.5	851.2	110.9	0
1980	846.9	266.0	886.7	175.0	0
1981	806.1	256.5	958.4	129.3	0
1982	747.4	247.0	907.3	156.4	0
1983	759.8	247.0	895.3	21.9	0
1984	664.6	247.0	904.2	50.6	0
1985	722.2	247.0	941.9	62.6	0
1986	742.1	256.5	899.8	**2.8**	0
1987	818.3	266.0	**1039.8**	144.1	0
1988	752.2	256.5	882.7	101.9	0
1989	915.4	247.0	916.0	264.0	0
1990	694.7	247.0	870.8	68.3	0
1991	923.2	247.0	925.0	257.8	0
1992	902.6	247.0	892.9	157.3	0
1993	965.2	256.5	**711.8**	**450.2**	0
1994	780.7	266.0	771.2	309.9	0
1995	829.4	256.5	819.6	315.3	0
1996	972.0	247.0	852.3	339.9	0
1997	920.2	247.0	846.7	315.5	0
1998	**596.5**	247.0	818.0	54.6	0
Mean	**815.6**	**252.6**	**884.5**	**184.3**	**0**
St. Dev.	**106.5**	**7.1**	**64.4**	**114.0**	**0**

Table 5.4 Model prediction of the nitrogen balance of SRF system treating dairy shed effluent (200 kg N ha^{-1} yr^{-1}) for 27 years. Bold numbers indicate maximum, minimum, means or standard deviation for the column. Concentrations are expressed as mg N L^{-1}, all other units kg N ha^{-1} yr^{-1}.

	N inputs	Mineral-isation	Uptake	Leachate quantity	Denit-rification	Volat-ilisation	Leachate conc. NH_4^+	Leachate conc. NO_3^-
1972	208.3	**19.8**	136.8	54.8	2.2	9.9	0.0	**24.8**
1973	216.0	20.3	139.8	52.6	1.8	**4.9**	0.1	30.3
1974	208.3	26.0	56.3	46.6	1.7	11.8	0.0	48.6
1975	200.6	21.7	146.4	145.4	4.7	20.0	0.0	52.2
1976	200.6	22.8	164.5	81.7	2.6	10.0	0.0	40.1
1977	200.6	24.0	59.7	105.5	2.6	11.6	0.0	28.6
1978	200.6	21.6	96.3	89.7	3.1	21.3	0.0	47.8
1979	208.3	24.1	**169.5**	56.0	1.8	12.9	0.0	49.7
1980	216.0	29.0	136.3	87.4	2.7	7.4	0.0	50.3
1981	208.3	27.5	**37.3**	86.8	2.7	22.8	0.0	66.3
1982	200.6	28.8	144.7	105.5	3.3	17.4	0.0	67.8
1983	200.6	27.9	151.8	16.4	0.3	8.5	0.0	73.6
1984	200.6	31.4	56.9	46.6	1.4	10.8	0.0	90.2
1985	200.6	30.9	110.3	72.4	2.1	19.7	0.0	116.2
1986	208.3	32.2	168.4	**3.9**	**0.0**	13.5	0.0	**144.7**
1987	216.0	37.1	117.8	172.0	5.2	10.4	0.0	119.1
1988	208.3	35.7	60.6	125.0	4.6	**23.6**	0.1	122.2
1989	200.6	36.3	146.6	**256.0**	8.5	16.1	0.0	92.0
1990	200.6	39.6	148.1	53.9	1.4	7.8	0.0	79.0
1991	200.6	42.5	59.1	183.4	7.7	15.5	0.0	71.3
1992	200.6	35.3	119.0	113.2	3.4	18.3	0.0	70.8
1993	208.3	35.2	98.6	214.0	**9.2**	8.5	0.1	46.2
1994	216.0	38.2	119.1	119.6	4.4	5.4	0.1	38.4
1995	208.3	**43.6**	40.2	154.2	6.0	17.8	0.1	51.8
1996	200.6	42.7	108.4	156.7	6.3	17.4	0.1	44.1
1997	200.6	42.0	134.9	117.5	4.9	8.9	0.1	36.4
1998	200.6	44.0	120.1	28.8	0.6	6.0	0.1	50.9
Mean	**205.1**	**31.9**	**112.9**	**101.7**	**3.5**	**13.3**	**0.1**	**61.8**
St. Dev.	**5.8**	**7.8**	**41.0**	**60.8**	**2.4**	**5.5**	**.05**	**27.5**

The water balance is dominated by the rainfall inputs and the ET outputs (Figure 5.6, Table 5.3). Interestingly the highest rainfall year (1977) coincides with the highest annual ET because extra summer rain fell on the site in times when water soil would normally be restricting ET.

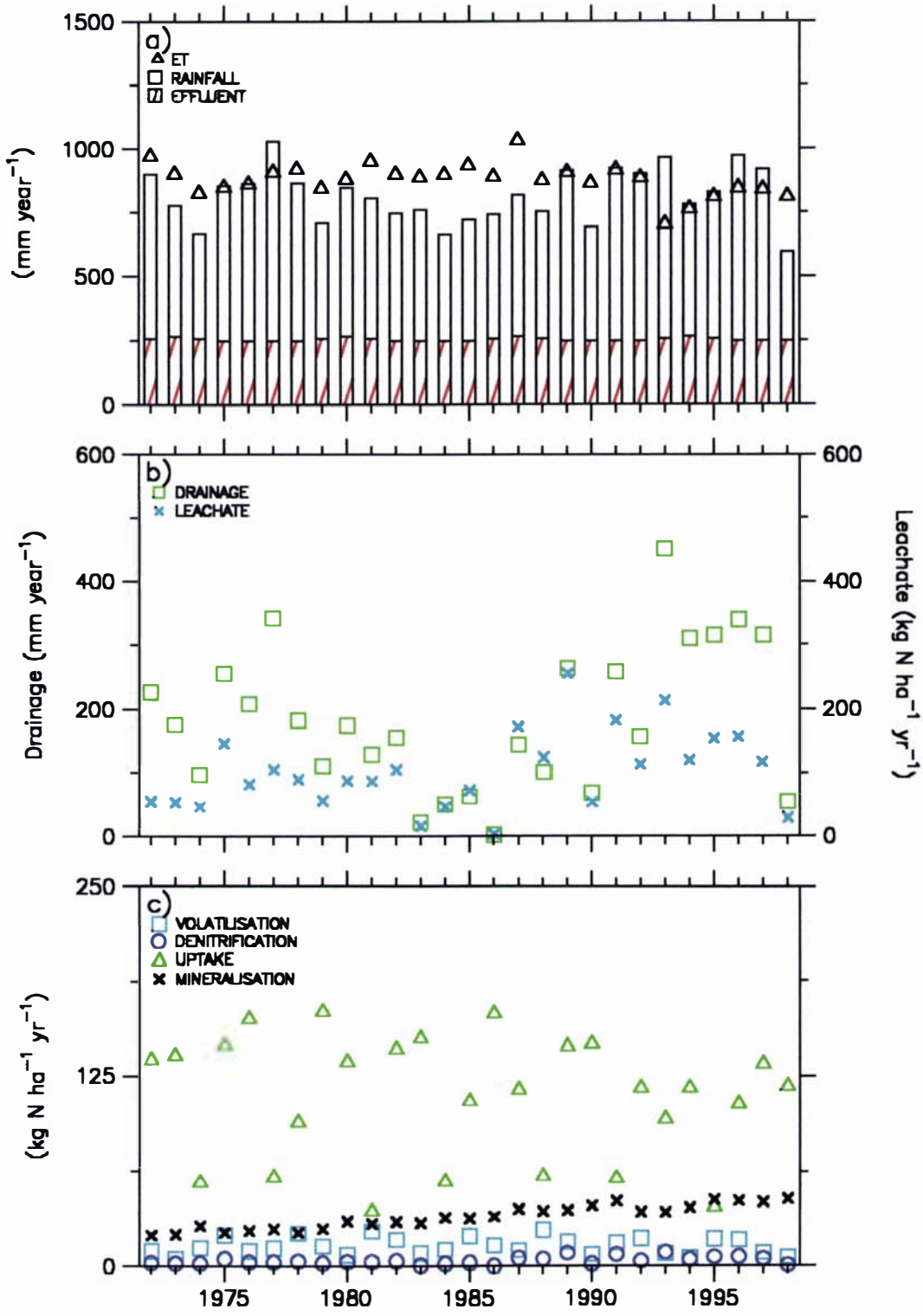

Figure 5.6 Annual water and nitrogen balances of a simulated SRF treatment system receiving dairy shed effluent application of 200 kg N ha^{-1} year^{-1} over 27 years a) Annual water inputs of rainfall and effluent irrigation and water outputs through ET. b) Drainage volume and quantity of nitrogen in the leachate. c) Nitrogen inputs from mineralisation and outputs through plant uptake, volatilization and mineralisation.

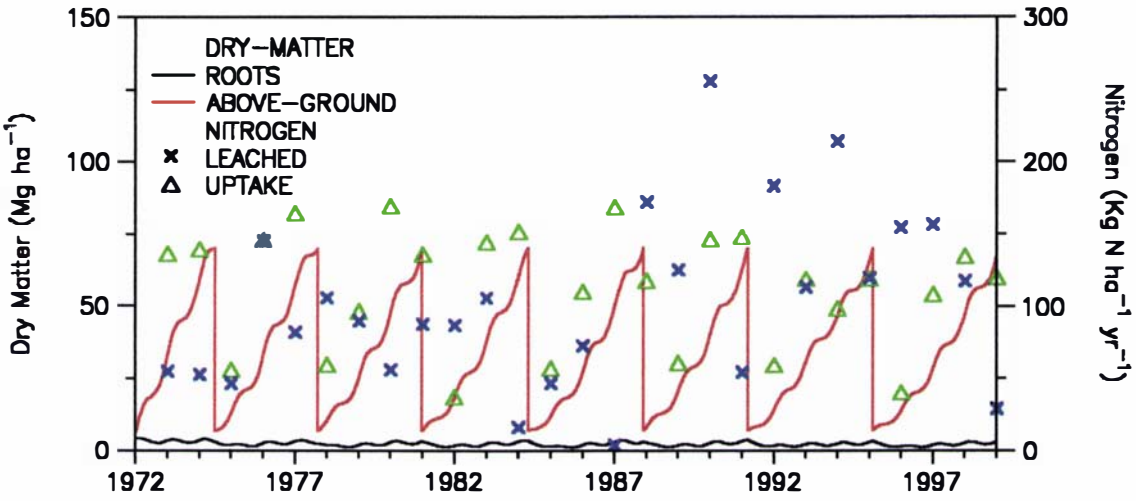

Figure 5.7 Dry matter production of a simulated SRF crop irrigated dairy-shed effluent. Weekly irrigation in 'summer' applies a total of 200 kg N ha^{-1} yr^{-1}.

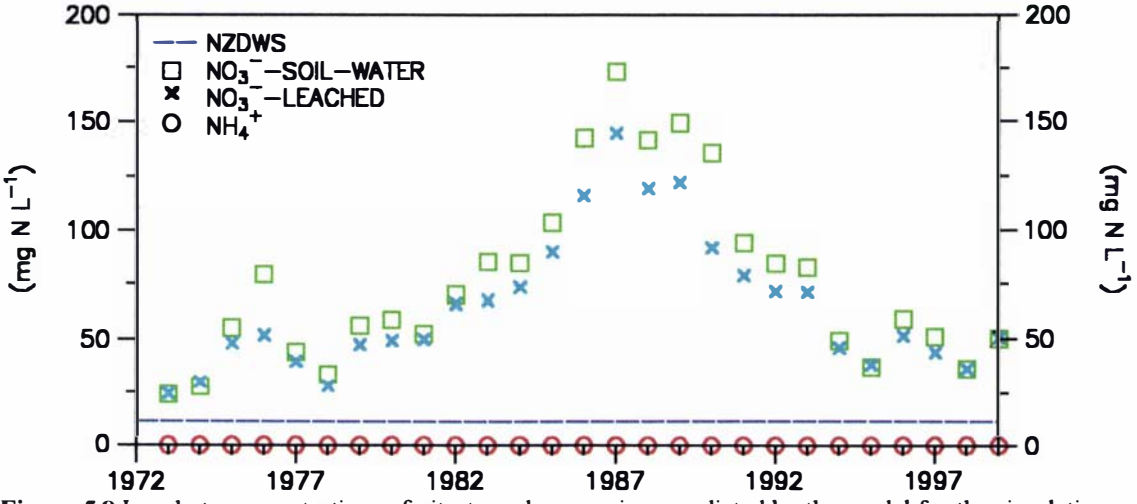

Figure 5.8 Leachate concentrations of nitrate and ammonium predicted by the model for the simulation of SRF dairy-shed effluent treatment. Weekly irrigation in 'summer' applies a total of 200 kg N ha^{-1} yr^{-1}.

The choice of literature values for crop coefficients has resulted in realistic rates of water use. Mean annual ET for the trees for this scenario, was 884 mm $year^{-1}$. This compares well with other measurements of tree water use by Eucalypts receiving effluent application in Australia. Plantation water use for the first three years of growth in the Wagga Wagga Effluent Plantation Project averaged 1005 mm (Myers et al., 1996). Prediction by the model in the Manawatu climate was lower than the Australian measurements, presumably because of the difference in climatic conditions.

Water balance predictions from the model indicate drainage to be highly variable from year to year (Table 5.3, Figure 5.6). Drainage ranged from 2.8 mm to 450 mm and exceeded 350 mm annually only once in 27 years. As expected, the annual hydraulic loading has a strong influence on the annual drainage volume. However the year of least drainage (1986) is not the year of least water inputs (1998). Thus, it can be that the timing of events of rainfall, rather than the annual loadings that have the greatest bearing on the annual drainage volume and leaching losses. In addition the tree biomass status can also influence the hydraulic balance.

While surface runoff is a component of the water balance calculation, at no stage does the model predict runoff occurring for this scenario. Irrigating at just 9.5 mm $week^{-1}$ in 'summer' only, onto the free-draining Manawatu soil means there is little possibility of surface runoff occurring.

The reference scenario yielded effluent inputs between 200 and 216 kg N ha^{-1} yr^{-1} into the system. Additional nitrogen inputs from mineralisation were comparatively low, ranging between 20 and 40 kg N ha^{-1} yr^{-1}. Mineralisation inputs slowly increased over the 27 years, as senescence plant material is added to the soil humus and litter stores. Mineralisation can increase markedly if a large fraction of harvested DM is returned to the soil as litter. However, here it was assumed that all of the harvested biomass is removed from the site.

Outputs of nitrogen through denitrification and volatilization are consistently low in comparison to uptake and drainage losses (Table 5.4, Figure 5.6). Denitrification only occurs when the soil has anaerobic conditions, thus only occurs when the soil moisture

content is high. Denitrification was low in this simulation because of the free draining nature of the soil. Water use by the trees contributed to low soil water contents, which were for most of the year below the threshold value above which denitrification occurs. Volatilization throughout the 25 years was always less than 24 kg N ha^{-1} yr^{-1}. This may be an artifact of the simple model for volatilization, which was computed as half the soil evaporation on the day of application, multiplied by the concentration of the effluent applied. This calculation obviously maintains volatilization low during the experiment. Evaporation of water from the soil makes up only a small fraction of the total ET of the system, when the trees are in full leaf. This further reduces volatilization losses from the soil surface. Thus volatilization losses are somewhat higher following harvest, because soil evaporation makes a greater contribution to total ET.

The model predicts plant uptake and leaching as the dominant losses of nitrogen from the system. The rate of nitrogen uptake clearly increases as the tree grows (Figure 5.7). The lower values of uptake occur following coppicing. Nitrogen leaching does not show such a clear relationship with tree growth.

Drainage volume and the leaching of nitrogen are linked intimately by the concentration of nitrogen in the soil profile. The annual average concentrations of nitrogen in the soil solution and the leachate are shown in Figure 5.8. In the model, when drainage beyond the rootzone occurs, the concentration of nitrogen in the leachate is that of the soil profile. The mean-annual soil nitrogen concentration is the average for every day of the year. Whereas, the mean-annual leachate nitrogen concentration is the average of those days when drainage was predicted to occur. Predictions of the annual-average nitrogen concentrations of the leachate are all in excess of the New Zealand Drinking Water

Standard (NZDWS) of 11.3 mg NO_3^--N L^{-1} (Figure 5.8). The highest annual-average nitrate concentration in the soil profile is 174 mg NO_3^--N L^{-1}. The highest annual leachate nitrate concentration of 144.7 mg NO_3^--N L^{-1} occurred in 1986, however less than 5 mm leached during that year. Further analysis of mean-annual nitrate concentration in the leachate will also ignore this year as the low hydraulic loading negates the effect of the high concentration. The next highest mean-annual leachate nitrate concentration was 122.2 mg N L^{-1} and predicted in 1988, which had over 100 mm of drainage. The most significant year of nitrate loss to ground water was 1989 when 218 mm of drainage occurred with an average concentration of 92 mg NO_3^- L^{-1}. This resulted in a leaching loss of 256 kg N ha^{-1} yr^{-1}. The highest concentrations of nitrate predicted for the leachate are similar to the highest monthly-average nitrate concentration observed in the leachate for *E. nitens* in the lysimeter experiment (150 mg NO_3^--N L^{-1}) (Figure 4.10). The model also predicted negligible ammonium in the soil water, in agreement with the measurements from the lysimeter study. This lends confidence to the range of values predicted for nitrogen concentration in the leachate.

For the entire 27 year simulation the mean-annual concentration of NO_3^--N in the soil profile and leachate were 76 and 62 mg NO_3^--N L^{-1} respectively. The relationship between the mean annual concentrations of nitrogen in the soil and the leachate varied between the years. For example, Figure 5.9a shows in 1991 the annual average soil nitrogen concentration is in good agreement with the average concentrations leaching. In contrast, Figure 5.9b shows in 1983 the nitrate concentrations were variable and the average soil nitrogen concentration was not a good predictor of the annual leachate concentrations. Higher water contents in the soil and greater leaching in winter dilutes the mass of nitrogen in the soil solution. The dilution from winter rains and the loss of

nitrogen through drainage events can be seen in Figure 5.9b. This suggests that measuring the nitrogen concentration of the soil in a land treatment system throughout the year may not be the best predictor of the leaching concentrations in an SRF land treatment system. A better indicator may be to measure the concentrations of the profile when the soil and the climatic conditions are favourable for leaching to occur.

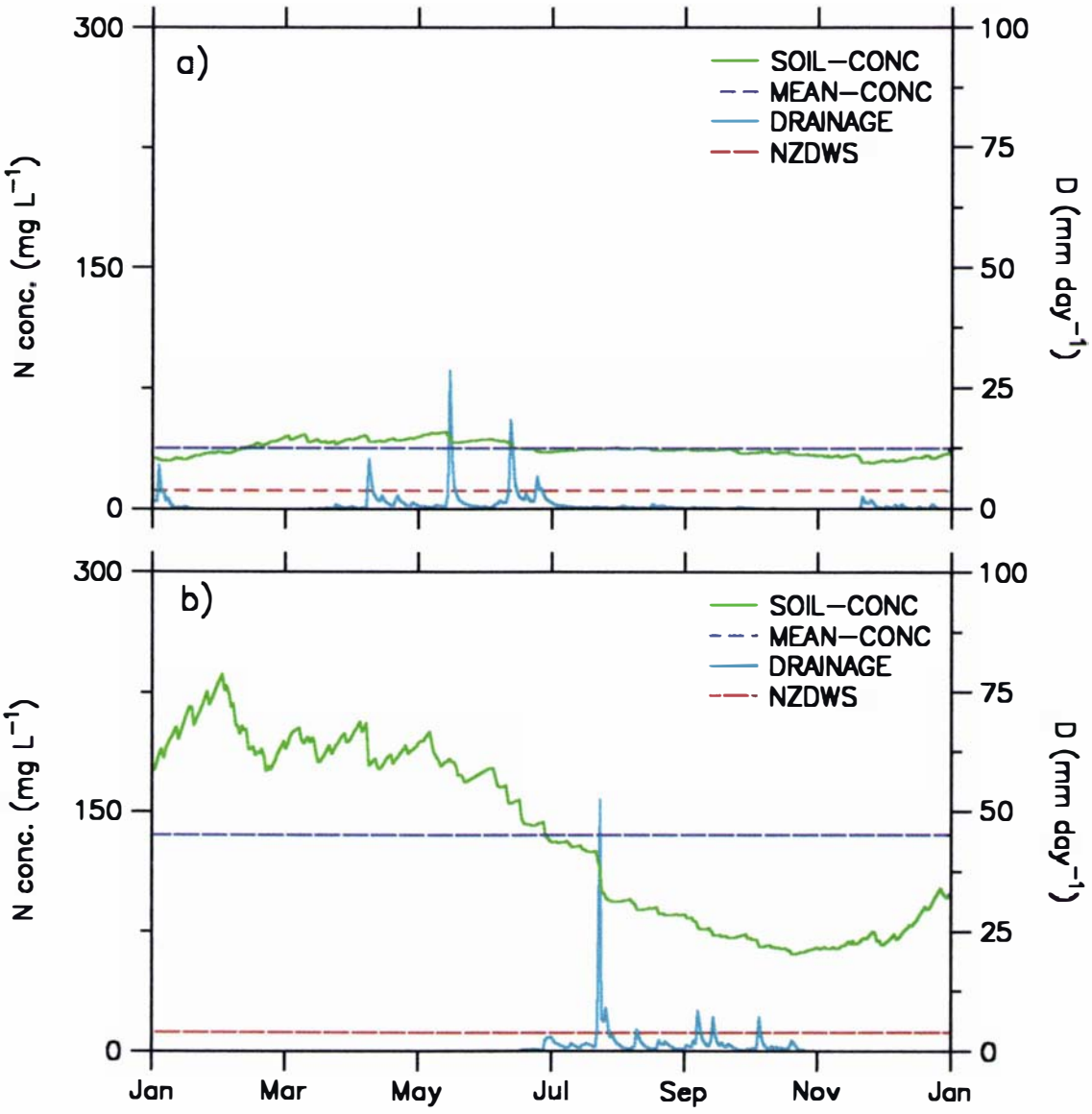

Figure 5.9 Comparison of daily soil nitrogen concentrations with the mean annual nitrogen concentrations showing drainage events a) year of small change in soil N concentration b) year of large change in soil nitrogen concentration.

If the high concentrations of nitrate predicted were to leach beyond the rootzone, then a considerable dilution in the groundwater would be required to render the system sustainable. The highest concentrations were predicted between 1985 and 1990, which followed several dry years when little drainage occurred. The result was due to an inevitable accumulation of nitrogen in the profile following regular application of high nitrogen concentrations in the effluent. The lack of further irrigation to the system in dry years reduces the chance that the applied nitrogen concentration will be diluted.

The model predicts that the main limitation to the land treatment system is likely to be the nitrate concentration in the leachate.

5.6 Sensitivity analysis

A sensitivity analysis for most parameters in the model was carried out. Sensitivity is measured as percentage change in the mean-annual nitrogen-concentration of the leachate (N_C) relative to the reference scenario. This is seen as a suitable measure as in much the same way the NZDWS is used as a deciding factor for Resource Consent decisions. Sensitivity is also measured in terms of the percentage change in the mean-annual nitrogen loading (N_L) leaching below the root zone. Nitrogen loading is seen as a good way to test the sensitivity of the model to its parameters as it is inherently linked to both the drainage volume and the concentration of nitrogen leaching to the ground water. The sensitivity analysis was carried out by varying the model parameters to half and double their reference values. However in the case where parameters can not vary to that extent, the limits to possible values were used.

5.6.1 Soil properties of water movement

Soil properties showed greater influence on the mean-annual nitrogen concentration of leachate (N_C) than on mean-annual nitrogen loading (N_L) to groundwater (Figure 5.10, Figure 5.11). The only soil parameter to have a large effect on N_L was the β constant. Lowering β lowers the hydraulic conductivity of the soil, thus drainage is reduced and the soil water content remains higher. Increased water availability increases the tree water use during dry periods. Further, if β is lowered by greater than 15 % the denitrification rates increase enough to lower leachate nitrogen concentrations. Increasing β also has the effect of lowering nitrogen concentrations, however this effect occurs because of greater drainage volume, due to increased hydraulic conductivity of the soil. Increasing β lowers denitrification losses, thus a small increase in nitrogen concentration is observed. However, increasing β by approximately 25 % reduces denitrification losses to zero, as soils are more free draining. Thus increasing β beyond this point does not increase the quantity of nitrogen leaching.

The sensitivity analysis varied both θ_s and θ_f by 10 % (Figure 5.10, Figure 5.11) Denitrification was similarly affected by this because the range of soil water contents in which denitrification could occur was altered. Increasing θ_s and θ_f increased the range of water contents in which denitrification could occur, thus increasing denitrification and decreasing the concentrations of the leachate. Similarly lowering the value of θ_m and θ_f increased the concentrations and loading of nitrogen going beyond the rootzone. It follows that the effect of decreased drainage quantity, and increased concentrations, to some extent cancel out in the overall leaching of nitrogen. Thus the effect of changing θ_s and θ_f on mean annual leaching quantity was small.

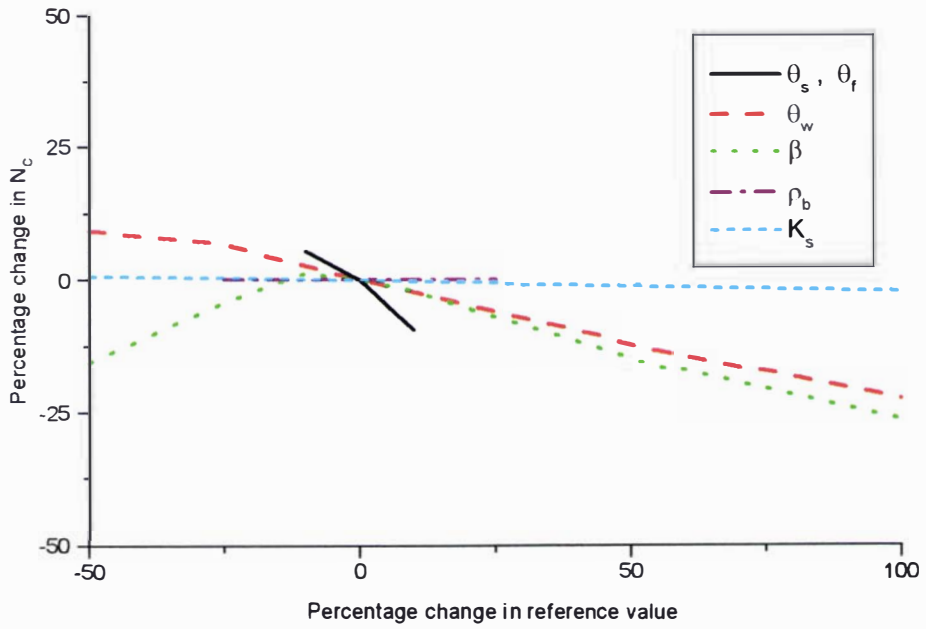

Figure 5.10 Sensitivity analysis of the change in mean-annual nitrogen concentrations in the leachate (N_C) to changes in the parameters of soil properties. Reference values are θ_s=0.44 m^3 m^{-3}, θ_f=0.39 m^3 m^{-3}, θ_w=0.08 0.44 m^3 m^{-3}, β = 0.0293, ρ_b=1.26 Mg m^{-3} and K_s= 750 mm day^{-1}.

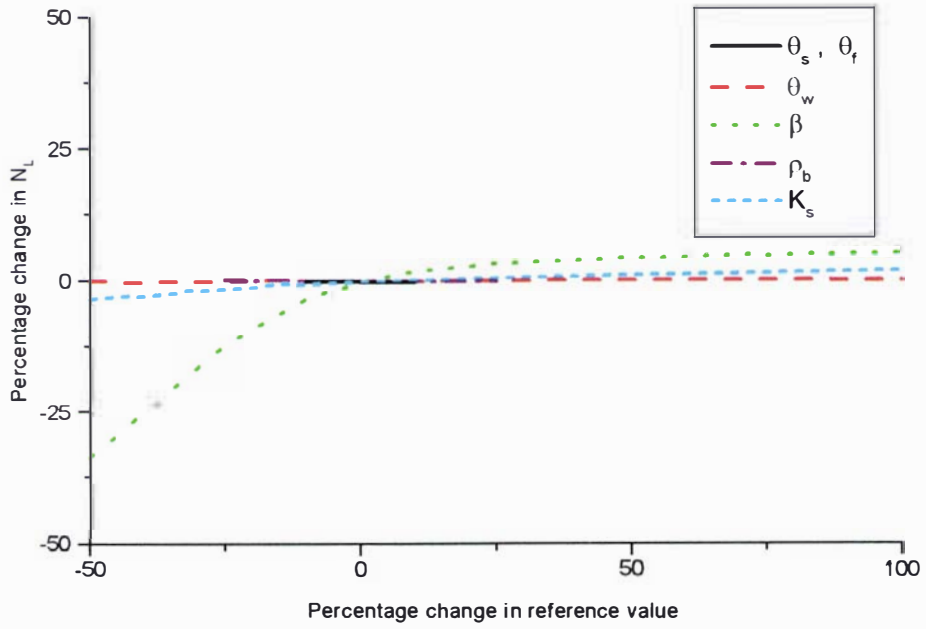

Figure 5.11 Sensitivity analysis of the change in mean-annual nitrogen loading beyond the rootzone (N_L) to changes in the parameters of soil properties. Reference values are θ_s=0.44 m^3 m^{-3}, θ_f=0.39 m^3 m^{-3}, θ_w=0.08 0.44 m^3 m^{-3}, β = 0.0293, ρ_b=1.26 Mg m^{-3} and K_s= 750 mm day^{-1}.

The difference between field capacity (θ_f) and wilting point (θ_w), is the available water content. Any reduction in the amount of available water means the trees are more likely to experience water stress, and thereby reduce their water and nitrogen uptake. Increasing the wilting pointing reduces the plant water use, thus drainage increases. With increased water in the soil profile, the nitrogen concentrations are lowered without a change in the nitrogen content. However, the effect of increased drainage combined with lower concentration has little effect on the mean annual loading to groundwater. Halving the saturated hydraulic conductivity (K_s) had the expected effect of lowering the leaching to ground water. But it was only 2 % lower and had no effect on leachate concentrations. Since the hydraulics of this system are flux-limited, increasing K_s had even less effect. In the model, bulk density has no effect on water flow.

The soil properties of the Manawatu fine sandy loam are well characterised. In the sensitivity analysis the parameter with the greatest effect was the β constant. The value used here, 0.0293, is derived by Green and Clothier (1999). The parameters of θ_s, θ_f, and θ_w were derived from the TDR measurements in the field study. Measurements of ρ_b and K_s also came from measurements in the lysimeters (Roygard and Vogeler, 1999). Thus, the soil parameters used in the simulation are appropriate. However, when applying the model to new soil types an estimation of β will be required.

5.6.2 Crop parameters relating to water movement

Of the crop parameters relating to water movement, it is the root zone depth, crop coefficient and the effective rainfall that were found to have the largest influence on the concentrations leaching to ground water (Figure 5.12, Figure 5.13). The proportion of rainfall reaching the soil surface (R_e) has a large influence on the annual nitrogen

leaching. The above parameters each influence the amount of water moving into or out of the soil profile. Increasing the amount of water in the profile, by decreasing plant water use or by increasing inputs of rainfall and soil storage capacity, leads to a decrease in the concentrations in the profile through dilution, and greater denitrification. However, increasing water inputs also increases the drainage volume. The combined effect of decreased concentration and increased volume leaching minimises the effect on the total quantity of nitrogen leaching. This indicates that additional water inputs to the system will increase the quality of leachate thus extra irrigation of water may be a management option considered to lower nitrogen concentrations in the leachate.

Several crop parameters that have a direct effect on water movement are taken from literature values. The parameter of greatest sensitivity is R_e. In the selection of this parameter, a literature review revealed R_e to be consistently between 75-90 % of total rainfall for Eucalypt forests (Section 3.2.3.1). In the model simulation a value R_e = 90 % of total rainfall was used to incorporate the continual harvesting of trees leading to periods where the canopy is not closed. Changing R_e to 75 % would increase the concentrations by 75 %, but decrease annual loading by around 10 %. However, such a value of R_e would likely only be achieved by not harvesting the trees. The results are sensitive to root depth, which was chosen to be 1 m for the reference scenario. Had half or double, this estimate been used, then the mean annual nitrogen concentration would vary by about 15 % and mean leaching would change by 10 %. Thus, root depth is a parameter that should be obtained via direct measurement when applying the model to a land treatment system. Lowering the crop coefficient (k_c) would increase the drainage, but would have little effect on the overall quantity of nitrogen leaching. The values of E_s and τ used in the simulation have little effect on the output.

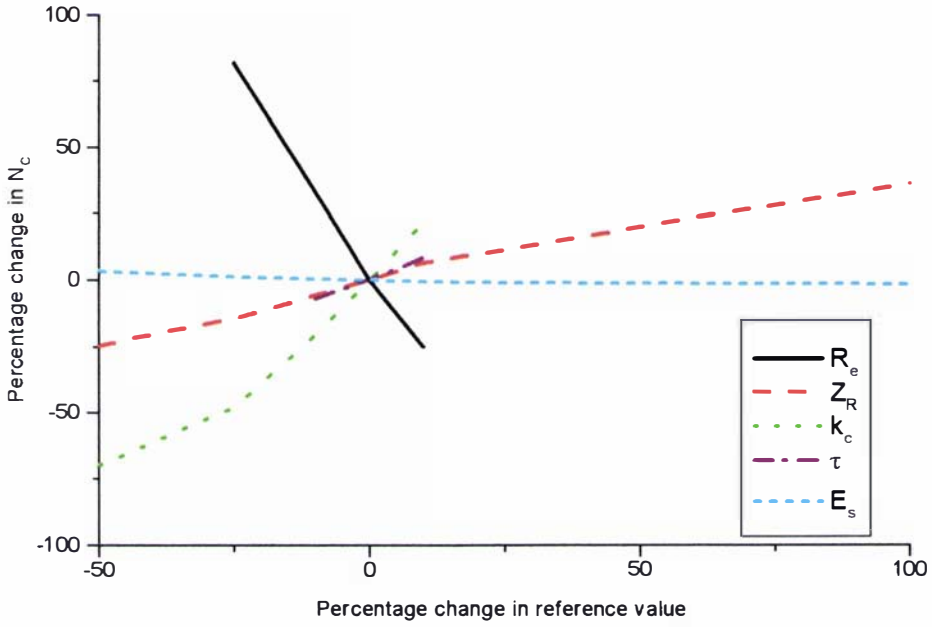

Figure 5.12 Sensitivity analysis of the change in mean-annual nitrogen concentrations in the leachate (N_C) to changes in crop parameters relating to water movement. Reference values are R_e =0.9, z_R =1.0 m, k_c=1.0, τ =0.5, and E_s=2.5 mm day^{-1}.

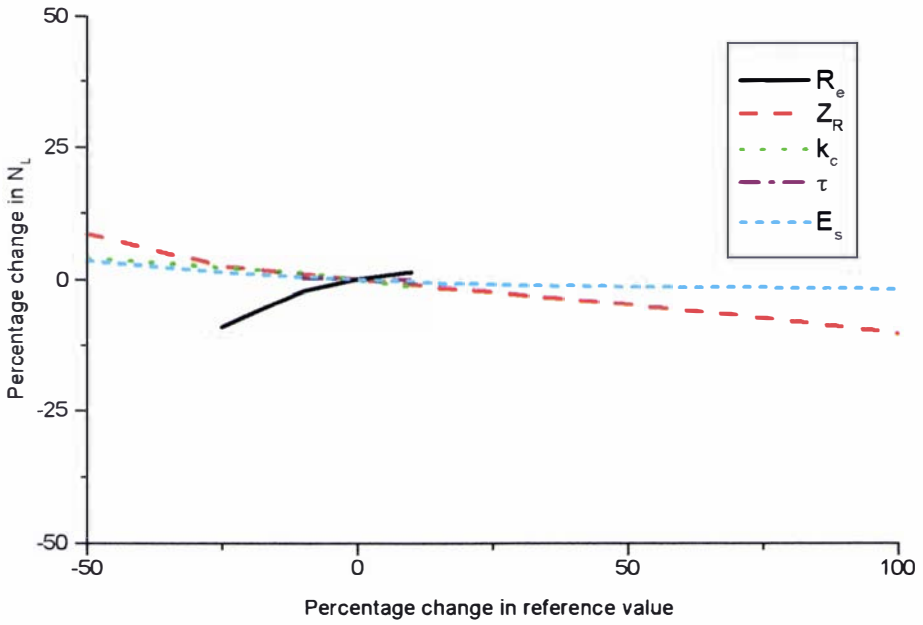

Figure 5.13 Sensitivity analysis of the change in mean-annual nitrogen loading beyond the rootzone (N_L) to changes in the crop parameters relating to water movement. Reference values are R_e =0.9, z_R =1.0m, k_c=1.0, τ (drought tolerance factor)=0.5, and E_s=2.5 mm day^{-1}.

5.6.3 Nitrogen transformation parameters

Changes in the nitrogen transformation rates do not have a large affect on mean nitrogen concentration or mean nitrogen leached (Figure 5.14, Figure 5.15). The denitrification and volatilization parameters have the greatest influence on the model outputs. The most sensitive of the denitrification parameters is the region below field capacity in which denitrification occurs (δ_D). Doubling δ_D lowers the mean nitrogen concentration and the mean nitrogen leached by about 12 %. In comparison, doubling the rate of denitrification (k_3) has a small effect of less than 4 %. If volatilization is increased from 0.5 to 1 times the evaporation rate on the day of effluent application, then the mean nitrogen concentration and the mean nitrogen leached reduce by less than 12 %.

Nitrification and ammonium adsorption coefficients have little effect on the mean nitrogen concentration and mean nitrogen leached in these long term scenarios. This because the rate of transformation of ammonium to nitrate is rapid, so k_{DA} has little affect (even with lower nitrification rates). The adsorption of nitrate (k_{DN}) is always zero in this soil so leaching occurs readily with water.

As stated above, the zone of denitrification below field capacity (δ_D) was taken as 0.06 $m^3\ m^{-3}$ following, Ruz-Jurez et al. (1994). If the model were to be applied to another soil type, then this parameter would need to be estimated as it has a reasonable influence on the output.

The other nitrogen-transformation parameter that had an influence on the results was the rate of volatilization (k_v). The value of k_v used here simulated only a small output of nitrogen via volatilization. Volatilization of dairy-shed effluent applied under forest

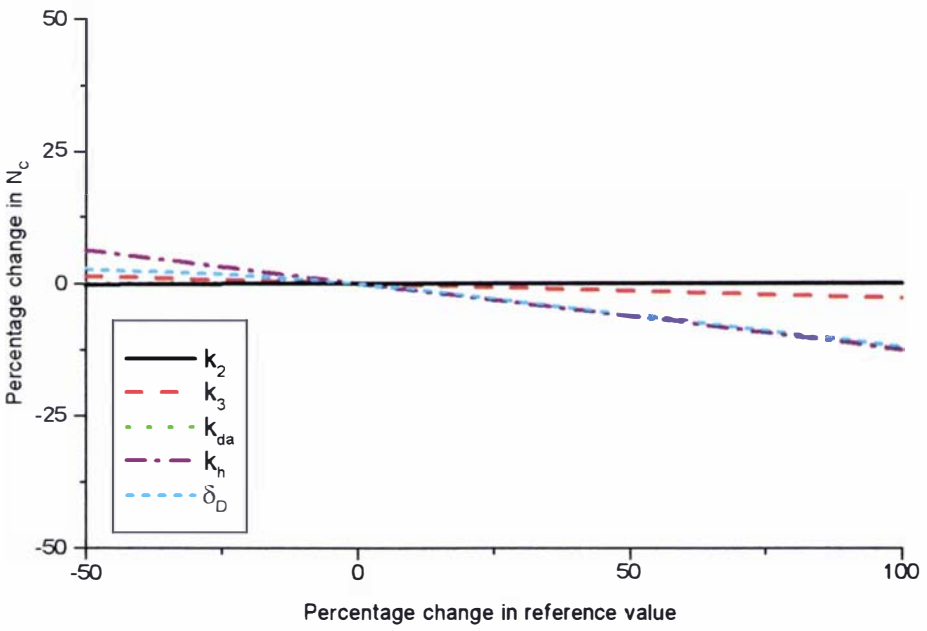

Figure 5.14 Sensitivity analysis of the change in mean-annual nitrogen concentrations in the leachate (N_C) to changes in the parameters of nitrogen transformation rates. Reference values are k_2=0.2 day^{-1}, k_3=0.06 day^{-1}, k_{DA}=5 L kg^{-1}, k_v=0.5 day^{-1} and δ_D=0.06 m^3 m^{-3}.

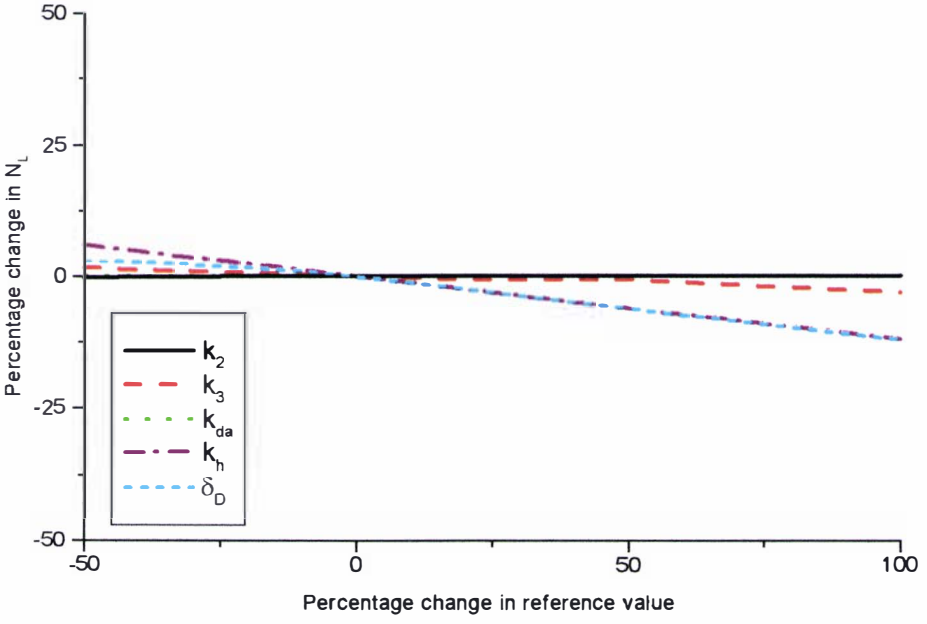

Figure 5.15 Sensitivity analysis of the change in mean-annual nitrogen loading beyond the rootzone (N_L) to changes in the parameters of nitrogen transformation rate. Reference values are k_2=0.2 day^{-1}, k_3=0.06 day^{-1}, k_{DA}=5 L kg^{-1}, k_v=0.5 day^{-1} and δ_D= 0.06 m^3 m^{-3}.

canopies, is likely to be small. However, if the model doubled the losses from volatilization, drainage concentration and mean annual leaching reduce by less than 15 %. Thus the sensitivity of the model output to volatilization losses is similar to denitrification and both should be parameterised for any new simulations of land treatment systems.

5.6.4 Crop parameters of growth and nitrogen uptake

As expected, crop parameters of growth and uptake have a large influence on leachate quality and quantity (Figure 5.16, Figure 5.17). Increasing uptake removes more nitrogen from the soil, making it less available for leaching. The parameter with the largest effect on nitrogen uptake was light utilisation efficiency (ε) which generates plant growth, followed by maximum leaf nitrogen concentration (N_f) and specific leaf area (σ_f). Doubling any of these three parameters (ε, N_f or σ_f) decreases both mean annual nitrogen concentration of the leachate and the mean annual nitrogen leaching by greater than 50 %, because of the direct effect on crop uptake.

The crop growth parameters have important influence on the model output. The parameters used in the reference scenario yield biomass production rates comparable to those measured at other sites (Madgwick et al., 1981; Frederick et al., 1984; Kincheff and Carter, 1991). The model also used reasonable values of maximum nitrogen content and specific leaf area as measured from the field experiment.

5.6.5 Parameters of mineralisation and senescence

Mineralisation is most sensitive to the senescence rate of leaves (Figure 5.18, Figure 5.19). Doubling the senescence rate of leaves results in a greater than 30 % increase in both the mean-annual nitrogen concentration, and the mean-annual leaching loss.

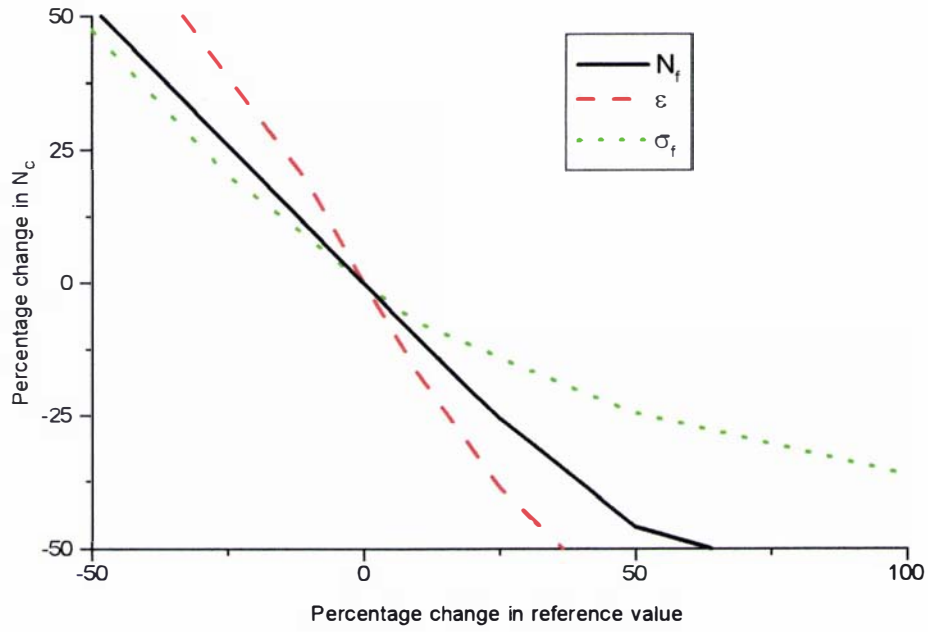

Figure 5.16 Sensitivity analysis of the change in mean-annual nitrogen concentrations in the leachate (N_C) to changes in the parameters of tree growth and uptake. The reference values are N_f = 3 %, ε=1.7 g DM MJ^{-1} and $\sigma_f = 5x10^{-4}$ ha-leaf kg-DM^{-1}.

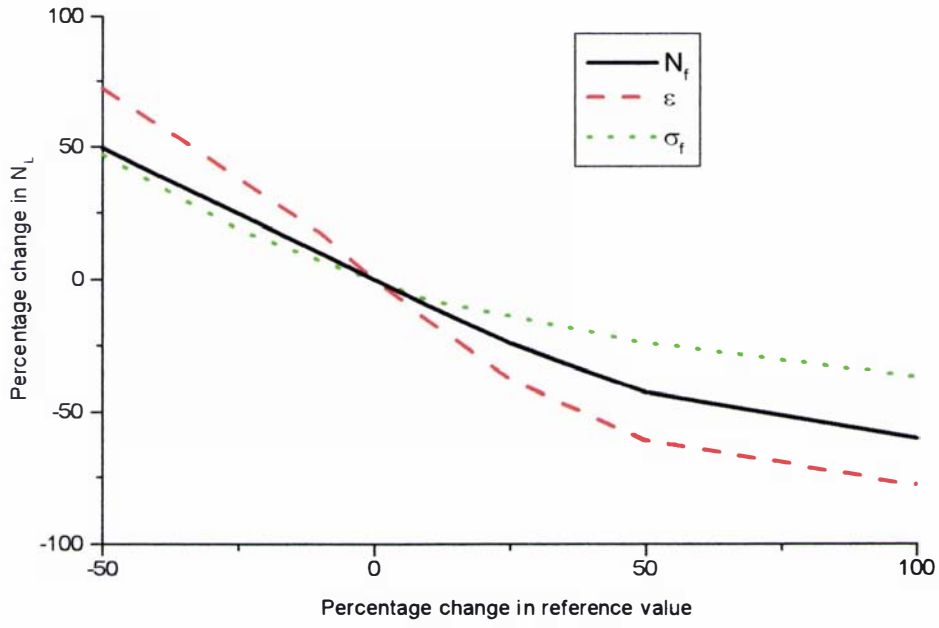

Figure 5.17 Sensitivity analysis of the change in mean-annual nitrogen loading beyond the rootzone (N_L) to changes in the parameters of tree growth and uptake. The reference values are N_f=3 %, ε=1.7 g DM MJ^{-1} and $\sigma_f = 5x10^{-4}$ ha-leaf kg-DM^{-1}.

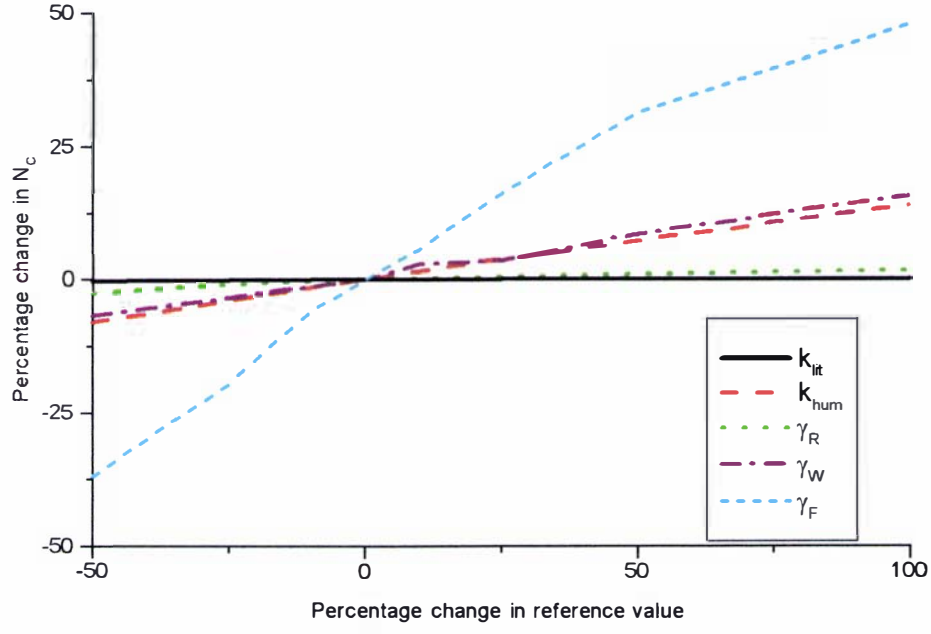

Figure 5.18 Sensitivity analysis of the change in mean-annual nitrogen concentrations in the leachate (N_C) to changes in the parameters of mineralisation. Reference values are k_{lit}=0.0080 day^{-1}, k_{hum}= 7x10^{-5} day^{-1}, γ_R= 0.0055 day^{-1}, γ_W=0.0003 day^{-1}, γ_F=0.0013 day^{-1}.

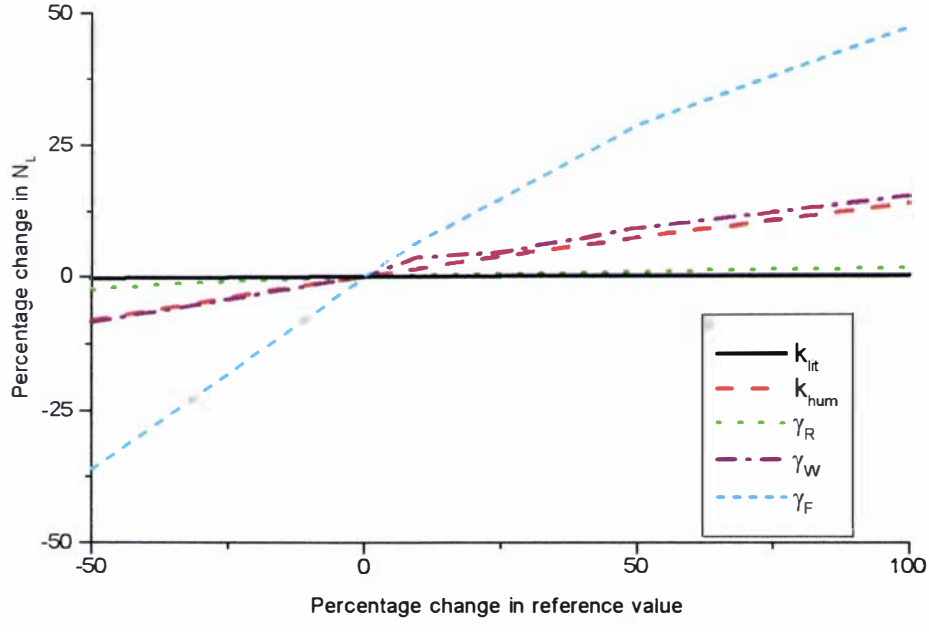

Figure 5.19 Sensitivity analysis of the change in mean-annual nitrogen loading beyond the rootzone (N_L) to changes in the parameters of mineralisation. Reference values are k_{lit}=0.0080 day^{-1}, k_{hum}=7x10^{-5} day^{-1}, γ_R= 0.0055 day^{-1}, γ_W=0.0003 day^{-1}, γ_F=0.0013 day^{-1}.

The next most-sensitive parameters for mineralisation were the senescence rate of woody biomass (γ_W) and mineralisation rate of humus (k_{hum}). However, doubling these had less than a 10 % change in the mean leachate concentrations and mean annual leaching. This is because leaves have the highest nitrogen contents of all plant tissues and so the decomposition of leaves has the greatest effect on leaching rate.

The rates of senescence used in the reference scenario were sourced from the literature values (King, 1993). The sensitivity of the model to leaf senescence rate, shows this is an important source of nitrogen to the system. A current PhD study at Massey University is investigating leaf turnover and mineralisation in effluent irrigated SRF land treatment systems (Guo, 1999). This new research will provide more understanding of the role of leaf litter recycling in SRF land treatment systems.

5.6.6 Summary

Nitrogen concentration of the drainage water was most sensitive to water movement parameters of z_R, R_c, β, θ_w, k_c. These all influence the amount of water draining from the rootzone and therefore were important parameters for the model. The most sensitive parameters for nitrogen fate were δ_D, ε, N_f, σ_f, γ_F, γ_R, and k_{hum}. It seems reasonable that changes in crop uptake, denitrification and volatilization losses as well as mineralisation will each change the amount of nitrogen leaching from the system. It is possible that these features of the water and nitrogen balances could be modified through management practices, in order to minimise nitrogen leaching.

5.7 Management strategies

The LPM model has been developed with the capacity to investigate simple changes in management practices for a SRF system treating dairy-shed effluent. Now, the model's applicability as a decision support tool is demonstrated, through altering one key aspect of management, the annual nitrogen loading rate. The nitrogen loading rate is seen as a major determining factor for sustainability of such systems, and thus it warrants investigation.

The reference scenario considered an annual loading rate of 200 kg N ha^{-1} yr^{-1}. This rate conforms to the current guidelines for land-based effluent schemes. The model is now used to investigate further loading rates of 50, 100, and 400 kg N ha^{-1} yr^{-1}. As with the previous scenario, all irrigation is applied during a 185-day 'summer' period. The change in loading rates was achieved by irrigating more or less effluent as required to achieve the rate. For example, to achieve an increase from 200 to 400 kg N ha^{-1} yr^{-1}, the weekly irrigation events were increased from 9.5 to 19 mm $week^{-1}$.

The mean-annual water and nitrogen budgets from these scenarios are presented in Table 5.5. Time-series data of the various loading rates, for 1986-1992 (Figure 5.20, 5.21, 5.22 and 5.23) show, through individual events, the effect of loading rate on leachate quantity and concentration. The increased loading, however, did not lead to run off being predicted. Increased hydraulic loading did however increase the volumes of drainage. Although effluent irrigation added a further 442 mm of water to the 400 kg N ha^{-1} yr^{-1} scenario, compared to the 200 kg N ha^{-1} yr^{-1}, the drainage volume only increased by about half of that, i.e. 221 mm. The rest of the extra irrigation water went into increasing ET, i.e. 216 mm with the remaining 5 mm increasing the soil water

storage. Figures 5.20b, 5.21b, 5.22b and 5.23b show the increase in water content of the soil over summer as a result of greater irrigation. As the soil water content remains low in summer for all application rates, it is foreseeable that increased irrigation in this period would increase mean annual ET, as was observed (Table 5.5).

Uptake of nitrogen was reduced at the application rates of 50 and 100 kg N ha^{-1} yr^{-1}, because of low soil water and nitrogen availability. That restricted growth and lowered the biomass production. The 50 kg N ha^{-1} yr^{-1} loading rate had only 6 harvests compared to the 3 other scenarios that had 7. Biomass production increased slightly when the loading rate was increased from 50 to 100 kg N ha^{-1} yr^{-1}. However further increase in nitrogen loading from 100 to 200 kg N ha^{-1} yr^{-1} did not cause significant increases in biomass production, indicating optimal growth was already achieved.

The mineralisation rates did not vary greatly for the loading rates. Denitrification increased with greater irrigation. This was a response to soil-water contents more often being in the range where denitrification occurs. Volatilization was on average lower for the 50 kg N ha^{-1} yr^{-1} application rate. This would have been due to less harvests being achieved for this simulation providing less evaporation for ammonia volatilisation. Evaporation is a larger proportion of ET after harvest. Thus when effluent is applied at such times the volatilization losses are near their maximum values.

The concentration and quantity of nitrogen leached greatly increased as loading rate increased (Figures 5.20, 5.21, 5.22, and 5.23). Only the 50 kg N ha^{-1} yr^{-1} scenario yielded a mean leachate concentration below the NZDWS. Figure 5.20c, however, shows that in the 50 kg N ha^{-1} yr^{-1} scenario, the concentration did exceed the NZDWS

following some events. To investigate further the effect of loading rates on leachate concentration and quantity the model was run for loading rates of 0, 25, 75, 125, 150 and 300 kg N ha^{-1} yr^{-1}. The model predicts that the mean-annual leachate-concentration will exceed NZDWS limit if the application rates exceed around 75 kg ha^{-1} yr^{-1} (Figure 5.24).

Table 5.5 Mean-annual budgets for water and nitrogen of a SRF system receiving different loading rates of dairy-shed effluent.

Application rate (kg ha^{-1} yr^{-1})	50	100	200	400
Water balance (mm)				
Rainfall	815.6	815.6	815.6	815.6
Irrigation	63.1	126.3	252.6	505.3
Evapotranspiration	757.9	802.8	884.5	974.1
Drainage	123.8	141.4	184.3	345.0
Run off	0	0	0	0
Nitrogen balance (kg N ha^{-1} yr^{-1})				
N applied	51.2	102.6	205.1	410.2
Mineralisation	29.2	31.1	31.9	31.8
N uptake	61.9	95.7	112.8	112.9
Denitrification	0.3	0.7	3.5	12.2
Volatilization	8.8	13.7	13.3	13.3
Leaching	9.32	21.6	101.7	293.9
Leachate conc. (mg L^{-1})				
NO_3^-	7.2	15.3	61.9	90.7
NH_4^+	0	0	0	0.1
Biomass				
No. harvests	6	7	7	7
MAI (Mg ha^{-1} yr^{-1})	14.0	17.0	18.5	18.5

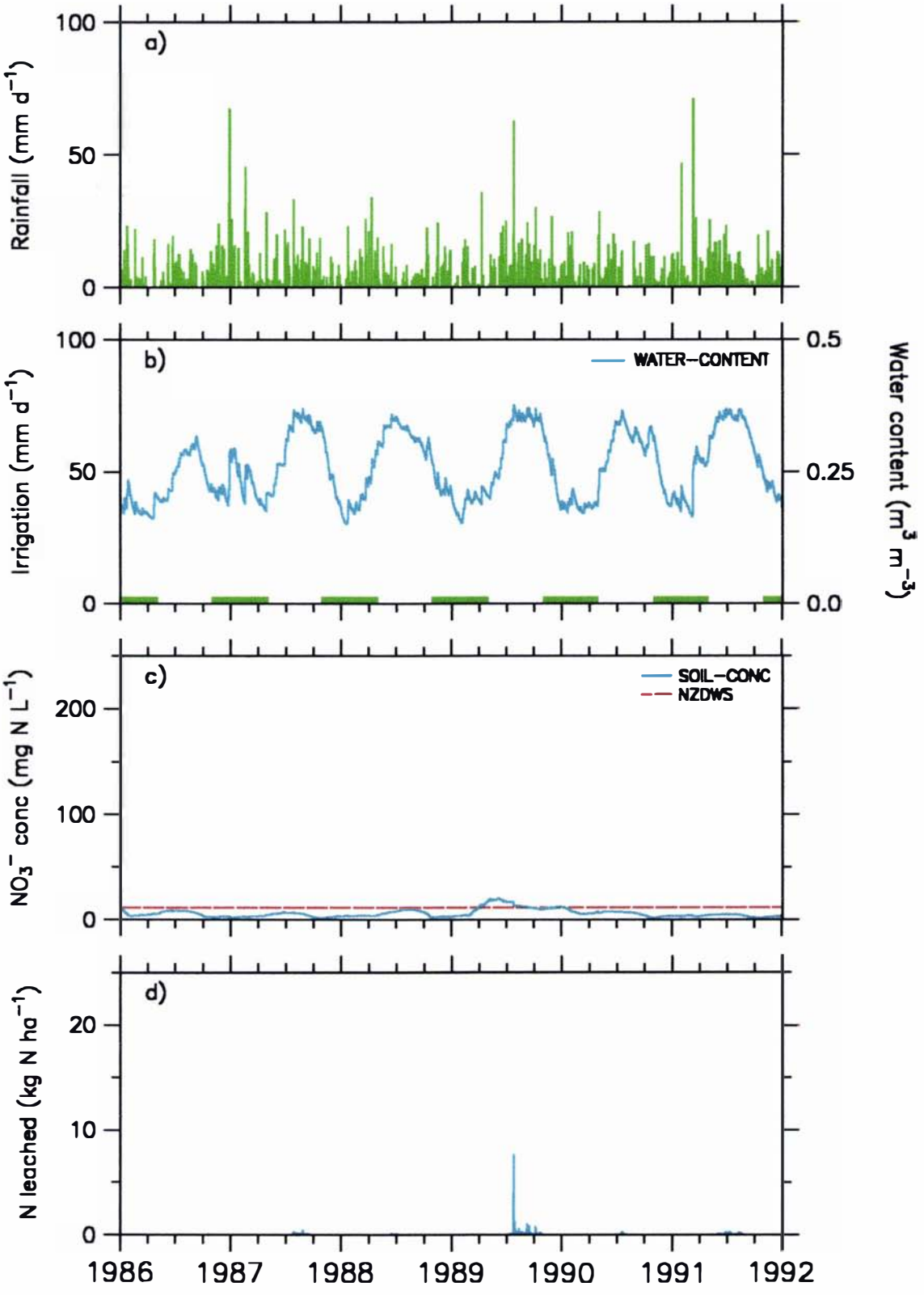

Figure 5.20 Model output for SRF recciving effluent application of 2.38 mm week^{-1} in the summer. Total annual effluent loading is 50 kg N ha^{-1} yr^{-1} a) daily rainfall, R (mm) b) weekly irrigation, I (mm) and water content of the soil profile, θ (m^3 m^{-3}) c) leachate nitrate concentration, NO_3^- conc. (mg L^{-1}) d) daily drainage of nitrogen from the root zone, N leached (kg ha^{-1})

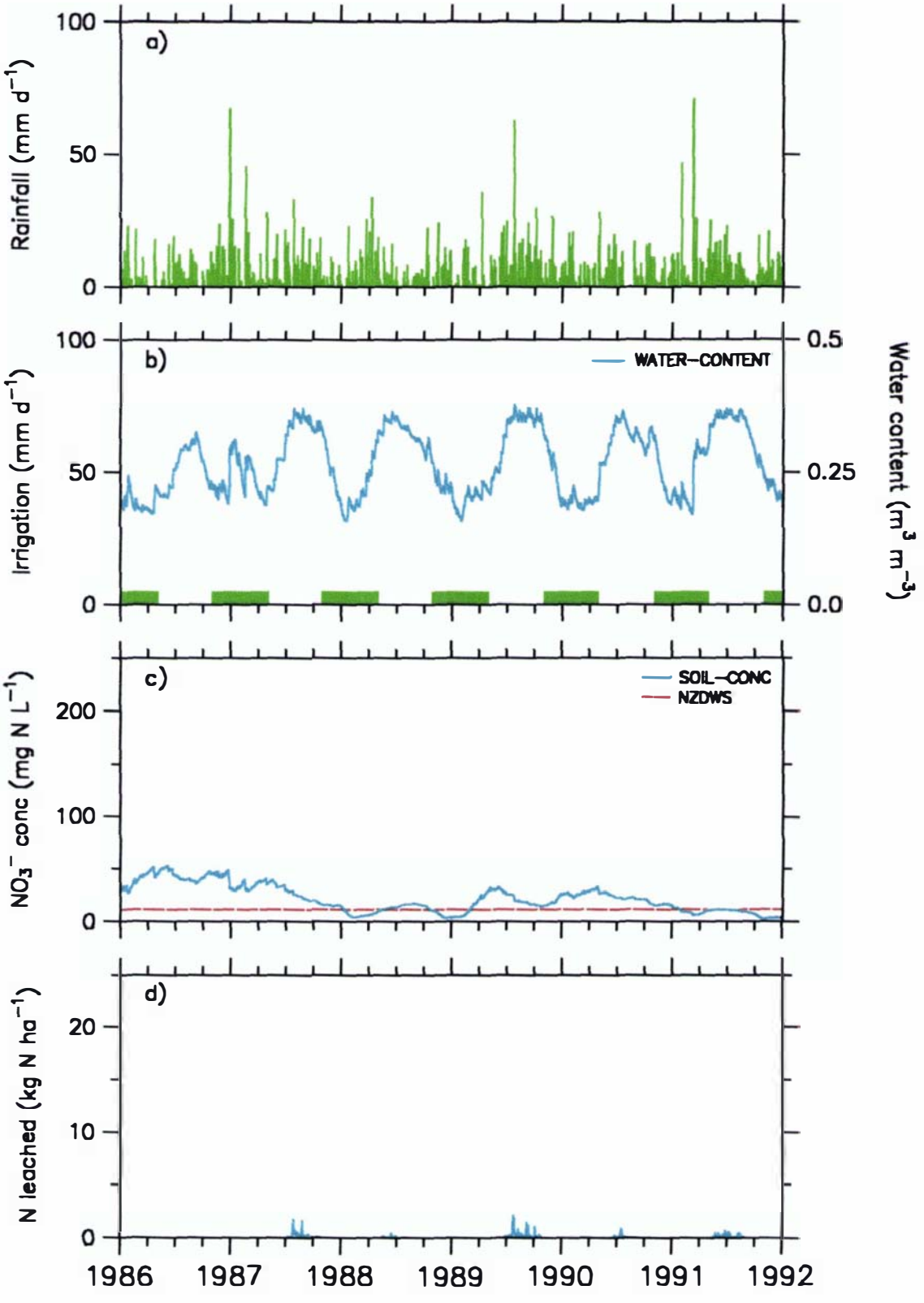

Figure 5.21 Model output for SRF receiving effluent application of 4.75 mm week^{-1} in the summer. Total annual effluent loading is 100 kg N ha^{-1} yr^{-1} a) daily rainfall, R (mm) b) weekly irrigation, I (mm) and water content of the soil profile, θ (m^3 m^{-3}) c) leachate nitrate concentration, NO_3^- conc. (mg L^{-1}) d) daily drainage of nitrogen from the root zone, N leached (kg ha^{-1})

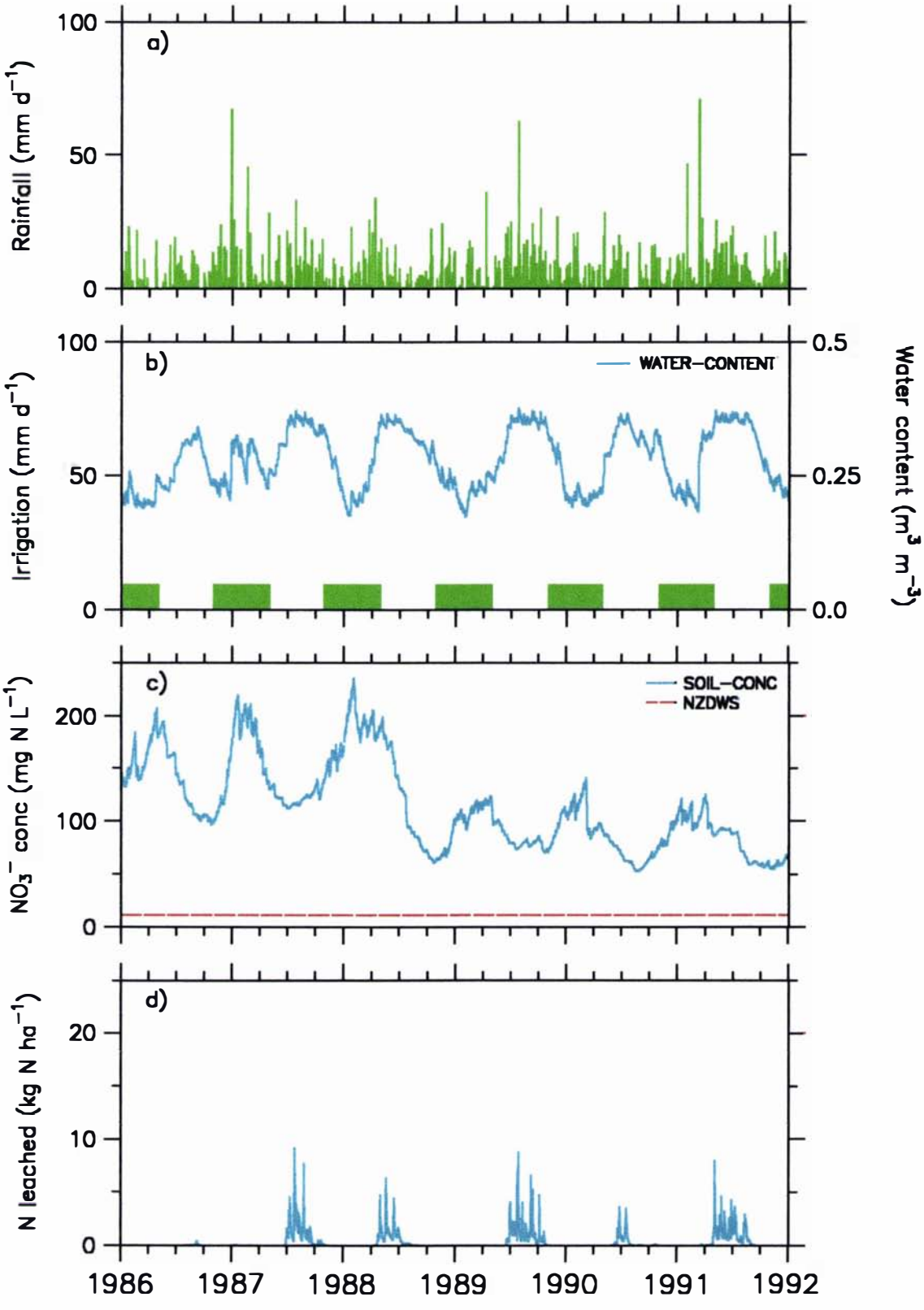

Figure 5.22 Model output for SRF receiving effluent application of 9.5 mm week^{-1} in the summer. Total annual effluent loading is 200 kg N ha^{-1} yr^{-1} a) daily rainfall, R (mm) b) weekly irrigation, I (mm) and water content of the soil profile, θ (m^3 m^{-3}) c) leachate nitrate concentration. NO_3^- conc. (mg L^{-1}) d) daily drainage of nitrogen from the root zone, N leached (kg ha^{-1}).

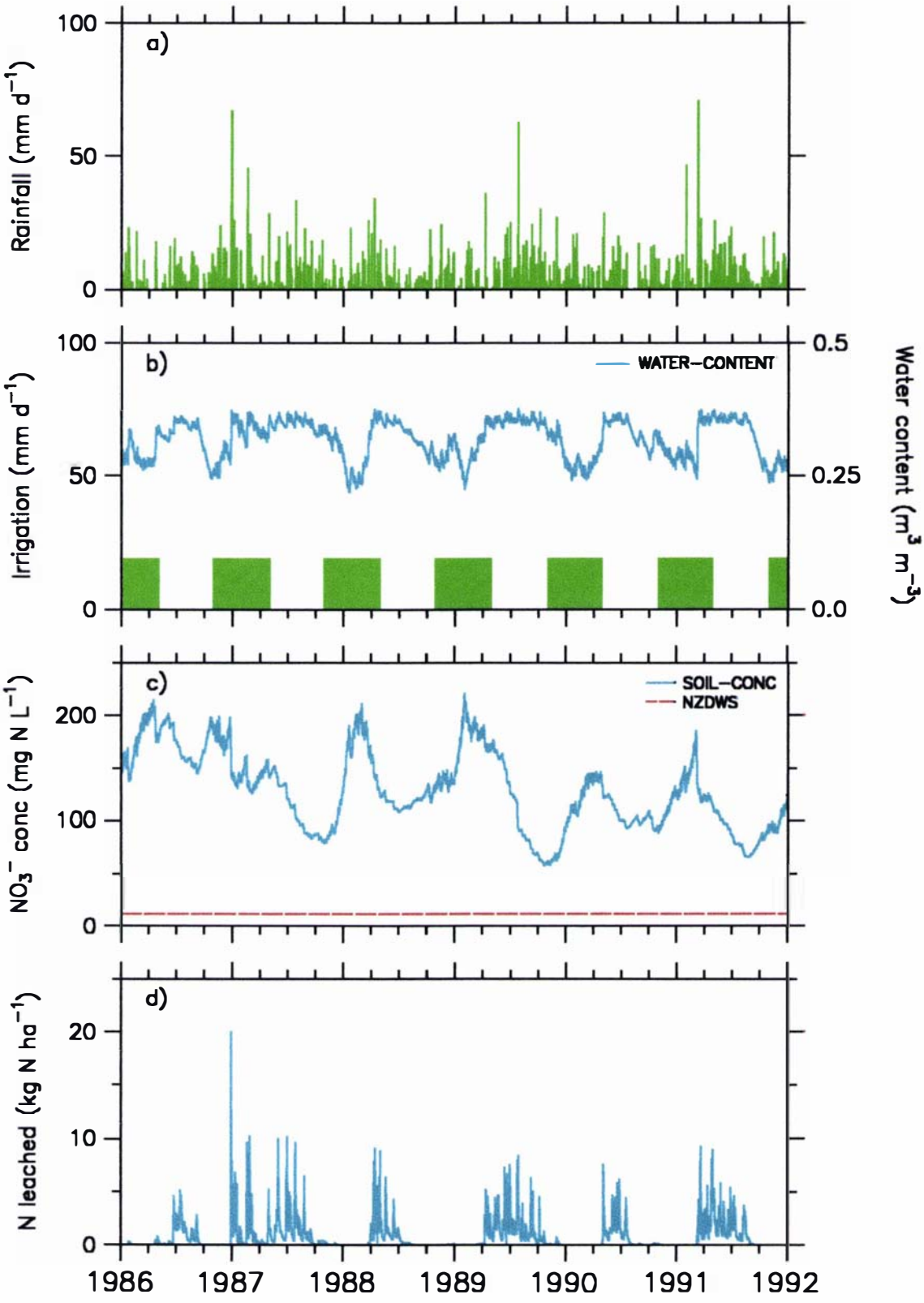

Figure 5.23 Model output for SRF receiving effluent application of 19 mm week^{-1} in the summer. Total annual effluent loading is 400 kg N ha^{-1} yr^{-1} a) daily rainfall, R (mm) b) weekly irrigation, I (mm) and water content of the soil profile, θ (m^3 m^{-3}) c) leachate nitrate concentration, NO_3^- conc. (mg L^{-1}) d) daily drainage of nitrogen from the root zone, N leached (kg ha^{-1}).

Nitrogen concentrations in the leachate did not increase linearly with loading rate (Figure 5.24). Rather nitrogen concentrations rapidly increased as the loading rate increased from 100-300 kg N ha^{-1} yr^{-1}. The leachate concentrations reached the maximum level of around 87-90 mg N L^{-1} at loading rates above 300 kg N ha^{-1} yr^{-1}. This is similar to the concentration in the effluent applied of 81.9 mg N L^{-1}. The concentration in the leachate is slightly increased above the effluent concentration. This slight increase is likely a combination of inputs from mineralisation and concentration by evaporative loss of soil water. The quantity of nitrogen going to groundwater increases steadily reaching 37 kg N ha^{-1} yr^{-1} for the inputs of 125 kg N ha^{-1} yr^{-1} (Figure 5.25). Beyond an input of 125 kg N ha^{-1} yr^{-1}, the mean-annual leachate quantity increases by nearly 1 kg N ha^{-1} yr^{-1} for every 1 kg N ha^{-1} yr^{-1} extra added.

If the critical factor of sustainability of the nitrogen inputs from effluent is determined by the concentration of nitrogen leaching to groundwater being on average less than the NZDWS, then a sustainable level of nitrogen loading is likely around or below 75 kg N ha^{-1} yr^{-1} (Figure 5.24). However without dilution in the aquifer over the long term, the ground water nitrate levels may increase.

If the critical factor of sustainability of the nitrogen inputs from effluent is determined by the total loading of nitrogen to groundwater, the hydrogeological issue of how much loading the aquifer can sustain must be addressed. This will be very site specific. However it is possible to use the LPM model to provide drainage fluxes and nitrogen concentrations. This information can provide loading-rate details, which could be used by a hydrogeologist to determine the likely mixing and dilution factor that will

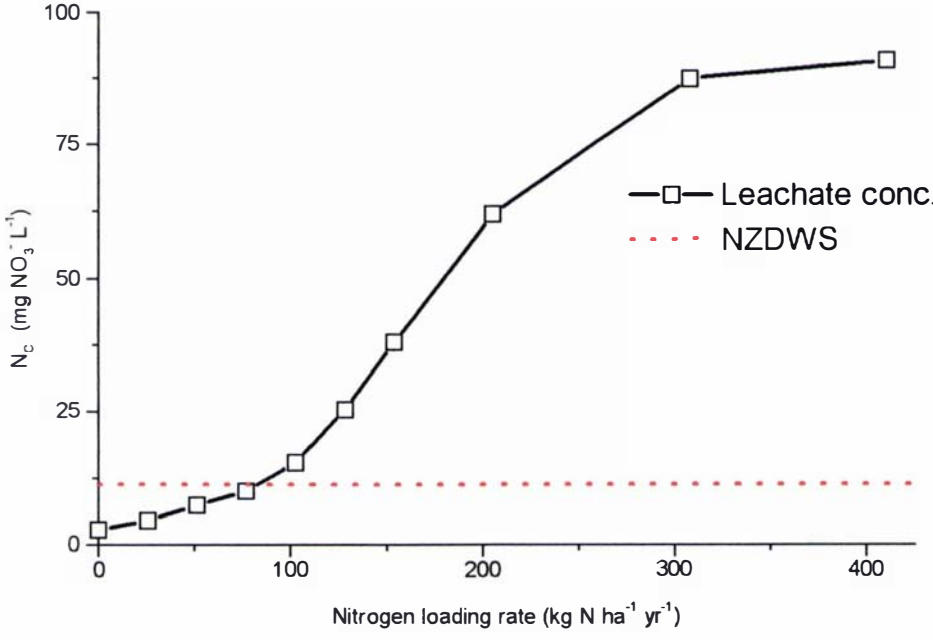

Figure 5.24 Effect of increasing nitrogen loading rate onto a SRF dairy-shed effluent system on mean-annual leachate nitrogen-concentration (N_C).

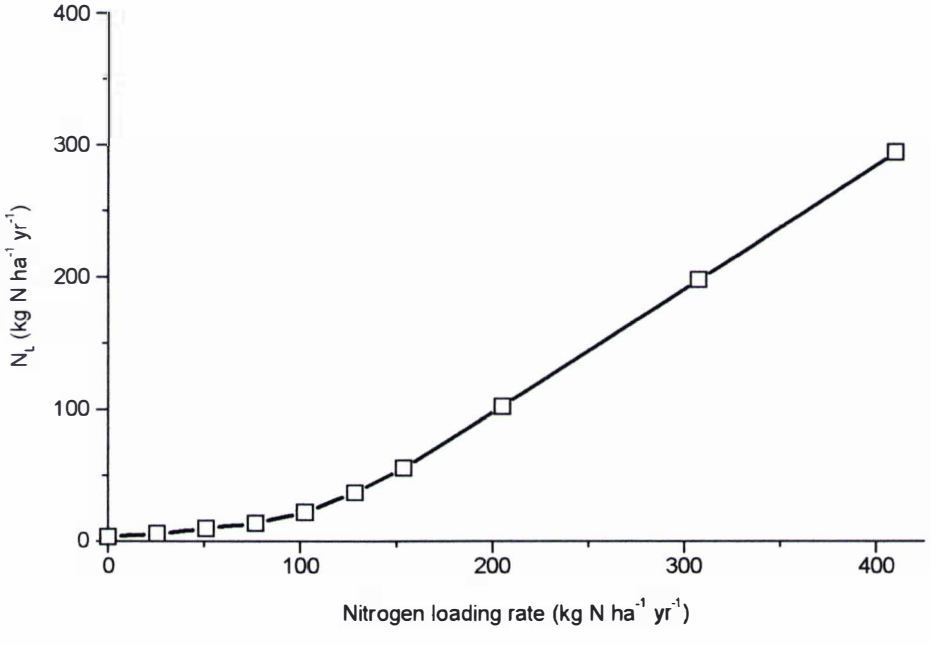

Figure 5.25 Effect of increasing nitrogen loading rate onto a SRF dairy-shed effluent system on mean annual nitrogen leaching (N_L).

determine the long term sustainability of the system. However, if the sustainable nitrogen loading rate to the aquifer is known then a figure similar to Figure 5.25 could be utilised to determine the annual loading rates that would be appropriate for a given land treatment system.

Thus the LPM model's ability for investigating management issues has been demonstrated. While many other management options could be investigated using LPM, the nitrogen loading rate is most likely to have the greatest influence on the nitrogen leaching to groundwater.

5.8 Conclusions

The model simulations of the bare-soil treatment showed broad agreement with measured data from the lysimeter experiment. Although the model predicted leachate nitrogen concentrations that were on average lower than the lysimeter data, the concentrations were well within the scatter of values from the lysimeter study. Simulation of the *E. nitens* treatment of the lysimeter study also showed good agreement with the field study results for leaching of nitrogen.

The model was then used to simulate the behaviour of a SRF plantation receiving dairy-shed effluent at a rate of 200 kg N ha^{-1} yr^{-1} over 27 years. This simulation predicted the occurrence of high nitrate concentrations in the leachate. This would be a limiting factor for the long term sustainability of such a system. A sensitivity analysis of the model results revealed the important parameters of water movement and nitrogen cycling that effect both nitrogen concentration and quantity in the leachate moving below the root zone. Water movement was most sensitive to root zone depth, effective rainfall, available water, and crop water use. The nitrogen fate parameters with greatest effect on leachate concentration and quantity were denitrification activity and volatilisation. Plant growth parameters of light utilisation efficiency, maximum leaf nitrogen concentration and specific leaf area strongly effected leachate nitrogen concentration and quantity. Mineralisation rates of the soil humus and the senescence rates of plant material also impacted on quantity and quality of nitrogen leaching.

The model's applicability as a decision support and advice tool was demonstrated through examining the impact of various effluent loading rates on the leachate concentration and leachate quantity. Based on leachate nitrate concentrations being on

average lower than the NZDWS of 11.3 mg N L^{-1}, the sustainable loading rate predicted for the simulated system was around 75 kg N ha^{-1} yr^{-1}.

Chapter 6 General Conclusions

6.1 Summary and conclusions

The three objectives of this study were;

1. To measure and calculate the water and nitrogen balances of three species of SRF and a bare soil control receiving dairy-shed effluent irrigation;
2. To parameterise the LPM model utilising data from the field experiment and literature values to simulate the water and nitrogen balances of the field experiment;
3. To demonstrate the applicability of the LPM model as a decision support tool.

This concluding chapter summarises the findings.

The lysimeter experiment investigated 3 SRF species and a bare soil control for their ability to renovate dairy-shed effluent. The study measured water and nitrogen budgets of the four treatments and also investigated the biomass production of the three species.

Rainfall during the 620 day experiment totaled 1830 mm. Effluent irrigation was applied over two irrigation periods using sprinkler irrigation. Hydraulic loading during the application periods was 21.5 mm $week^{-1}$. In the first irrigation period a total of 618 mm was applied from December 5, 1995 to June 16, 1996. In the second irrigation period, 670 mm was applied from September 16, 1996 to April 14, 1997. Nitrogen loading during the first irrigation period totaled 70 g N $lysimeter^{-1}$ (equivalent to 279 kg N ha^{-1}). Nitrogen loading during the second irrigation period totaled 148 g N $lysimeter^{-1}$ (equivalent to 592 kg N ha^{-1}). Thus total application during the experiment totaled to 871 kg N ha^{-1}. The higher total loading in the second season was due to higher concentrations of nitrogen in the effluent.

The total tree biomass production of the *E. nitens* was significantly greater than *E. saligna*, with *S. kimuyanagi* being intermediary. The woody biomass production, however was significantly greater for both *E. nitens* and *S. kimuyanagi* in comparison to the *E. saligna*. Biomass production per hectare were estimated from the lysimeter-grown trees. These should however be treated with caution. Total above-ground biomass production estimates for the 2.5 years, based on a stocking density of 4000 stems ha^{-1} was equivalent to 15.6, 30.6, and 21.3 Mg ha^{-1} $year^{-1}$ for *E. saligna*, *E. nitens*, and *S. kimuyanagi* respectively. Although scaling up biomass estimates from small plot trials and particularly lysimeters introduces associated errors, the estimates fell within the ranges measured elsewhere in New Zealand.

The *E. saligna* trees were infested by the gall wasp *Ophelimus eucalypti* but the effect of this parasite on growth was difficult to determine and warrants further investigation. If *E. saligna* is to be used for SRF plantations in New Zealand the effect of this pest might need to be considered.

The water balances of the four treatments were markedly different. The modified bare-soil evaporation model of Kerr (1974) was in good agreement with field measurements of bare-soil evaporation. The bare-soil treatments had high amounts of drainage throughout the course of the experiment. For the bare soil treatments, rainfall and effluent inputs had a considerable influence on the amount of leachate from the soil zone.

The inclusion of trees to the system added transpiration to the hydraulic regime, which greatly effected the water balance. *E. saligna*, initially with small leaf area, behaved at

first similar to the bare-soil treatment. At this stage, the ET demand in the tree was lower than water inputs, resulting in drainage for a considerable part of the experiment, as was observed for the for the bare soil treatment. Once the leaf area increased, the ET demand was greater than water inputs and drainage ceased.

Both *E. nitens* and *S. kimuyanagi* had high ET demand during the first irrigation season, which lowered the soil water storage close to the wilting point. In the second irrigation season, both species again used water at a rate which lowered water content to the wilting point. The *E. nitens* showed no significant drop in leaf area however the *S. kimuyanagi* showed significant loss in leaf area. This dramatic physiological change in willow was attributed to water stress. Growth of *E. nitens* and *S. kimuyanagi* was probably limited by this water stress. Water use of the trees is likely to have been greater had there been a greater supply of irrigation at the times of water stress.

Maximum water use estimates for the trees are slightly higher than those reported elsewhere. In part, this can be attributed to difficulties in ascertaining the ground area to which the tree water use relates.

Water use of the *E. nitens* and *S. kimuyanagi* differed over the winter. The deciduous *S. kimuyanagi* shed leaves in winter decreasing both ET and rainfall interception. This resulted in greater winter drainage from *S. kimuyanagi* than from the evergreen *E. nitens* trees. Consequently the leaching period of the *E. nitens* was shorter than that of the *S. kimuyanagi*. Both these treatments leached for shorter periods than *E. saligna*. Leaching of the bare-soil treatment was consistently high throughout the experiment. Water use was found to have a large impact on drainage volume and timing.

This study has been successful in determining the nitrogen balance for the four treatments receiving dairy-shed effluent. The nitrogen balance was calculated as the difference between inputs, (effluent application) and outputs (plant uptake and leaching) and adjusted for changes in soil storage. Complete closure of the mass balance of nitrogen, however, was not achieved. Outputs and storage changes from *E. nitens* were 4 % less than inputs. For bare soil, *S. kimuyanagi* and *E. saligna* treatments, the balance exceeded the inputs by 12, 13 and 30 %. Some of the assumptions involved in the measurements and subsequent calculations of various components of the balance may have contributed to this. Factors not measured in the balance included volatilization and denitrification. Volatilization and denitrification losses were each estimated to be less than 5 % of the total inputs from effluent application. It was concluded, that these may have played only a small part in not achieving mass balance. The major contributors to not achieving mass balance were assumed to be the inclusion of plant uptake during the establishment year and errors in estimation of the change in soil nitrogen storage over the experiment.

In this study, trees have been shown to improve effluent treatment because high evapotranspiration rates reduced the volume of leachate passing beyond the root zone. Further, uptake of nitrogen by the trees reduced the quantities of nitrogen available for leaching. In this study both *E. nitens* and *S. kimuyanagi* were considered more suitable than the other 2 treatments evaluated for land treatment. The low nitrogen concentration in the leachate under the *S. kimuyanagi* was the key criterion which determined the suitability of this tree species for land treatment of effluent. The low total loading of nitrogen to the groundwater of the *E. nitens* treatments was the key criterion in

determining *E. nitens* suitability. Both of these species showed high levels of biomass production and nutrient accumulation.

However nitrogen concentrations in the leachate of all treatments were greater than the New Zealand drinking water standard of 11.3 mg NO_3^--N, during certain periods of the experiment. From the lysimeter experiment it was concluded, that the leachate nitrogen concentrations might have been reduced had the nitrogen loading rate through effluent application been kept low.

The project then moved from experimental study to modelling water and nitrogen balances of SRF land treatment systems. Ultimately, the aim of the model was to investigate the effect of changes in management practices on sustainability in terms of nitrogen leaching of SRF systems treating dairy-shed effluent. The model selected for this purpose was a lumped parameter model (LPM), similar to that recently developed by Ling and El-Kadi (1998).

The model simulations of the bare soil treatment showed broad agreement with measured data of lysimeter experiment. Although the model predicted leachate nitrogen concentrations that were on average lower than the lysimeter data, the predicted concentrations were well within the scatter of values from the lysimeter study. Simulation of the *E. nitens* treatment of the lysimeter study showed good agreement with the field study results for leaching of nitrogen.

The model was used to simulate the behaviour of a SRF plantation receiving dairy-shed effluent at a rate of 200 kg N ha^{-1} yr^{-1} over 27 years. This simulation predicted the

occurrence of high nitrate concentrations in the leachate. This would be a limiting factor for the long term sustainability of such a system. A sensitivity analysis of the model results revealed the important parameters of water movement and nitrogen cycling that effect both nitrogen concentration and quantity in the leachate moving below the root zone. Water movement was most sensitive to root zone depth, effective rainfall, available water and crop water use. The nitrogen fate parameters with greatest effect on leachate concentration and quantity were denitrification activity and volatilisation. Plant growth parameters of light utilisation efficiency, maximum leaf nitrogen concentration and specific leaf area strongly effected leachate nitrogen concentration and quantity. Mineralisation rates of the soil humus and the senescence rates of plant material also impacted on quantity and concentration of nitrogen leaching.

The models applicability as a decision support tool was demonstrated by examining the impact of various effluent loading rates on the leachate concentration and leachate quantity. The model indicated that the maximum sustainable nitrogen loading rate to maintain the leachate nitrogen concentration below the NZDWS of 11.3 mg N L^{-1} was around 75 kg N ha^{-1} yr^{-1}.

The major finding of both the lysimeter experiment and the modelling study was limitation of the high concentrations leaching from SRF dairy-shed effluent treatment systems. The model clearly provides a platform from which to investigate many other possible scenarios of management to investigate minimising the leaching of the high concentrations of nitrogen into the ground water.

6.2 Future directions

This study of SRF land treatment of dairy-shed effluent has highlighted several areas of research that could be further pursued. The lysimeter study indicated that the loading rate of over 500 kg N ha^{-1} yr^{-1} was unsustainable due to the high concentrations of nitrate leaching. Future experimental research could focus on nitrogen leaching at application rates lower than this level. Further land irrigation of effluents with high concentration of nitrogen is likely to result in high concentration of nitrogen in the leachate. It would be interesting to examine the effect of varying nitrogen concentration in the effluent on the leachate concentration. The nitrogen concentration in the effluent can be varied through dilution using irrigation water.

Future experiments and/or modelling could address the issues of optimising irrigation to minimise leaching. From the lysimeter experiment it was speculated that increased irrigation in the summer might have increased plant uptake and reduced nitrogen leaching in the winter. Future investigations of SRF using lysimeters may be advised to use larger, deeper lysimeters, so as not to restrict the root growth of the trees. Incorporating more than one tree per lysimeter may also be advantageous so as to better simulate the plantation scale.

In comparison to the evergreen trees, the deciduous trees had increased drainage flux in winter and this decreased the concentrations of nitrate leaching. The difference in the leaching patterns for the evergreen and deciduous trees warrants further investigation.

If SRF systems are to be utilised for dairy-shed effluent treatment many management decisions require further investigation. The coppicing phase of the short rotations

presents many challenges in terms of providing marketable fuelwood, whilst maintaining a sustainable system. In the lysimeter experiment, however a large proportion of the leaching from the tree treatments occurred during the coppicing phase

The computer model provides platform from which changes in management can be investigated. Further consolidation of the sensitive parameters of the model is an area of research that would be quite beneficial.

The model can be used for investigating crops other than SRF with careful parameterisation. Indeed the model has been utilised to investigate both pine trees and pasture for the land treatment of municipal effluent (Clothier et al. 1999). As pasture is the dominant crop in land treatment for dairy farm effluent in New Zealand such research would be very beneficial.

The model was unable to simulate deciduous trees. Incorporating seasonal changes in senescence into model would allow further investigation of deciduous trees with the model. As the deciduous trees preformed well in the lysimeter experiment this is seen as important area for future development of the model.

References

Asdak, C., Jarvis, P. G. and Gardingen, P. V., 1998. Evaporation of intercepted precipitation based on an energy balance in unlogged and logged forest areas of central Kalimantan, Indonesia. Agricultural and Forest Meteorology, 92: 173-180.

Aston, A. R., 1979. Rainfall interception by eight small trees. Journal of Hydrology, 42: 383-396.

Atkinson, D., 1985. Spatial and temporal aspects of root distribution as indicated by the use of a root observation laboratory. In: A. H. Fisher, D. Atkinson, D. Read and M. B. Usher (Editors), Biological interactions in soil. Blackwell Scientific Publications, Oxford.

Atkinson, D. and Chauhan, J. S., 1987. The effect of paclobutrazol on the water use of fruit plants at two temperatures. Journal of Horticultural Science, 62(421-426): 43-65.

Bari, M. A. and Schofield, N. J., 1992. Lowering of a shallow, saline water table by extensive Eucalypt reforestation. Journal of Hydrology, 133: 273-291.

Barton, L., McLay, C. D. A., Schipper, L. A. and Smith, C. T., 1998. Procedures for characterising denitrification rates in a wastewater-irrigated forest soil. Australian Journal of Soil Research, 36: 997-1008.

Battaglia, M., Cherry, M. L., Beadle, C. L., Sands, P. J. and Hingston, A., 1998. Prediction of leaf area index in Eucalypt plantations: effects of water stress and temperature. Tree Physiology, 18: 521-528.

Bell, R. W., Schofield, N. J., Loh, L. C. and Bri, M. A., 1990. Ground water response to reforestation in the Darling Range of Western Australia. Journal of Hydrology, 119: 179-200.

Beauchamp, E. G., Kidd, G. E. and Thurtnell, G., 1982. Ammonia volatilization from liquid dairy manure applied in the field. Canadian Journal of Soil Science, 62: 11-19.

Binkley, D. and Ryan, M. G., 1998. Net primary production and nutrient cycling in replicated stands of *Eucalyptus saligna* and *Albizia facaltaria*. Forest Ecology and Management, 112: 79-85.

Birk, E., 1995. Fertiliser use in the management of Pine and Eucalypt Plantations in Australia: A review of past and current practices. New Zealand Journal of Forestry Science, 24(2/3): 289-320.

Bond, W. J., 1998. Effluent irrigation - an environmental challenge for soil science. Australian Journal of Soil Research, 36: 543-555.

Bremner, J. M. and Keeney, D. R., 1966. Determination and isotope-ratio analysis of different forms of nitrogen in soil : 3. Exchangeable ammonium, nitrate and nitrite

by extraction distillation methods. Soil Science Society of America Proceedings, 30: 577-582.

Bremner, J. M. and Mulvaney, C. S., 1982. Nitrogen-Total. In: A. L. Page (Editor), Methods of soil analysis, part 2, 2nd edition. Am Soc Agron, Madison, WI, pp. 595-624.

Cameron, K. C., Di, H. J. and McLaren, R. G., 1997. Is soil an appropriate dumping ground for our wastes? Australian Journal of Soil Research, 35: 995- 1035.

Cannell, M. G. R., Millne, R., Sheppard, L. J. and Unsworth, M. H., 1987. Radiation interception and productivity of willow. Journal of Applied Ecology, 24: 261-78.

Carey, P. L., Rate, A. W. and Cameron, K. C., 1997. Fate of nitrogen in pig slurry applied to a New Zealand pasture soil. Australian Journal of Soil Research, 35: 941-59.

Christersson, L., 1987. Biomass production by irrigated and fertilized *Salix* clones. Biomass, 12: 83-95.

Clothier, B., Green, S. and Roygard, J., 1999. Modelling sustainability of effluent application to soils in the Manawatu, Hort Research, Palmerston North. Report on Hort Research Contract No. 3622.

Clothier, B. E., Sauer, T. J. and Green, S. R., 1988. The movement of ammonium nitrate into the unsaturated soil during unsteady absorbtion. Soil Science Society of America Journal, 52: 340-344.

Clothier, B. E., Scotter, D. R. and Kerr, J. P., 1977. Water retention in soil underlain by a coarse- textured layer: theory and field application. Soil Science, 123: 392-399.

De Bell, D. D., Cole, T. G. and Whitesell, C. D., 1997. Growth and development of and yield of pure and mixed stands of *Eucalyptus* and *Albizia*. For Sci, 43: 286 -298.

De Willigen, P., 1991. Nitrogen turnover in the soil-crop system; comparison of fourteen simulation models. Fertiliser Research, 27: 141-149.

Di, H. J., Cameron, K. C., Moore, S. and Smith, N. P., 1998a. Nitrate leaching and pasture yields following the application of dairy shed effluent or ammonium fertiliser under spray or flood irrigation: results of a lysimeter study. Soil Use and Management, 14: 209-214.

Di, H. J., Cameron, K. C., Moore, S. and Smith, N. P., 1998b. Nitrate leaching from dairy shed effluent and ammonium fertiliser applied to a free draining pasture soil under spray or flood irrigation. New Zealand Journal of Agricultural Research, 41: 263-270.

Dickmann, D. I., Nguyen, P. V. and Pregitzer, K. S., 1996. Effects of irrigation and coppicing on above ground growth, physiology and fine root dynamics of two field grown hybrid poplar clones. Forest Ecology and Management, 80(1-3): 163-174.

Doorenbos, J. and Pruitt, W. O., 1976. Guidelines for predicting crop water use requirements., FAO irrigation and drainage paper 24, 2nd ed. 156 pp, Rome.

Dunin, F. X. and Aston, A. R., 1984. The development and proving of models of large scale evapotranspiration: An Australian study. Agricultural Water Management, 8: 305-323.

Dye, P. J., 1996. Response of *Eucalyptus grandis* trees to soil water deficits. Tree Physiology, 16: 233-238.

Edgar, J. and Stewart, H. T. L., 1979. Wastewater disposal and reclamation using Eucalyptus and other trees. Prog. Wat. Tech., 11(4/5): 163-173.

Edwards, W. R. N. (1986). Precision weighing lysimetry for trees, using a simplified tared-balance design. Tree Physiology 1: 127-144.

Ericsson, T., 1995. Nutrient dynamics and requirements of forest crops. New Zealand Journal of Forestry Science, 24(2/3): 133-167.

FAO, 1979. Eucalypts for planting. FAO Forestry paper no 11, Food and Agricultural Organisation United Nations, Rome.

Falkiner, R. A. and Smith, C. J., 1997. Changes in soil chemistry in effluent irrigated *Pinus radiata* and *Eucalyptus grandis* plantations. Australian Journal of Soil Research, 35: 131-147.

Feikema, P., Heuperman, A., Morris, J. and Connell, L., 1999. Trees, salt and groundwater: Is this system sustainable? Hydrology and solute concentration under a *Eucalyptus* plantation with a shallow water table. Natural Resource Management, 2(1): 18-22.

Feller, M. C., 1981. Water balances in *Eucalyptus regnans, E. obliqua* and *Pinus radiata* forest in Victoria. Austrailian Forestry., 44: 153-161.

Firestone, M. K., 1982. Biological denitrification. In: F.J. Stevenson (Editor), Nitrogen in Agricultural Soils. American Society of Agronomy, Madison, pp. 289-326.

Forsyth, K. A., 1996. Regional Plans and water quality. In: I.G. Mason (Editor), Tertiary Treatment Options for Dairyshed and Piggery Wastewaters. Massey University, Massey University, Palmerston North, New Zealand, pp. 9-17.

Frederick, D. J., Madgwick, H. A. I., Jurgensen, M. F. and Oliver, G. R., 1984. Dry matter production and nutrient content of 5-year old *Eucalyptus nitens* growing on soil mounds in New Zealand. In: D. C. Grey, A. P. G. Schinau, C. J. Schutz and A. van Laar (Editors), IURFO Symposium on "Site and productivity of Fast-growing plantations", Pretoria and Pietermaritzburg, pp. 589 - 96.

Frederick, D. J., Madgwick, H. A. I., Oliver, G. R. and Jurgensen, M. F., 1985. Dry matter and nutrient accumulation of 8-year-old *Eucalyptus saligna* growing at Taheke Forest. New Zealand Journal of Forestry Science, 15(2): 251-254.

Gash, J. H., 1979. An analytical model of rainfall interception by forests. Quart. J. R. Met. Soc., 105: 43-55.

Green, S. R. and Clothier, B. E., 1999. The rootzone dynamics of soil water uptake by the roots of a mature apple tree. Plant and Soil, 206: 63-79.

Guo, L., 1999. Nutrient cycling in Eucalyptus short rotation forests. PhD Thesis, Massey University, Palmerston North, New Zealand.

Hewitt, A. E. 1993. New Zealand Soil Classification. Landcare Research New Zealand Ltd: Lincoln , New Zealand.

Helvey, J. D. and Patric, J. M., 1965. Canopy and litter interception of rainfall by hardwood of eastern United States. Water Resources Research, 1: 193-206.

Hickey, C. W., Quin, J. M. and Davis Colley, J., 1989. Effluent characteristics of dairy shed oxidation ponds and their impacts on rivers. New Zealand Journal of Marine and Freshwater Research, 23: 569 - 584.

Hopmans, P., Stewart, H. T. L., Flinn, D. W. and Hillman, T. J., 1990. Growth, Biomass production and nutrient accumulation by seven tree species irrigated with municipal effluent at Wodonga, Australia. Forest Ecology and Management, 30: 203-211.

Hutson, J. L. and Wagenet, R. J., 1993. A pragmatic field scale approach for modelling pesticides. Journal of Environmental Quality, 22: 494-499.

Jackson, I. J., 1975. Relationships between rainfall parameters and interception by tropical forest. Journal of Hydrology, 24: 215-238.

Johnsson, H., Bergstrom, L., Jansson, P-E. and Paustian, K., 1987. Simulated nitrogen dynamics and losses in a layered agricultural soil. Agriculture Ecosystems and Environment, 18: 333-356.

Kendell, A., McDonald, A. and Williams, A., 1997. The power of biomass. Chemistry and Industry, 9: 327-366.

Kerr, J. P., 1974. Evaporation from a drying soil, Proceedings Agronomy Society of New Zealand, pp. 30-33.

Kincheff, S. J. and Carter, P. C. S., 1991. Commercial experience with short rotation hardwords, Workshop Proceedings: Renewable energy from woody biomass., Massey University, Palmerston North, New Zealand.

King, D.A., 1993. A model analysis of the influence of root and foliage llocation on forest production and competition between trees. Tree Physiology, 12: 119-135.

Leuning, R. (1990). Modelling stomatal behaviour and photosynthesis of *Eucalyptus grandis*. Australian Journal of Plant Physiology, 17: 159-175.

Linder, S. and Flower-Ellis, J. G. K., 1992. Environmental and physiological constraints to forest yield. In: A. Teller, P. Mathy and J. N. R. Jeffers (Editors), Responses to forest ecosystems to environment changes. Elsevier Applied Science.

Ling, G., 1996. Assessment of nitrate leaching in the unsaturated zone on Oahu, University of Hawaii at Manoa, Honolulu.

Ling, G. and El-Kadi, A. I., 1998. A lumped parameter model for nitrogen transformation in the saturated zone. Water Resources Research,, 34(2): 203-212.

Lodhiyal, L. S., Singh, R. P. and Singh, S. P., 1995. Structure and function of an age series of poplar plantations in central Himilaya I. Dry matter dynamics. Annals of Botany, 76(191-199).

Lowe, H. T., 1994. Utilisation of Short Rotation Forestry for On-Site Fuel Wood Grown as Part of an Effluent Disposal Scheme. M.Sc Thesis, Massey University, Palmerston North, New Zealand, 159 pp.

Lloyd, C. R., Gash, J. H. C. and Shuttleworth, W. J., 1988. The measurement and modelling of rainfall interception by an Amazonian rain forest. Agricultural and Forest Meteorology, 43: 277-294.

Lu, Y. 1997. The effect of dairy, piggery and wool scour effluents on willow growth and the soil characteristics. M.Sc thesis, Department of Soil Science, Massey University, Palmerston North, New Zealand.

Mackie-Dawson, L. A. and Atkinson, D., 1991. Methodology for the study of roots in field experiments and the interpretation of results. In: D. Atkinson (Editor), Plant root growth: An ecological perspective. Blackwell Scientific Publications, Cambridge, pp. 25-47.

Madgwick, H. A. I., Beets, P. and Gallagher, S., 1981. Dry matter accumulation, nutirent and energy content of the above ground portion of 4 year old stands of *Eucalyptus nitens* and *E. Fastigata.* New Zealand Journal of Foresry Science, 11(1): 53-59.

Mahimairaja, S., Bolan, N. S., Hedley, M. J. and Macgregor, A. N., 1990. Evaluation of methods of measurement of nitrogen in poultry and animal manures. Fertiliser Research, 24: 141-148.

Markus, D.K., McKinnon, J. P. and Buccafuri, A. F., 1985. Automated analysis of nitrite, nitrate and ammonium nitrogen in soils. Soil Science Society of America Journal, 49: 1208-1215.

Miller, H.G., 1984. Forest fertilisation: some guiding concepts. Forestry, 54(2): 157-167.

Miller, J. T., Cannon, P.G. and Ecroyd, C. E. 1992. Introduced forest trees in New Zealand: recognition, role, and seed source. 11.*Eucalyptus nitens* (Deane et Maiden) Maiden. New Zealand Forest Research Institute, Rotorua, New Zealand. FRI Bulletin no 124.

Ministry of Health, 1995. Drinking water standards for New Zealand. MoH, Wellington pp 87.

Minitab, 1998. Minitab for Windows 95, release 12.1. Minitab Inc.

Monnett, G. T., Reneau, R. B. and Hagedorn, C., 1995. Effects of domestic wastewater spray irrigation on denitrification rates. Journal of Environmental Quality, 24: 940-946.

Morris, J. D. and Wehner, B. A., 1987. Daily and annual water use by four Eucalypt species irrigated with wastewater at Robinvale. Research Rep. No. 329, 31 p, Department of Conservation, Forests and Lands, Victoria.

Myers, B. J., 1992. Effluent Loading rates for irrigated plantations - a water balance model. Australian Forestry, 55: 39-47.

Myers, B. J. Benyon, R. G. Theiveyanathan, S., Criddle, R. S., Smith C. J., Falkiner, R. A., 1998. Response of Effluent irrigated *Eucalyptus grandis* and *Pinus radiata* to vapor pressure deficits. Tree Physiology, 18: 565 - 573.

Myers, B. J., Bond, W. J., Falkiner, R. A., O 'Brien, N., Polglase, P. J., Smith, C. J., Theiveyanathan, S. 1994. Wagga Effluent Plantation Project Technical Report. 17, CSIRO. CSIRO, Australia, Division of Forestry, User Series.

Myers, B. J., Bond, W., Falkiner, R., O'Brien, N., Polglase, P., Smith, C., Theiveyanathan, S., 1995. Effluent Irrigated Plantations: Design and Management, CSIRO Division of Forestry, Canberra, Australia. Technical paper No. 2.

Myers, B. J., Polglase, P. J., Benyon, R. G., Bond, W. J., Falkiner, R. A., O 'Brien, N. D., Smith, C. J., Snow, V. O., Theiveyanathan, S. 1997. Environmentally-sound management of water and nutrients in Australian effluent-irrigated plantations, The forest alternative symposium: principles and practice of residual use, Seattle, Washington,.

Myers, B. J. and Talsma, T., 1992. Site water balance and tree water status in irrigated and fertilised stands of *Pinus radiata*. Forest Ecology and Management, 52: 17–42.

Myers, B. J., Theiveyanathan, S., O'Brien, N. D. and Bond, W. J., 1996. Growth and water use of effluent-irrigated *Eucalyptus grandis* and *Pinus radiata* plantations. Tree Physiology, 16: 211-219.

Myrold, D. D. and Tiedje J. M., , 1985. Establishment of denitrification capacity in soil: effects of carbon, nitrate and moisture. Soil Biology and Biochemistry, 17: 819-822.

Nelson, D. W. and Sommers, L. E., 1980. Total nitrogen analysis of soil and plant tissues. J. Assoc. of Anal. Chem., 63: 770-778.

Nicholas, I. D., Carnus, J. M., Oliver, G. R., 1997. Comparative performance of tree species in New Zealand wastewater irrigation systems. In: H. Wang and J-M. Carnus (Editors), New Zealand Waste Water Association and New Zealand Land Treatment

Collective Proceedings of Land Treatment and Wetland Workshop. Rotorua. Forest Research Institute, Rotorua, New Zealand, pp. 44-52.

Nicholas, I. D., 1997. Hardwoods in land treatment schemes current status of New Zealand and Australian research. In: H. Wang and J-M. Carnus (Editors), Land Application of Biosolids, New Zealand Land Treatment Collective Proceedings of Technical Session No 15, Nelson. Forest Research Institute, Rotorua, New Zealand. pp. 34-45.

Nilsson, L-O., 1993. Carbon sequestration in Norway spruce in south of Sweden as influenced by air pollution, water availability and fertilization. W.A.S.P, 70: 177-86.

Nilsson, L-O. and Ericsson, T., 1986. Influence of shoot age and nutrient uptake pattern in a willow plantation. Canadian Journal of Forest Research, 16: 185-190.

Pereria, J. S., Linder, S., Araujo, M. C., Pereira, H. and Erricsson, T., 1994. Biomass production with optimised nutrition in *Eucalyptus globoulus* plantations. In: J. S. Pereira (Editor), Eucalyptus for Biomass Production. The State-of-the-Art. CEC, Brussels, pp. 101-121.

Pereria, J. S., Linder, S., Araujo, M. C., Pereira, H. and Erricsson, T., 1989. Optimisation of biomass production in *Eucalyptus globulus* plantations. In: J. S. Pereira and J. J. Landsberg (Editors), Biomass productivity by fast growing trees. Kluwer Academic Publishers, Dordrecht, The Netherlands, pp. 101-121.

Perry, D. A. and Maghembe, J., 1989. Ecosystem concepts and current trends in forest management: time for reappraisal. Forest Ecology and Management, 26: 123-140.

Polgase, P. J., Tompkins, D., Stewart, L. G. and Falkiner, R. A., 1995. Mineralization and leaching of nitrogen in an effluent irrigated pine plantation. Journal of Environmental Quality, 24: 911-920.

Ranney, J. W., Wright, L. L. and Layton, P. A. 1987. Hard wood energy crops: the technology of intensive culture. Journal of Forestry, 85: 17-28.

Resource Management Act, 1991, GPH Print.

Roberts, A. H. C., O Connor, M. B. and Longhurst, R. D., 1992. Wastes as plant nutrient sources: issues and options. In: P. E. H. Gregg and L. D. Currie (Editors), The use of wastes and by-products as fertilisers and soil amendments for pastures and crops. Occasional Report No 6 Fertiliser and Lime Research Center, Massey University, Palmerston North, pp. 44-55.

Roygard, J. K. F., Green, S. R., Clothier, B. E., Sims, R. E. H. and Bolan, N. S., 1999. Short rotation forestry for land treatment of effluent - A lysimeter study. Australian Journal of Soil Research. 37 (5) in press.

Roygard, J.K.F and Vogeler, I., 1999. Unpublished data, HortResearch, Palmerston North, New Zealand

Rutter, A. J., 1975. Chapter 4: The hydrological cycle in vegetation. In: J. L. Monteith (Editor), Vegetation and the atmosphere, pp. 111-154.

Rutter, A. J., Kershaw, K. A., Robins, P. C. and Morton, A. J., 1971. A predictive model of rainfall interception in forests, 1. Derivation of the model from observations in a plantation of Corisican pine. Agric. Meteorol., 9: 367 - 384.

Rutter, A. J. and Morton, A. J., 1977. A predictive model of rainfall interception in forests III. Sensitivity of the model to stand parameters and meteorological variables. Journal Applied Ecology, 14: 567-588.

Ruz-Jurez, B. E., White, R. E. and Ball, P. R., 1994. Long-term measurement of denitrification in three contrasting pastures grazed by sheep. Soil Biology and Biochemistry, 26(1): 29-39.

Rytter, R. M. and Hansson, A. C., 1996. Seasonal amount, growth and depth distribution of fine roots in an irrigated and fertilised *Salix viminalis* L. plantation. Biomass and Bioenergy, 11(2/3): 129-137.

Sasse, J. and Sands, R., 1996. Comparative responses of cuttings and seedlings of *Eucalyptus gobulus* to water stress. Tree Physiology, 16: 287-294.

Schilke-Gartley, K. L. and Sims, J. T., 1993. Ammonia volatilization from poultry manure-amended soils. Biology and Fertility of soils, 16: 5-10.

Schofield, N. J. and Bari, M. A., 1991. Valley reforestation to lower saline groundwater tables: results from Stene's Farm. Western Australia'. Australian Journal of Soil Research, 29: 635-650.

Senelwa, K. A., 1997. The air gasification of woody biomass from short rotation forests: Opportunities for small scale biomass-electricity systems. PhD Thesis, Massey University, Palmerston North, New Zealand, 265 pp.

Sexstone, A. J., Parkin, T. B. and Tiede, J. M., 1985. Temporal response of soil denitrification rates to rainfall and irrigation. Soil Science society of America Journal, 49(99-103).

Sharma, M. L., 1984. Evapotranspiration from a Eucalyptus community. Agricultural Water Management, 8: 41-45.

Silva, R. G., Cameron, K. C., Di, H. J. and Hendry, T., 1999. A lysimeter study of the impact of cow urine, dairy shed effluent, and nitrogen fertiliser on leaching. Australian Journal of Soil Research, 37: 357-369.

Sims, R. E. H., Hodson, R. W. and Watters, M. P., 1994. Root production of short rotation coppice Eucalyptus grown under an effluent disposal system. In: P. H. Chartier, A. A. C. M. Beenakers and G. Grassi (Editors), Biomass for energy environment agriculture and industry volume 1, 8th E. C. Conference.

Sims, R. E. H. and Riddell-Black, D., 1998. Sustainable production of short rotation forest biomass crops using aqueous waste management systems. Biomass and Bioenergy, 15(1): 75-81.

Sisson, J. B., Ferguson, A. H, and Van Genuchten, M. T., 1980. Simple method for predicting drainage from field plots. Soil Science Society of America Journal. 44: 1147-1152.

Slatyer, R. O. (1967). Plant water relationships. New York, Academic Press.

Smith, C. J., Freney, J. R. and Bond, W. J., 1996. Ammonia volatilization from soil irrigated with urban sewage effluent. Australian Journal of Soil Research, 34: 789-802.

Smith, M. K., 1974. Throughfall, stemflow and interception in pine and Eucalypt forests. Australian Forestry, 36: 190-197.

Smucker, A. J. M., McBurney, S. L. and Srivistava, A. K., 1982. Quantitative separation of roots from compacted soil profiles by the hydropnuematic elutriation system. Agronomy Journal, 74: 500 - 503.

Snowdon, P. and Benson, M. L., 1992. Effects of combinations of irrigation and fertilisation on the growth and above-ground biomass production of *Pinus radiata*. Forest Ecology and Management, 52: 87-116.

Sprosen, M. S., Ledgard, S. F. and Thom, E. R., 1997. Nitrate leaching is similar in N2-fixing grass clover pasture and N-fertilised grass only pasture at similar N inputs., Proccedings of the New Zealand Grassland Association, Mangere, pp. 125-128.

Stanley, W. G. and Montagnini, F., 1999. Biomass and nutrient accumulation in pure and mixed plantations of indigenous tree species grown on poor soils in the humid tropics of Costa Rica. Forest Ecology and Management, 113: 91-103.

Sun, D. and Dickinson, G., 1993. Responses to salt stress of 16 *Eucalyptus* species, *Grevillea robusta*, *Lophostemon confertus* and *pinus caribaea* var. *hondurensis*. Forest Ecology and Management, 60: 1-14.

Teklehaimanot, Z., and Jarvis, P. G., 1991. Direct measurement of evaporation of intercepted water from forest canopies. Journal of Applied Ecology 28, 603 – 618.

Thornley, J. H. M. and Cannell, M. G. R., 1992. Nitrogen relations in a forest plantation - soil organic matter ecosystem model. Annals of Botany, 70: 137-151.

Topp, G. C., and Davis, J. L., and Annan, A. P., 1982. Electromagnetic determination of soil water content using TDR: I. Applications to wetting fronts and steep gradients. Soil Science Society of America Journal, 46, 672-678.

Tomer, M. D., Schipper, L. A., Knowles, S. F., Rijkse, W.C., McMahon, S. D., Smith, C. T., Thorn, A., Charleson, T., 1997. A land -based system for treatment of municipal wastewater at Whakarewarewa Forest, New Zealand. Characterisation of soil, plant,

groundwater, and wetland components., New Zealand Forest Research Institute, Rotorua. FRI Bulletin 199.

Tungcul, R. O., Sims, R. E. H. and Mason, I. G., 1996. Response of short rotation forestry to dairy farm pond effluent irrigation., Proceedings of the 7th National Bioenergy Conference, South Eastern USA Regional Biomass Program. 15 -20 September, Nashville Tennessee. pp. 793 -799.

Vanclooster, M., Viane, P., Diels, J. and Christians, K., 1994. WAVE, a mathematical model for simulating water and agrochemicals in the soil environment. Reference and Users Manual, Release 2.0. Institute for Land and Water Mangement, Katholieke University, Lueven, Belgium.

van Genuchten, M. Th., 1981. Analytical solutions for the movement of solutes. Journal of Hydrology, 49:213-233.

van Kraayenoord, C. W. S., Slui, B. and Knowles, F. B., 1995. Introduced forest trees in New Zealand: recognition, role, and seed source. 15. The willows *Salix* species. New Zealand Forest Research Institute, Rotorua, New Zealand. FRI Bulletin no 124.

Watt, J., McIndoe, I., Green, S. R., 1998. Crop Water Requirements for irrigation in the Waikato Region. Landcare Research Contract Report: LC9798/ 021.

Weatherburn, M. W., 1967. Phenol-Hypoclorite reaction for determination of nitrogen. Anal. Chem., 39: 971-974.

Welbank, P. J. and Williams, E. D., 1968. Root growth of a barley crop estimated by sampling with portable powered soil-coring equipment. Journal of Applied Ecology, 5: 477-481.

White, D. A., 1999. Quantifying the effect of cumulative water stress on stomatal conductance of *Eucalyptus globulus* and *Eucalyptus nitens*: a phenomenological approach. Australian Journal of Plant Physiology, 26: 17-27.

Young, H. E. and Carpenter, P. N., 1975. Sampling variation of nutrient element content within and between trees of the same species. In: H. E. Young (Editor), Olso biomass studies, International Union of Forest Research Organisations. S4.01 mensuration, growth and yield, working party on the mensuration of the forest biomass. College of Life Sciences and Agriculture, University of Maine at Orono, Maine, USA.

Zinke, P. J., 1967. Forest interception studies in the United States. In: W. E. Sopper and H. W. Lull (Editors), Forest Hydrology. Pergamon, Oxford, pp. 137-161.

APPENDICES

Appendix A

Mathematical description of the model

Provided by the LPM code writer, Steve Green, (Environment and Risk Management Group, HortResearch, Palmerston North). From Clothier, Green and Roygard (1999).

Model Equations and Example Applications

The following procedure is used to calculate the soil water and nitrogen balance, and the corresponding drainage fluxes quitting the base of the root zone.

Step 1: Root-zone Water Balance

The total amount of stored water in the soil, $S = z_R\theta$, is a dynamic property of the root-zone. It changes daily to reflect the 'inputs' of rainfall and irrigation that are received, and the 'outputs' or 'losses' of water due to plant uptake, surface evaporation and runoff, plus any deep drainage of water beyond the root-zone. We calculate a water balance for the root-zone, by summing all the inputs and outputs of water, *viz.*

$$\Delta S = (I + P) - (R + D + ET) \qquad [1]$$

where ΔS [mm] represents the change in the total amount of water stored in the root zone soil. Inputs to this store are; the total amount of irrigation, I [mm], and the total amount of precipitation, P [mm]. Some of the surface-applied irrigation, and/or precipitation, may be lost as surface runoff, R [mm], and some may eventually be lost due to deep drainage, D [mm], beyond the root-zone. The other component of this water balance is the evapo-transpiration, ET [mm]. This critical term represents the amount water removed from the soil by the roots, to be lost evaporatively as transpiration, plus that lost as surface evaporation of water directly from the soil.

Step 2: Plant water use (*ET*)

Plant water use depends on both the ambient meteorological conditions and physiological stage of plant development. A two-step procedure is used to calculate plant water use, based on guidelines given by the Food and Agriculture Administration (FAO) of the United Nations (Doorenbos and Pruitt, 1977). Measured data for global radiation from the sun, air temperature, humidity and windspeed were used to calculate a reference evaporation rate, ET_o [mm d^{-1}]. From the modified Penman equation, we obtain

$$ET_0 = \left(\frac{s}{s+\gamma}\right)R_N + \left(\frac{\gamma\, f(U)}{s+\gamma}\right)D_A \qquad [2]$$

where R_n [mm d^{-1}] is the net radiation is expressed in units of an equivalent evaporation rate, D_a [kPa] is the difference between the saturation vapour pressure at mean air temperature and the mean actual vapour pressure of the air, s [Pa $^oC^{-1}$] is the slope of the saturation vapour-pressure versus temperature curve, γ [66.1 Pa] is the psychrometeric constant, and f(U) is a wind-related function given by

$$f(U) = 2.7(1 + U/100). \qquad [3]$$

Here U [km d^{-1}] is the 24-hr wind run at 2-m height. This reference value, ET_o, defines the rate of evaporation expected from an extensive surface of green grass cover of short, uniform height, actively growing, completely shading the ground, and not short of water.

To account for the effect of plant physiological characteristic, a crop coefficient, K_c, is used to relate the reference evaporation rate, ET_o, to the actual crop water use, ET. For routine calculations of crop evapotranspiration, the following equation can be used:

$$ET = K_C . ET_O \qquad [4]$$

where K_c is a dimensionless number that normally varies between about 0.2 and 1.1. The particular value of the crop coefficient, K_c, determines the evapotranspiration of a disease-free crop grown in a large field under optimum soil water and fertility conditions and achieving full production potential under a given growing environment. In other words it defines the maximum rate of water use expected from a particular crop. Various factors affect the value of K_c, including crop characteristics, crop planting or sowing dates, rate of crop development, length of growing season and climatic conditions. K_c is set equal to a maximum value by the user, but this is reduced when the plants were under water and nutrient stress, as described below.

Step 3: Drainage of water

The drainage of water below the bottom of the root zone, D, was calculated using the analytical solution of Sisson et al. (1980). The method is relatively simple and is based on the soil's hydraulic properties. The vertical water movement through a uniform soil is described by Richards' equation

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left[K \frac{\partial H}{\partial z}\right] \qquad [5]$$

where θ is the volumetric water content [$m^3\ m^{-3}$], K is the hydraulic conductivity [$m\ s^{-1}$], H is the total hydraulic head [m] which comprises the soil's pressure head, h [m], minus the total gravitational head, z [m], and t is the time [s]. An analytical solution to Eq. [5] can be derived by choosing a suitable $K(\theta)$ relation and assuming a 'unit gradient' in the total potential head, i.e. $dH/dz=-1$, which commonly occurs when a uniform soil profile is draining freely only under the influence of gravity.

Sisson et al. (1980) used three published $K(\theta)$ relationships to generate three analytical solutions for the drainage phase following an infiltration event. Here we consider just the solution based on a power-law hydraulic conductivity function of the form

$$K(\theta) = K_m\left(\frac{\theta}{\theta_S}\right)^{1/\beta} \qquad [6]$$

where K_m and θ_m represent maximum values of the hydraulic conductivity and soil water content, respectively, and β is a slope factor which dictates how fast the hydraulic conductivity declines as the soil dries. The 'unit-gradient' approximation for a draining soil profile yields a linear relationship between the total depth of water, S [m], and the log of time [t], which is given by

$$S = C - Y \log_e(t) \qquad [7]$$

where both Y and C are constants and depend on particular values of K_m, θ_m and β. It follows from our use of a power-law conductivity function, that

$$Y = \beta/(\beta-1), C = (1-\beta)\theta_M z\left[\frac{A}{z}\right]^Y, \text{ with } A = \left[\frac{K_M}{\beta\theta_S}\right] \qquad [8]$$

For practical purposes we are interested in the total amount of water stored in the root zone, and so we can set z equal to the depth of the roots, which we will call z_R [m]. It follows

from Eq. [8] that the amount of drainage, D [m], occurring in one day from an initial storage value, S_I [m], will be equal to

$$D = Y \ln[1 + \exp[(S_1 - C)/Y]]. \quad [9]$$

The 'lumped parameters' needed to estimate the drainage of water quitting the root zone are the two constants Y and C, as defined above, and the initial storage value, S_I. This equals the total depth of water stored in the soil profile, from the soil surface to a depth of z_R.

Plant uptake of N

This component of the model is based on a simple carbon and nitrogen balance describing growth in terms of light interception, and nitrogen uptake in terms of carbon allocation and tissue nitrogen concentrations. The main determinant of the N-uptake is the daily biomass production per unit ground area, G [kg/m^2/d]. This is calculated from

$$G = \varepsilon \Phi \quad [10]$$

where Φ [MJ/m^2/d] is the daily photosynthetically active radiation (PAR) intercepted by the foilage and ε [kg/MJ], is a conversion efficiency which varies with the amount of N available to the plants. Foliage light interception is expressed in terms of Beer's law as

$$\Phi = \Phi_O\left(1 - \exp(-k\sigma F\right) \quad [11]$$

where Φo [MJ/m^2/d] is the incident global radiation, F [kg leaf/m^2 ground] is the foliage mass density, σ [m^2 leaf/kg leaf] is the specific leaf area, and k [=0.5] is the extinction coefficient.

We consider the conversion efficiency for dry-matter production to be related to the amount of soil-nitrogen, N [mg/m2], available to the plants. We use the following quadratic relationship that is consistent with the general relationships between growth and foliar nutrient concentration determined by physiologists (King, 1993):

$$\varepsilon = \frac{2N}{N_S}\left(1 - \frac{N}{2N_S}\right)\varepsilon_o \quad [12]$$

if $N < N_S$ and $\varepsilon = \varepsilon o$ if $N > N_S$. We define N_S to be a threshold value of soil-N above which maximum productivity can be achieved, and εo is the maximum possible value of ε. Here we consider Ns to be the value of soil-N that would supply 2 weeks of uptake at the maximum rate.

Net production of above and below ground biomass

Plant biomass is expressed in terms of the growth and senescence of the plant organs. For each plant organ we write out a mass balance equation that considers:

- inputs of DM due to carbon allocation
- losses of DM as the plants senesce, and
- removal of DM at harvest (or thinning)

The total mass of foliage, F [kg/m^2] is calculated from

$$\frac{dF}{dt} = n_F G - \left(\gamma_F F + H_F\right), \quad [13]$$

the total mass of roots, R [kg/m2], is calculated from

$$\frac{dR}{dt} = n_R G - \left(\gamma_R R + \rho H_F\right), \quad [14]$$

and the total mass of woody tissue (stemwood, branches and secondary roots), W [kg/m2], is calculated from

$$\frac{dW}{dt} = n_W G - (\gamma_W W + H_W), \qquad [15]$$

where n is the fraction of biomass partitioned to the foliage, the fine roots and the woody tissue, respectively, γ is the rate of senescence, H is the harvest fraction, and ρ is the ratio of root-mass to foliage mass. Seasonal changes in allocation and senescence were not included in the model because we were concerned with the long term consequences of these allocation patterns. Also, we assume the fine root mass adjusts after harvest to maintain an optimum ratio ρ.

The model assumes that plant growth will achieve the maximum potential only if soil water and soil nitrogen (NO_3^- and NH_4^+) are non-limiting. The net uptake of nitrogen from the soil is set equal to the amount of nitrogen incorporated into the new biomass, minus the fraction of nitrogen that has been retranslocated, λ, from the old or senescing tissues. Assuming a constant nitrogen concentration [N] for each plant organ, then the potential uptake of nitrogen from the soil, U [kg/ha/d] is defined as

$$U = (n_F G - \lambda_F \gamma_F F)[N]_F + (n_W G - \lambda_W \gamma_W W)[N]_W + (n_R G - \lambda_R \gamma_R R)[N]_R \qquad [16]$$

This uptake requirement can be met only if sufficient nitrogen exists in the soil. We assume a linear reduction in U whenever the total soil nitrogen available within the root zone falls below the threshold value of Ns which is defined above.

Carbon and Nitrogen dynamics of the soil organic matter

The decomposition of soil biomass adds to the amount of mineral nitrogen in the soil. This process is known as mineralisation. Mineralisation is modelled by dividing the soil organic matter into two pools – a fast cycling litter pool and an almost stable humus pool following Johnsson et al. (1987). This two-pool model then considers the amount of soil carbon and soil nitrogen that cycle within soil organic material. The relative amounts of these two components changes daily to reflect inputs of new biomass and losses of older biomass as it decomposes. The nitrogen demand for the internal cycling of soil-C and soil-N is regulated by the C/N ratio of the soil biomass, r_O, which is one of the model inputs.

Decomposition of soil litter carbon (C_L) is a function of a specific rate constant (K_L) which is influenced by temperature and soil moisture. The products of decomposition are CO_2, stabilized organic material (humus) and, conceptually, microbial biomass and metabolites. The relative amounts of these products is determined by a synthesis efficiency constant (f_E) and a humification fraction (f_H). The following mass balance equations, which represent the inputs minus the outputs of soil-C and soil-N, form the basis to model the turnover of carbon and nitrogen in the litter pool:

$$\frac{\partial C_L}{\partial t} = \left[(1 - f_H) f_E - 1\right] . K_L . C_L + F_{C,L} \qquad [17]$$

$$\frac{\partial N_L}{\partial t} = \left[(1 - f_H) f_E \frac{1}{r_O} - \frac{N_L}{C_L}\right] . K_L . C_L + F_{N,L} \qquad [18]$$

where F represents the amount of fresh organic matter which is added to the soil biomass. During harvest we assume 10% of the fresh organic matter goes into the litter pool while the remaining 90% is added to the humus pool.

A similar set of mass balance equations are used to describe the turn-over of carbon and nitrogen in the the humus pool:

$$\frac{\partial C_H}{\partial t} = f_E . f_H . K_L . C_L - K_H . C_H + F_{C,H} \quad [19]$$

$$\frac{\partial N_H}{\partial t} = \frac{f_E . f_H}{r_O} K_L . C_L - K_H . N_H + F_{N,H} \quad [20]$$

Decomposition of soil humus (C_H) is assumed to follow first-order kinetics with a specific rate constant (K_H) which depends on temperature and soil moisture. The other terms in these mass balance equations have already been described above.

All carbon and nitrogen turn-over reactions can result in a net production (mineralisation) or a net consumption (immobilisation) of ammonium, depending on the C/N ratio of the biomass, r_O, in the two pools. From a consideration of mass balances, any increase in NH_4^+-N, due to mineralisation, must be equal the decrease in organic-N from the two organic matter pools. Thus, we solve the following mass-balance equation for mineralisation

$$\frac{\partial NH_4^+}{\partial t} = \left[\frac{N_L}{C_L} - \frac{f_E}{r_O}\right] K_L . C_L + K_H . N_H \quad [21]$$

Mineralisation occurs whenever $\partial NH_4^+/\partial t > 0$, otherwise immobilisation will occur. The model also recognises that, if no ammonium is available for immobilisation, then nitrate can be used according to the following equation:

$$\frac{\partial NO_3^-}{\partial t} = -\frac{f_E}{r_O} K_L . C_L \quad [22]$$

During all simulations reported here we chose typical values for most of the parameters: the rate constants were K_L=0.003 d^{-1} and K_H=0.00015 d^{-1}; constant values were used for the efficiency of carbon turn-over, f_E=0.4, the humification fraction, f_H=0.2, and the C/N ratio of the soil biomass, r_O=10.0, as suggested by Johnnson et al. (1987). Because excess amounts of ammonium were always added in the effluent we found net mineralisation was occurring (and no immobilisation).

Mass balance equations for Urea fertilizer

The lumped parameter model allows for an input of mineral nitrogen in the form of Urea fertilizer. This option was included in the model scenarios in order to simulate a broadcast application of fertilizer e.g. at the start of spring-time. Once the Urea is applied to the soil surface its fate is determined by two processes:

- losses due to the hydrolysis of Urea to ammonium, and
- losses as Urea leaches beyond the root zone.

The total mass of Urea, M_U [mg/m2], in the root zone to a depth of z_R [mm] is found by solving the following mass balance equation:

$$\frac{dM_U}{dt} = z_R \frac{d\theta U}{dt} = (I_1 X_1) - (k_1 z_R \theta U + D.U) \quad [23]$$

where U [mg/L] is the concentration of Urea in soil solution, $I_L X_I$ [mg/m2] is the total mass of Urea added as fertilizer, and k_1 [1/d] is a rate constant to describe the hydrolysis of Urea to Ammonium, and $D.U$ [mg/m^2/d] represents the drainage of Urea beyond the root zone.

Mass balance equations for Ammonium

The lumped parameter model allows for the input of ammonium in the irrigation water, as would mimic the land based disposal of effluent. Once the ammonium is applied to the soil surface its fate is determined by seven competing processes:

- inputs from the hydrolysis of Urea
- inputs from the mineralisation of soil biomass
- retardation due to the adsorption of ammonium to the soil particles
- losses due to the volatilisation of ammonia gas
- losses due to the nitrification of ammonium into nitrate
- losses due to the drainage of ammonium below the root zone
- losses due to plant uptake

The total mass of Ammonium, M_A [mg/m^2], in the root zone to a depth of z_R [mm] is found by solving the following mass balance equation:

$$\frac{dM_A}{dt} = z_R \frac{d\theta\, R_A A}{dt} = \left(I_I X_I + k_1 z_R \theta\, U + S_M\right) - \left(z_R \varepsilon\, k_V A + k_2 z_R \theta\, A + P_A + DA\right) \quad [24]$$

where A [mg/L] is the concentration of ammonium in soil solution, $I_L X_I$ [mg/m^2] is the total mass of ammonium added in the effluent, S_M [mg/m^2] is rate of mineralisation, $\varepsilon = (\theta s - \theta)$ [m^3/m^3] is the gas-filled porosity, k_V [1/d] is Henry's constant for volatilisation, P_A [mg/m^2/d] is the rate of plant uptake, k_2 [1/d] is the a rate constant to describe the nitrification of ammonium to nitrate, and $D.A$ [mg/m^2/d] represents the drainage of ammonium beyond the root zone.

Mass balance equations for Nitrate

The lumped parameter model allows for the input of nitrate in the irrigation water, as would mimic the land based disposal of effluent. Once the nitrate is applied to the soil surface its fate is determined by the following five processes:

- Inputs of nitrate from effluent application,
- Inputs from the nitrification of Ammonium
- Retardation due to the adsorption of nitrate (= 0 in Manawatu soil)
- Losses due to immobilisation (= 0 here)
- Losses from denitrification,
- Losses due to plant uptake
- Losses due to the drainage of nitrogen beyond the root zone.

The total mass of nitrogen, M_N [mg/m^2], in the root zone to a depth of z_R [mm] is found by solving the following mass balance equation:

$$\frac{dM_N}{dt} = z_R \frac{d\theta\, R_N N}{dt} = \left(I_I X_I + k_2 A\right) - \left(S_M + k_3 z_R \theta\, N + P_N + DN\right) \quad [25]$$

where N [mg/L] is the concentration of nitrate in soil solution, $I_L X_N$ [mg/m^2] is the total mass of ammonium added in the effluent, S_M [mg/m^2] is rate of immobilisation, k_3 [1/d] is

a rate constant to describe nitrification losses, P_N [mg/m^2/d] is the rate of plant uptake, and $D.N$ [mg/m^2/d] represents the drainage of water beyond the root zone.

The above mass-balance equations are solved analytically, using Laplace transforms, to generate a mathematical expression for the average concentration of urea, ammonium and nitrate within the rootzone. Our simple approach is similar to model nitrogen transformation in soil that was recently developed by Ling and El-Kadi (1998).

Appendix B

Parameter files used for model simulations

B.1 Parameter file for simulation of the bare soil lysimeter treatment

```
c ------------------------------------------------------------
c ...   Simulation of the bare soil lysimeter
c ...   A lumped parameter model to calculate the leaching of fertilizer
c ...   beyond the root zone of a crop.
c ...   A reference ET is calculated for a given crop, based on crop
c ...   factors determined from planting dates etc.
c ...   Drainage is based on Sisson et al (1980)
c ...   N transformations based on Ling and El-Kadi (1998)
c ------------------------------------------------------------
c ...   Specify the ET model:
c ...   IF ET model = 1 THEN
c ...   read in Rg, Ta, Rf and calc ET
c ...   IF ET model = 2 THEN
c ...   read in Rg, Ta, Rf, Ir, Tx, Tn, Tw, Ws and calc LE
c ------------------------------------------------------------
   2     ! ET model: <1> LE = à(s/s+ç)Rg ; <2> LE = (sRg + sçDa.ra)/(s+ç)
   1.0   ! effective rainfall coefficient
   2.5   ! et max for soil evaporation
c ------------------------------------------------------------
c ... Specify the annual cycle of Kc - the crop coefficient
c
c     K2                  /:~~~~~~~~~~~~~~:\
c                        /  :              :  \
c     K1     ___________/    :              :    \___________
c            :          :    :              :    :          :
c           T1         T2    T3             T4   T5         T6
c
c ------------------------------------------------------------
   12         !  no of Kc values (e.g. 6 needed for one crop cycle)
  35  0.      ! enter annual cycle of Kc   J   Kc(T1)
  28  0.      !                            F   Kc(T2)
  28  0.      !                            M   Kc(T3)
  35  0.      !                            A   Kc(T4)
  28  0.      !                            M   Kc(T5)
  27  0.      !                            J   Kc(T6)
  28  0.      !                            J   Kc(T7)
  28  0.      !                            A   Kc(T8)
  28  0.      !                            S   Kc(T9)
  28  0.      !                            O   Kc(T10)
  35  0.      !                            N   Kc(T11)
  37  0.      ! make sure we have 365 values of Kc (T12) D
c ------------------------------------------------------------
c ... Specify crop factors
c ------------------------------------------------------------
  1.0         ! root depth Zr [m]
  0.50        ! drought tolerance (as a fraction of available water)
  0.000       ! light utilization efficiency [g-DM/MJ]
c ------------------------------------------------------------
c ... Define plant N-uptake parameters (a logistic fn)
c ------------------------------------------------------------
  0.0013      ! GF = SENESCENCE RATE OF LEAVES
  0.0003      ! GS = SENESCENCE RATE OF STEM
```

```
  0.0055      ! GR = SENESCENCE RATE OF ROOTS
  0.0         ! NF = DRY MATTER ALLOCATION TO LEAVES
  0.0         ! NS = DRY MATTER ALLOCATION TO STEM
  0.0         ! NR = DRY MATTER ALLOCATION TO ROOTS
  0.50        ! LF = FRACTION OF LEAF-N RECYCLED
  0.50        ! LS = FRACTION OF STEM-N RECYCLED
  0.50        ! LR = FRACTION OF ROOT-N RECYCLED
  0.0000      ! SLA = SPECIFIC LEAF AREA [ha-leaf/kg-DM]
  0.030       ! NCFS = OPTIMUM N CONTENT OF LEAF [kg-N/kg-DM]
  10.00       ! NCRIT = SOIL-N [kg-N/ha] REQUIRED FOR GROWTH
  00000.      ! HMFT = HARVEST ABOVE GROUND DM WHEN IT IS > HMFT [KG/HA]
  0000.       ! HMFC = MINIMUM LEAF-DM [KG/HA]
  0.99        ! HFST = FRACTION OF STEM/BRANCH REMOVED DURING HARVEST
  0.10        ! HFDM = FRACTION OF HARVESTED DM RETURNED AS LITTER
  26.0        ! LAST DATE OF THINNING [YEARS]
c -------------------------------------------------------------
c ... Specify SOIL factors
c -------------------------------------------------------------
  0.44        ! ém    maximum water content [m3/m3]
  0.390       ! éf    field capacity [0.2 bar]
  0.08        ! éc    wilting point [15.0 bar]
  750.0       ! Km    saturated hydraulic conductivity [mm/d]
  0.0293      ! á     beta parameter        [-]
  1.2600      ! pb    soil bulk density     [kg/L]
c -------------------------------------------------------------
c ... Specify soil moisture and temperature response factors
c -------------------------------------------------------------
  0.60        ! es    saturation activity
  0.10        ! é1    increasing interval
  0.08        ! é2    decreasing interval
  3.0         ! QTEN factor
  20.0        ! Tb base temperature
  0.06        ! éd to reduce denitrification
  2.0         ! d factor
c -------------------------------------------------------------
c ... Specify the solute transport factors
c -------------------------------------------------------------
  0.000       ! k1 rate constant hydrolysis of Urea           [day-1]
  0.200       ! k2 rate constant nitrification of Ammonia     [day-1]
  0.006       ! k3 rate constant denitrification of Nitrate   [day-1]
  5.000       ! Kda distribution coefficient for Ammonium     [L/Kg]
  0.000       ! Kdn distribution coefficient for Nitrate      [L/Kg]
  0.50        ! Kh volatilization constant for Ammonia        [day-1]
c -------------------------------------------------------------
C ... Specify nitrogen mineralization parameters
c -------------------------------------------------------------
  20.0        ! ro = c/n ratio of the soil biomass
   0.4        ! fe = carbon turnover efficiency [d-1]
   0.2        ! fh = humification coefficient
   0.000      ! Kman = decomposition rate for soil manure
   0.008      ! Klit = decomposotion rate for soil litter
   0.00007    ! Khum = decomposition rate for soil humus
c -------------------------------------------------------------
c ... specify annual C/N returns to system (mainly as leaf fall)
c -------------------------------------------------------------
   000.0      ! ignore
     0.5      ! carbon ratio in organic matter (OM)
    20.0      ! C/N ratio of OM
     0.9      ! humus fraction, litter = 1.0-humus fraction
c -------------------------------------------------------------
c ... Specify the initial water and solute conditions
```

```
c -------------------------------------------------------------
  0.302      ! éi initial water content      [m3/m3]
  0.000      ! Ui initial Urea conc in soil solution [mg/l]
  0.000      ! Ai initial Ammonium conc in soil solution [mg/l]
  20.0       ! Ni initial Nitrate conc in soil solution [mg/l]
  000.       ! Fi initial foliage DM         [kg/ha]
  000.       ! Si initial stem DM            [kg/ha]
  000.       ! Ri initial root DM            [kg/ha]
c -------------------------------------------------------------
c ... Specify initial organic C/N pool values
c -------------------------------------------------------------
     0.0     ! CMAN = CARBON IN MANURE POOL     (kg/ha)
     0.0     ! NMAN = NITROGEN IN MANURE POOL   (kg/ha)
   378.0     ! CLIT = CARBON IN LITTER POOL     (kg/ha)
    18.9     ! NLIT = NITROGEN IN LITTER POOL   (kg/ha)
  3402.0     ! CHUM = CARBON IN HUMUS POOL      (kg/ha)
   170.0     ! NHUM = NITROGEN IN HUMUS POOL    (kg/ha)
c -------------------------------------------------------------
c ... Specify the Irrigation regime (must supply atleast 5 parameters)
c ... <1>means user supplied BELOW
c ... <2>means automatic [X mm]from t(beg)to t(end) at dt intervals [day]
c ... <3> means irrigate [X mm]if wc(i) < [threshold = éc + dtol*(ém-éc)]
c -------------------------------------------------------------
   1         ! Irrigation <1>=as supplied, <2>=automatic, <3>=as needed
   0.0       ! ATMOS inputs of N [kg/ha/y]
   0.0       ! Allow a 10% IR deficit capacity [mm] before applying IR
   20.0      ! IR total    [mm]    [1 mm = 1 L/m2]
   1.0       ! t(beg)      [day]
 10000.0     ! t(end)      [day]
   7.0       ! IR interval [day]
 170.0       ! no effluent from here
 260.0       !             to here
c -------------------------------------------------------------
c ... Specify the Fertilizer regime (must supply atleast 5 parameters)
c ... <1> means as supplied
c ... <2>means automatic (fertigation)from F(beg)to F(end) at F-int [day]
c ... for each application enter day, X-U, X-A, X-N [kg/ha]
c -------------------------------------------------------------
  1           ! Fertilizer type   <1>=wet   <2>=dry (NOT IMPLETED)
  3.0         ! t_p        [hours]
  0.0 0.0 0.0  ! total U, A, N applied with IR [mg/L]
c -------------------------------------------------------------
C ... details of user supplied fertigations
c -------------------------------------------------------------
41  ! mm   Urea   NH4     NO3  no of irrigations
1     20    0     36.0    6.3
8     20    0     41.0    1.2
15    20    0     39.2    1.2
22    20    0     42.4    0.6
29    20    0     48.3    4.8
36    20    0     48.3    4.6
43    20    0     51.4    1.7
50    20    0     43.1    1.8
57    20    0     34.6    4.5
64    20    0     36.9    1.2
71    20    0     33.3    1.2
78    20    0     31.8    0.9
85    20    0     30.3    0.6
92    20    0     47.0    0.5
99    20    0     48.5    0.3
106   20    0     59.1    0.5
```

113	20	0	56.1	0.2
120	20	0	64.4	0.2
127	20	0	72.7	0.3
134	20	0	62.1	0.2
141	20	0	59.1	0.2
148	20	0	54.6	0.2
155	20	0	51.5	0.2
162	20	0	51.5	0.1
169	22.6	0	40.3	1.6
259	19.7	0	102.3	0.4
266	19.7	0	107.6	0.2
273	19.7	0	107.9	1.1
280	19.7	0	112.4	2.0
287	19.7	0	120.8	0.7
294	19.7	0	124.3	0.4
301	19.7	0	133.4	0.5
308	19.7	0	140.2	0.6
315	19.7	0	136.4	1.0
322	19.7	0	122.4	0.6
329	19.7	0	119.4	0.6
336	19.7	0	119.0	0.6
343	19.6	0	114.3	0.6
350	19.6	0	118.1	0.6
357	19.6	0	111.2	0.5
364	19.6	0	111.2	1.0

B.2 Parameter file for simulation of the E. nitens lysimeter treatment

```
c-----------------------------------------------------------------------
c ...   Simulation of the E.nitens lysimeter data
c ...   A lumped parameter model to calculate the leaching of fertilizer
c ...   beyond the root zone of a crop.
c ...   A reference ET is calculated for a given crop, based on crop
c ...   factors determined from planting dates etc.
c ...   Drainage is based on Sisson et al (1980)
c ...   N transformations based on Ling and El-Kadi (1998)
c ---------------------------------------------------------------
c ... Specify the ET model:
c ... IF ET model = 1 THEN
c ...   read in Rg, Ta, Rf and calc ET
c ... IF ET model = 2 THEN
c ...   read in Rg, Ta, Rf, Ir, Tx, Tn, Tw, Ws and calc LE
c ---------------------------------------------------------------
   2     ! ET model: <1> LE = à(s/s+ç)Rg ; <2> LE = (sRg + sçDa.ra)/(s+ç)
   1.0   ! effective rainfall coefficient
   2.5   ! et max for soil evaporation
c ---------------------------------------------------------------
c ... Specify the annual cycle of Kc - the crop coefficient
c
c     K2                  /:~~~~~~~~~~~~~:\
c                       /  :             :  \
c     K1     ___________/    :             :    \___________
c            :          :    :             :    :          :
c            T1         T2   T3            T4   T5         T6
c
c ---------------------------------------------------------------
   12          ! no of Kc values (e.g. 6 needed for one crop cycle)
  35 2.0       ! enter annual cycle of Kc   J   Kc(T1)
  28 2.9       !                            F   Kc(T2)
  28 3.4       !                            M   Kc(T3)
  35 3.8       !                            A   Kc(T4)
  28 3.9       !                            M   Kc(T5)
  27 4.0       !                            J   Kc(T6)
  28 3.5       !                            J   Kc(T7)
  28 3.2       !                            A   Kc(T8)
  28 3.0       !                            S   Kc(T9)
  28 2.8       !                            O   Kc(T10)
  35 2.2       !                            N   Kc(T11)
  37 2.6       ! make sure we have 365 values of Kc (T12) D
c ---------------------------------------------------------------
c ... Specify crop factors
c ---------------------------------------------------------------
  1.0          ! root depth Zr [m]
  0.8          ! drought tolerance (as a fraction of available water)
  2.8          ! light utilization efficiency [g-DM/MJ]
c ---------------------------------------------------------------
c ... Define plant N-uptake parameters (a logistic fn)
c ---------------------------------------------------------------
  0.0013       ! GF = SENESCENCE RATE OF LEAVES
  0.0003       ! GS = SENESCENCE RATE OF STEM
  0.0055       ! GR = SENESCENCE RATE OF ROOTS
  0.150        ! NF = DRY MATTER ALLOCATION TO LEAVES
  0.680        ! NS = DRY MATTER ALLOCATION TO STEM
  0.1700       ! NR = DRY MATTER ALLOCATION TO ROOTS
```

```
  0.500      ! LF = FRACTION OF LEAF-N RECYCLED
  0.50       ! LS = FRACTION OF STEM-N RECYCLED
  0.50       ! LR = FRACTION OF ROOT-N RECYCLED
  5.4E-4     ! SLA = SPECIFIC LEAF AREA [ha-leaf/kg-DM]
  0.03       ! NCFS = OPTIMUM N CONTENT OF LEAF [kg-N/kg-DM]
  10.00      ! NCRIT = SOIL-N [kg-N/ha] REQUIRED FOR GROWTH
  80000.     ! HMFT = HARVEST ABOVE GROUND DM WHEN IT IS >HMFT [KG/HA]
  1000.      ! HMFC = MINIMUM LEAF-DM [KG/HA]
  0.95       ! HFST = FRACTION OF STEM/BRANCH REMOVED DURING HARVEST
  0.0        ! HFDM = FRACTION OF HARVESTED DM RETURNED AS LITTER
  26.0       ! LAST DATE OF THINNING [YEARS]
c ------------------------------------------------------------
c ... Specify SOIL factors
c ------------------------------------------------------------
  0.36       ! ém    maximum water content [m3/m3]
  0.310      ! éf    field capacity [0.2 bar]
  0.080      ! éc    wilting point [15.0 bar]
  750.0      ! Km    saturated hydraulic conductivity [mm/d]
  0.0293     ! á     beta parameter        [-]
  1.2600     ! pb    soil bulk density     [kg/L]
c ------------------------------------------------------------
c ... Specify soil moisture and temperature response factors
c ------------------------------------------------------------
  0.60       ! es    saturation activity
  0.10       ! é1    increasing interval
  0.08       ! é2    decreasing interval
  3.0        ! QTEN factor
  20.0       ! Tb base temperature
  0.06       ! éd to reduce denitrification
  2.0        ! d factor
c ------------------------------------------------------------
c ... Specify the solute transport factors
c ------------------------------------------------------------
  0.000      ! k1 rate constant hydrolysis of Urea          [day-1]
  0.200      ! k2 rate constant nitrification of Ammonia    [day-1]
  0.006      ! k3 rate constant denitrification of Nitrate  [day-1]
  5.000      ! Kda distribution coefficient for Ammonium    [L/Kg]
  0.000      ! Kdn distribution coefficient for Nitrate     [L/Kg]
  0.500      ! Kh volatilization constant for Ammonia       [day-1]
c ------------------------------------------------------------
C ... Specify nitrogen mineralization parameters
c ------------------------------------------------------------
  20.0       ! ro = c/n ratio of the soil biomass
   0.4       ! fe = carbon turnover efficiency [d-1]
   0.2       ! fh = humification coefficient
   0.000     ! Kman = decomposition rate for soil manure
   0.0080    ! Klit = decomposotion rate for soil litter
   0.00007   ! Khum = decomposition rate for soil humus
c ------------------------------------------------------------
c ... specify annual C/N returns to system (mainly as leaf fall)
c ------------------------------------------------------------
   0.0       ! ignore
     0.5     ! carbon ratio in organic matter (OM)
    20.0     ! C/N ratio of OM
     0.9     ! humus fraction, litter = 1.0-humus fraction
c ------------------------------------------------------------
c ... Specify the initial water and solute conditions
c ------------------------------------------------------------
  0.260      ! éi initial water content      [m3/m3]
  0.000      ! Ui initial Urea conc in soil solution     [mg/l]
  0.000      ! Ai initial Ammonium conc in soil solution [mg/l]
```

```
  5.0          ! Ni initial Nitrate conc in soil solution  [mg/l]
  4320         ! Fi initial foliage DM          [kg/ha]
  2160         ! Si initial stem DM             [kg/ha]
  4320         ! Ri initial root DM             [kg/ha]
c ------------------------------------------------------------
c ... Specify initial organic C/N pool values
c ------------------------------------------------------------
     0.0       ! CMAN = CARBON IN MANURE POOL     (kg/ha)
     0.0       ! NMAN = NITROGEN IN MANURE POOL   (kg/ha)
   378.0       ! CLIT = CARBON IN LITTER POOL     (kg/ha)
    18.9       ! NLIT = NITROGEN IN LITTER POOL   (kg/ha)
  3402.0       ! CHUM = CARBON IN HUMUS POOL      (kg/ha)
   170.0       ! NHUM = NITROGEN IN HUMUS POOL    (kg/ha)
c ------------------------------------------------------------
c ... Specify the Irrigation regime (must supply atleast 5 parameters)
c ... <1> means user supplied BELOW
c ... <2> means automatic [X mm]from t(beg)to t(end)at dt intervals [day]
c ... <3> means irrigate[X mm]if wc(i)< [threshold = éc + dtol*(ém-éc)]
c ------------------------------------------------------------
   1           ! Irrigation <1>=as supplied,<2>=automatic, <3>=as needed
   0.0         ! ATMOS inputs of N [kg/ha/y]
   0.0         ! Allow a 10% IR deficit capacity [mm] before applying IR
   25.0        ! IR total    [mm]    [1 mm = 1 L/m2]
   1.0         ! t(beg)      [day]
 10000.0       ! t(end)      [day]
   7.0         ! IR interval [day]
 120.0         ! no effluent from here
 300.0         !             to here
c ------------------------------------------------------------
c ... Specify the Fertilizer regime (must supply atleast 5 parameters)
c ... <1>means as supplied
c ... <2>means automatic (fertigation)from F(beg)to F(end) at F-int [day]
c ... for each application enter day, X-U, X-A, X-N [kg/ha]
c ------------------------------------------------------------
  1            ! Fertilizer type   <1>=wet   <2>=dry (NOT IMPLETED)
  3.0          ! t_p        [hours]
  0.0 0.0 0.0 ! total U, A, N applied with IR [mg/L]
c ------------------------------------------------------------
C ... details of user supplied fertigations
c ------------------------------------------------------------
41   ! mm  Urea     NH4     NO3    !  no. of irrigations
1      20    0       36.0    6.3
8      20    0       41.0    1.2
15     20    0       39.2    1.2
22     20    0       42.4    0.6
29     20    0       48.3    4.8
36     20    0       48.3    4.6
43     20    0       51.4    1.7
50     20    0       43.1    1.8
57     20    0       34.6    4.5
64     20    0       36.9    1.2
71     20    0       33.3    1.2
78     20    0       31.8    0.9
85     20    0       30.3    0.6
92     20    0       47.0    0.5
99     20    0       48.5    0.3
106    20    0       59.1    0.5
113    20    0       56.1    0.2
120    20    0       64.4    0.2
127    20    0       72.7    0.3
134    20    0       62.1    0.2
```

141	20	0	59.1	0.2
148	20	0	54.6	0.2
155	20	0	51.5	0.2
162	20	0	51.5	0.1
169	22.6	0	40.3	1.6
259	19.7	0	102.3	0.4
266	19.7	0	107.6	0.2
273	19.7	0	107.9	1.1
280	19.7	0	112.4	2.0
287	19.7	0	120.8	0.7
294	19.7	0	124.3	0.4
301	19.7	0	133.4	0.5
308	19.7	0	140.2	0.6
315	19.7	0	136.4	1.0
322	19.7	0	122.4	0.6
329	19.7	0	119.4	0.6
336	19.7	0	119.0	0.6
343	19.6	0	114.3	0.6
350	19.6	0	118.1	0.6
357	19.6	0	111.2	0.5
364	19.6	0	111.2	1.0

B.3 Parameter file for simulation of an E. nitens plantation receiving dairy shed effluent applications at a rate of 200 kg N ha^{-1} yr^{-1}.

```
C ------------------------------------------------------------
c ...   E. nitens plantation simulation 200 kg N ha yr
C ...   A lumped parameter model to calculate the leaching of fertilizer
c ...   beyond the root zone of a crop.
c ...   A reference ET is calculated for a given crop, based on crop
c ...   factors determined from planting dates etc.
c ...   Drainage is based on Sisson et al (1980)
c ...   N transformations based on Ling and El-Kadi (1998)
c ------------------------------------------------------------
c ... Specify the ET model:
c ... IF ET model = 1 THEN
c ...   read in Rg, Ta, Rf and calc ET
c ... IF ET model = 2 THEN
c ...   read in Rg, Ta, Rf, Ir, Tx, Tn, Tw, Ws and calc LE
c ------------------------------------------------------------
   2     ! ET model: <1> LE = à(s/s+ç)Rg ; <2> LE = (sRg + sçDa.ra)/(s+ç)
   0.9   ! effective rainfall coefficient
   2.5   ! et max for soil evaporation
c ------------------------------------------------------------
c ... Specify the annual cycle of Kc - the crop coefficient
c
c     K2                  /:~~~~~~~~~~~~~:\
c                       /  :             :  \
c     K1     __________/    :             :    \__________
c           :          :    :             :    :          :
c           T1         T2   T3            T4   T5         T6
c
c ------------------------------------------------------------
   12         ! no of Kc values (e.g. 6 needed for one crop cycle)
  35 1.       ! enter annual cycle of Kc   J   Kc(T1)
  28 1.       !                            F   Kc(T2)
  28 1.       !                            M   Kc(T3)
  35 1.       !                            A   Kc(T4)
  28 1.       !                            M   Kc(T5)
  27 1.       !                            J   Kc(T6)
  28 1.       !                            J   Kc(T7)
  28 1.       !                            A   Kc(T8)
  28 1.       !                            S   Kc(T9)
  28 1.       !                            O   Kc(T1●)
  35 1.       !                            N   Kc(T11)
  37 1.       ! make sure we have 365 values of Kc (T12) D
c ------------------------------------------------------------
c ... Specify crop factors
c ------------------------------------------------------------
  1.0         ! root depth Zr [m]
  0.5         ! drought tolerance (as a fraction of available water)
  1.7         ! light utilization efficiency [g-DM/MJ]
c ------------------------------------------------------------
c ... Define plant N-uptake parameters (a logistic fn)
c ------------------------------------------------------------
  0.0013      ! GF = SENESCENCE RATE OF LEAVES
  0.0003      ! GS = SENESCENCE RATE OF STEM
  0.0055      ! GR = SENESCENCE RATE OF ROOTS
  0.15        ! NF = DRY MATTER ALLOCATION TO LEAVES
  0.68        ! NS = DRY MATTER ALLOCATION TO STEM
```

```
  0.17        ! NR = DRY MATTER ALLOCATION TO ROOTS
  0.50        ! LF = FRACTION OF LEAF-N RECYCLED
  0.50        ! LS = FRACTION OF STEM-N RECYCLED
  0.50        ! LR = FRACTION OF ROOT-N RECYCLED
  5.40E-4     ! SLA = SPECIFIC LEAF AREA [ha-leaf/kg-DM]
  0.03        ! NCFS = OPTIMUM N CONTENT OF LEAF [kg-N/kg-DM]
  10.00       ! NCRIT = SOIL-N [kg-N/ha] REQUIRED FOR GROWTH
  70000.      ! HMFT = HARVEST ABOVE GROUND DM WHEN IT IS > HMFT [KG/HA]
  1000.       ! HMFC = MINIMUM LEAF-DM [KG/HA]
  0.90        ! HFST = FRACTION OF STEM/BRANCH REMOVED DURING HARVEST
  0.00        ! HFDM = FRACTION OF HARVESTED DM RETURNED AS LITTER
  30.0        ! LAST DATE OF THINNING [YEARS]
c -------------------------------------------------------------
c ... Specify SOIL factors
c -------------------------------------------------------------
  0.44        ! ém    maximum water content [m3/m3]
  0.39        ! éf    field capacity [0.2 bar]
  0.08        ! éc    wilting point [15.0 bar]
  750.        ! Km    saturated hydraulic conductivity [mm/d]
  0.0293      ! á     beta parameter        [-]
  1.26        ! pb    soil bulk density     [kg/L]
c -------------------------------------------------------------
c ... Specify soil moisture and temperature response factors
c -------------------------------------------------------------
  0.60        ! es    saturation activity
  0.10        ! é1    increasing interval
  0.08        ! é2    decreasing interval
  3.0         ! QTEN factor
  20.0        ! Tb base temperature
  0.06        ! éd to reduce denitrification
  3.0         ! d factor
c -------------------------------------------------------------
c ... Specify the solute transport factors
c -------------------------------------------------------------
  0.560       ! k1 rate constant hydrolysis of Urea          [day-1]
  0.20        ! k2 rate constant nitrification of Ammonia    [day-1]
  0.006       ! k3 rate constant denitrification of Nitrate  [day-1]
  5.0         ! Kda distribution coefficient for Ammonium    [L/Kg]
  0.000       ! Kdn distribution coefficient for Nitrate     [L/Kg]
  0.500       ! Kh volatilization constant for Ammonia       [day-1]
c -------------------------------------------------------------
C ... Specify nitrogen mineralization parameters
c -------------------------------------------------------------
  20.0        ! ro = c/n ratio of the soil biomass
   0.4        ! fe = carbon turnover efficiency [d-1]
   0.2        ! fh = humification coefficient
   0.000      ! Kman = decomposition rate for soil manure
   0.008      ! Klit = decomposotion rate for soil litter
   0.00007    ! Khum = decomposition rate for soil humus
c -------------------------------------------------------------
c ... specify annual C/N returns to system (mainly as leaf fall)
c -------------------------------------------------------------
   1000.0 !!!IGNORE ! Organic matter returned to system [kg/ha/y]
     0.5      ! carbon ratio in organic matter (OM)
    20.0      ! C/N ratio of OM
     0.9      ! humus fraction, litter = 1.0-humus fraction
c -------------------------------------------------------------
c ... Specify the initial water and solute conditions
c -------------------------------------------------------------
  0.30        ! éi initial water content      [m3/m3]
  0.000       ! Ui initial Urea conc in soil  [mg/kg dry soil]
```

```
  0.000       ! Ai initial Ammonium conc soil [mg/kg dry soil]
  10.0          ! Ni initial Nitrate conc soil  [mg/kg dry soil]
  4320          ! Fi initial foliage DM         [kg/ha]
  2160          ! Si initial stem DM            [kg/ha]
  4320          ! Ri initial root DM            [kg/ha]
c ------------------------------------------------------------
c ... Specify initial organic C/N pool values
c ------------------------------------------------------------
    0.0         ! CMAN = CARBON IN MANURE POOL     (kg/ha)
    0.0         ! NMAN = NITROGEN IN MANURE POOL   (kg/ha)
  378.0         ! CLIT = CARBON IN LITTER POOL     (kg/ha)
   18.9         ! NLIT = NITROGEN IN LITTER POOL   (kg/ha)
 3402.0         ! CHUM = CARBON IN HUMUS POOL      (kg/ha)
  170.0         ! NHUM = NITROGEN IN HUMUS POOL    (kg/ha)
c ------------------------------------------------------------
c ... Specify the Irrigation regime (must supply atleast 5 parameters)
c ... <1>means user supplied BELOW
c ... <2>means automatic [X mm]from t(beg)to t(end)at dt intervals [day]
c ... <3>means irrigate [X mm] if wc(i) < [threshold = éc + dtol*(ém-éc)]
c ------------------------------------------------------------
   2            !IRRIGATION <1>=as supplied, <2>=automatic, <3>=as needed
   0.0          ! ATMOS inputs of N [kg/ha/y]
   0.0          ! Allow a 10% IR deficit capacity [mm] before applying IR
   9.5          ! IR total    [mm]    [1 mm = 1 L/m2]
  1.0           ! t(beg)      [day]
 10000.0        ! t(end)      [day]
   7.0          ! IR interval [day]
   120          ! no effluent from here
   300          !             to here
c ------------------------------------------------------------
c ... Specify the Fertilizer regime (must supply atleast 5 parameters)
c ... <1> means as supplied
c ... <2> means automatic (fertigation)from F(beg)to F(end)at F-int[day]
c ... for each application enter day, X-U, X-A, X-N [kg/ha]
c ------------------------------------------------------------
  1             ! Fertilizer type   <1>=wet   <2>=dry (NOT IMPLETED)
  3.0           ! t_p        [hours]
  0.0 81.0 0.2  ! total U, A, N applied with IR [mg/L]
c ------------------------------------------------------------
C ... details of user supplied fertigations
c ------------------------------------------------------------
0   ! mm   Urea   NH4     NO3  !  no of irrigations
1     20    0     36.0    6.3
```

Appendix C

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Short rotation forestry for land treatment of effluent – A lysimeter study

J. K. F. Roygard[A,B], *S. R. Green*[A], *B. E. Clothier*[A], *R. E. H. Sims*[C], *and N. S. Bolan*[B]

[A]Environment and Risk Management Group, Hort Research, Palmerston North, New Zealand

[B]Institute of Natural Resources, Massey University, Palmerston North, New Zealand.

[C]Institute of Technology and Engineering, Massey University, Palmerston North, New Zealand.

Abstract

Land treatment of wastewater using short rotation forestry (SRF) has potential as a sustainable method for disposal of dairy-farm effluent. We compared 3 SRF species; 2 evergreen species of eucalypts (*Eucalyptus nitens, E. saligna*) and a deciduous willow (*Salix kinuyanagi*), in the land treatment of dairy-farm effluent. The trees were grown in lysimeters (1.8 m diameter, 1.0 m depth), and a bare soil treatment was used as a control. The application of dairy-farm oxidation-pond effluent totalled 218 g N/lysimeter (equivalent to 870 kg N/ha) over 2 irrigation seasons (December 1995-June 1996 and September 1996-April 1997). Effluent was applied weekly in summer at a rate of 18.9 mm/week. No effluent was applied during the winter period. The evapotranspiration (ET) rates of the trees, and the volumes and nitrogen contents of the leachates are compared for a winter period (4 weeks) and a summer period (5 weeks). The biomass accumulation and the uptake of nitrogen by the 3 tree species were also investigated.

The SRF trees improved the renovation levels of dairy-farm effluent and produced biomass suitable for energy conversion. Of the 3 tree species, only the *S. kinuyanagi* treatments maintained leachate nitrate concentrations below the New Zealand drinking water standard of 11.3 mg NO_3^--N/L throughout both the winter and summer periods. The *E. nitens* treatment produced significantly more oven-dry biomass (19.1 kg/tree) than the *E. saligna* trees (9.7 kg/tree) (P=0.05). The *S. kinuyanagi* treatment had intermediate production (13.3 kg/tree) and was not significantly different from the other 2 tree species (P=0.05). The nutrient accumulation was not significantly different among the species (P=0.05). *S. kinuyanagi* was considered the best overall performer for the land treatment of dairy-farm effluent, based on the concentrations of leachate moving beyond the root zone.

Additional keywords: Dairy-shed effluent, *Eucalyptus nitens*, *Eucalyptus saligna*, nitrate, *Salix kinuyanagi*.

Introduction

Dairy farming is a major producer of wastewater in New Zealand. Under the Resource Management Act (1991) New Zealand dairy farmers are required to dispose of dairy-shed effluent in a manner which has no adverse effect on the environment. Dairy-shed effluent in New Zealand is most commonly treated via two-pond systems. The effluent from these pond systems may then be released into streams or rivers, however nutrient removal by two-pond systems is proving insufficient to protect the quality of receiving waters (Hickey *et al.* 1989). One further treatment option is to apply ponded-effluent to soil growing trees. This system has the potential to meet the needs of regulatory agencies because the tree roots are able to strip the nutrients from wastewater as it percolates through the soil. There are also economic incentives to utilise wastes containing nutrients for crop production. The fertiliser value of New Zealand's dairy-shed effluent, pig slurry, and poultry manure has been estimated in New Zealand to be NZ$36 million per year (Roberts *et al.* 1992).

Salix, *Eucalyptus*, and *Populus* species have recently been advocated in short rotation forestry (SRF) systems and their potential in land treatment systems is currently being investigated (Tungcul *et al.* 1996; Myers *et al.* 1996; Nicholas *et al.* 1997; and Nicholas 1997). The fast initial growth-rate of SRF crops suggesting high water and nutrient uptake, as well as the coppicing abilities of SRF crops are advantageous in land treatment systems (Nicholas *et al.* 1997); while biomass from SRF is suitable for energy conversion. SRF crops have been researched with municipal wastewater, meat processing effluent and dairy-farm effluent (Myers *et al.* 1994; Myers *et al.* 1996; Tungcul *et al.* 1996; Nicholas *et al.* 1997 and Nicholas 1997).

Vital to the successful design of an environmentally sustainable system of land-treatment is the evaluation of soil water and nutrient balances in the rootzone (Bond 1998). Nitrogen is the most important nutrient in the case of renovation of dairy-farm effluent by SRF crops. It is important to prevent nitrate concentrations building up in groundwater via leaching, as ground water is widely used as a potable water source in many countries including New Zealand. Monitoring the quantity of nitrate leaching to groundwater is an essential part of ensuring a system is operating in an environmentally sustainable manner. In New Zealand, the drinking water standard or Maximum Permissible Level (MPL) is 11.3 mg NO_3-N/L (Ministry of Health 1995).

Climatic conditions and the inputs of water and nitrogen change seasonally, as do the use and fate of water and nitrogen in land treatment systems. It might be expected that deciduous trees would have lower rates of water use in winter, in comparison to the evergreen trees which transpire throughout the whole year. The influence of the difference

in the rates of water use by the deciduous and evergreen trees will effect both nutrient removal and nitrate leaching. Consequently, the objectives of this study were to understand better the key processes of tree water use and nitrogen leaching from the rootzone of 3 species, with a bare soil control, receiving dairy-farm effluent. For comparison a four-week period in early winter (June-July), and a five-week period in early summer (December-January) were studied. In addition, we compared the biomass production and nitrogen accumulation of the 3 species over 2.5 years. We hypothesised that bare soil is, on its own, not suitable for the land treatment of wastewater, but trees will improve the level of effluent treatment that can be achieved.

Methods

Twelve, in-ground lysimeters containing 3 replicates of 3 tree species (*Eucalyptus nitens*, *E. saligna* and *Salix kinuyanagi*), and 1 bare soil treatment, were established in a field at Aokautere, near Palmerston North, New Zealand. The lysimeters were 1.8 m in diameter, 1.0 m deep were filled with Manawatu fine sandy loam (Weathered fluvial recent (Hewitt 1993); as described by Clothier et *al.* 1977). The soil was repacked into the lysimeters to the original bulk density. All lysimeters were left for a period of one year before data collection began. This delay allowed trees to establish and the soil to settle into the repacked lysimeters. A single tree was planted in each lysimeter, in November of 1994. The eucalypts (evergreen) were planted as 3-month old seedlings, and the willows (deciduous) as unrooted cuttings. The control lysimeters contained bare soil only. Trees of the same species were planted around the lysimeters at a density about 4000 stems/ha to create conditions approximating a small plantation. Dairy-farm effluent (secondary-pond treated) was applied weekly during the 2 irrigation seasons of December 1995 - June 1996 and September 1996 - April 1997. In the summer period, effluent was applied weekly at a

rate of 18.9 mm/week. But no effluent was applied during the winter period. The total hydraulic loading was 720 mm/year, with a nitrogen loading of 218 g/lysimeter (approximately 870 kg N/ha). Data collection from the lysimeters began in December 1995 when the trees were one-year old and continued through until September 1997. The above ground biomass of the trees in lysimeters was harvested in April 1997. The rotation time of 2.5 years is within the range (2–10 years) expected to be used for SRF crops grown for land treatment of dairy-farm effluent.

Insert Figure 1 about here

The lysimeter facility (Fig. 1) enabled measurement of all inputs and outputs of water and nitrogen. Effluent application by microjets onto the individual lysimeters was controlled by a pump and solenoid valves. Rainfall was recorded on-site. Soil water storage was measured via Time Domain Reflectometry (TDR) using probes installed at 5 soil depths in each lysimeter. Two TDR probes were inserted vertically into the soil to depths of 100 mm and 250 mm. The other 3 TDR probes were inserted horizontally at depths of 250, 500 and 750 mm. Soil water storage was calculated from the TDR data that were collected at least 5 times per week. Leachate volumes were recorded manually during the winter period and by tipping-bucket flow-meters in the summer period. Evapotranspiration (ET) (mm) was calculated from a simple water balance equation.

$$ET = I + R - D - \Delta S$$

where I (mm) is effluent irrigation, R (mm) is effective rainfall, D (mm) is the drainage of leachate and ΔS (mm) is the change in soil water storage.

Ammoniacal-(NH_4^+), nitrate-(NO_3^-) and nitrite-(NO_2^-) nitrogen concentrations in the surface applied effluent and drainage leachate were monitored regularly. Effluent samples were collected at the time of application, and then leachate samples were collected 3 -7 times per week, depending on leachate volumes. Nitrogen concentration was determined by following the nitroprusside method for NH_4^+-N analysis (Weatherburn 1967) and a diazotization coupling reaction (Griess-Ilosvay reaction) method for $N0_2^-$-N and NO_3^--N analysis (Bremner and Mulvaney 1982). Biomass accumulation and nitrogen contents of plant components were measured at the time of harvest in April 1997. Biomass production was scaled up from the lysimeters based on a stocking density of 4000 stems/ha. Individual trees from each lysimeter were separated into biomass components following the methods of Young and Carpenter (1975). These components were then oven dried to determine the oven dry weight of the biomass produced. Kjeldahl N digestion (Markus *et al.* 1985) was used to determine the nitrogen contents of subsamples of the biomass components for each tree.

One *E. saligna* and 1 bare soil replicate were not included in the analysis due to faults in their tipping bucket flow meters. Minitab was used for all statistical analysis. ANOVA tests were used to compare the treatments. No statistical effect was found among replicates.

Results

The average daily rainfalls recorded in the winter (3.1 mm/day) and the summer periods (2.8 mm/day) were similar.

Evapotranspiration

ET of the trees during summer (means between 4.59-5.71 L/day) was significantly higher (P=0.05) than the evaporation losses from the bare soil (mean 1.85 L/day) due to the high transpiration rate of the trees (Table 1). ET values for the 3 tree species in the summer period were not significantly different (P= 0.05). Winter ET values from the evergreen trees was significantly higher than the bare soil treatment (P=0.05). In winter, the ET of *S. kinuyanagi* (deciduous, 1.10 L/day) was significantly (P=0.05) lower than the *E. nitens* (2.49 L/day) and not significantly different from the *E. saligna* (1.93 L/day) and the bare soil (0.68 L/day) (P=0.05).

Insert Table 1 about here

Leachate volume

Because evaporation from the bare soil was much less than the ET of the trees there was a significantly greater volume of leachate (P=0.05), in both winter and summer (Table 1). In winter, leachate volumes for the 2 evergreen *Eucalyptus* species were significantly less (P=0.05) than from deciduous *S. kinuyanagi* treatment. Neither *S. kinuyanagi* nor *E. nitens* treatments leached during the summer period. Leachate was collected from *E. saligna* treatments after some summer rainfall, and effluent application events (Table 1).

Insert Figure 2 about here

Leachate N concentration

Figure 2 shows the winter-time concentration of nitrogen in the leachate compared to the maximum permissible level (MPL) of 11.3 mg NO_3^--N/L allowed for in drinking water. Each of the bare soil, *E. nitens*, and *E. saligna* treatments recorded leachate concentrations above the MPL throughout the winter period. Meanwhile the *S. kinuyanagi* treatments were very close to the MPL throughout the winter period.

Biomass production

The average quantity of biomass harvested from the 2.5-year-old trees varied between tree species. *E. nitens* gave the highest dry matter yield (19.1 kg/tree), followed by *S. kimuyanagi* (13.3 kg/tree), and with *E. saligna* yielding the lowest (9.7 kg/tree). The *E. nitens* produced significantly (P=0.05) more biomass than *E. saligna*. The *S. kimuyanagi* had an intermediate biomass yield, not significantly different (P=0.05) from the other 2 species. For a forest planted at 4000 stems/ha, the mean annual yield increment of these species would be equivalent to: *E. nitens* 30.6 oven dry tonnes (odt)/ha.year, *S. kimuyanagi* 21.3 odt/ha.yr, and for *E. saligna* 15.6 odt/ha.year. The trees were grown in lysimeters so approximation to biomass yields in field plantations should be treated with caution.

Nitrogen accumulation.

The above-ground biomass of *E. nitens* treatments contained some 116 g N/tree. This equates to about 53 % of the nitrogen applied. *S. kimuyanagi* treatments stored 122 g N/tree (56 %), while *E. saligna* stored 71 g N/tree (33 %). N accumulation was not significantly different between tree species because of the large variation between the trees within the treatments (P=0.05).

Discussion

In our study, dairy-shed effluent contained 75 - 125 mg NH_4^+-N/L, which is similar to the average value of 87 mg NH_4^+-N/L measured by Hickey *et al.* (1989) in a survey of dairy-effluent ponds in the Manawatu province in New Zealand. This concentration is six to eleven times greater than the MPL (Fig. 2), and would require extra treatment prior to discharge.

Further treatment of the effluent can be achieved using SRF trees in a land treatment system. The trees in the present study were able to remove 33-56 % of applied-nitrogen, thereby preventing the nitrogen leaching into water ways or groundwater. Hopmans *et al.* (1990) measured nitrogen uptake rates of *E. saligna*, *E. grandis*, *E. camadulensis*, *Populus deltoides*, *P. deltoides* x *P. nigra*, *Casuarina cunninghamiana* and *Pinus radiata* receiving municipal effluent at Wodonga, Australia. Effluent application added the equivalent of 400 kg N/ha/yr over a 44 month period. Hopmans *et al.* (1990) found no significant difference nitrogen uptake among the tree species, and N uptake averaged only 19 %, with a maximum N uptake of 28 %. The trees in the Wodonga experiment appear to be less efficient at taking up nitrogen than were the trees in this experiment. One possible explanation for this is that larger quantities of water were applied at Wodonga, than our study, possibly increasing nitrogen leaching concomitantly decreasing tree uptake at Wodonga.

Not all the nitrogen was, however, taken up by the trees in our study. Nitrate did pass beyond the rootzone where it would be expected to continue its passage downwards to contaminate the ground water. The amount of N leached from each tree treatment was similar. In winter the leaching ranged between 0.2 - 0.4 kg N/ha.day. There were no losses in the summer, except for a small amount from the *E. saligna* trees (0.02 kg N /ha/day). In all cases, the trees significantly (P=0.05) decreased nitrogen leaching losses in comparison to the bare soil treatment. N leaching losses from the bare soil treatment totalled 1.1 kg N/ha.day in the winter, and 2.5 kg N/ha.day in summer. The amount was higher in the summer as this was when the effluent was applied.

The differing water use by the *S. kimuyanagi* enhanced the quality of the water leaching beyond the root zone. The *S. kimuyanagi* (deciduous) treatments had significantly more winter leachate volume than the evergreen *Eucalyptus* trees (P=0.05). This results from lower water use by the deciduous trees during winter. Also, interception of rainfall by the canopy would be greater by the evergreen trees than for the deciduous trees, further increasing the hydraulic loading. The deciduous treatments leached a similar mass of nitrogen, but the concentration was diluted by the larger leachate volume. This dilution had the favourable result of lowering the N concentration to below the MPL.

Tree water uptake rates in our study were moderately less than the rates reported in Australian plantations. The *E. saligna*, *E. nitens*, *S. kimuyanagi* treatments summer ET rates were 5.71, 5.51 and 4.59 mm/day respectively. Australian reports of the rates of water use by effluent irrigated plantations vary. Dunin and Aston (1984) reported a maximum water-use of 7 mm/day in summer for a native eucalypt forest with non-limiting soil water availability in coastal New South Wales, Australia. Myers *et al.* (1996) reported maximum daily water use rates of 8.0 mm/day for 3 year old *E. grandis* trial plots irrigated with municipal effluent at Wagga Wagga, NSW, Australia. Tungcul *et al.* (1996) reported values ranging from 3.8 mm/day-9.65 mm/day for *Salix* species receiving effluent application at Aokautere, New Zealand, on a cloud free day in summer. The lower rates in our study are likely due to restricted water availability. Water inputs totalled 5.5 mm/day during summer not accounting for rainfall interception. Thus the *E. nitens* and *E. saligna* were utilising all applied water and some from the soils water storage. The willow was likely using all the water that it was receiving as the water content of the soil averaged 10.4 % during this period. Low ET in comparison to other studies may be a consequence of this limited availability of water.

A useful by-product of SRF effluent treatment crops is the production of woody biomass that can be used as an energy source. Our biomass production (scaled up from lysimeter measurements) yielded approximately 30 oven dry tonnes (odt)/ha.yr at a plantation density of 4000 stems/ha. This compares well with one of the highest biomass production values recorded for *E. nitens* in Rotorua, New Zealand. In Rotorua, *E. nitens* planted at 2200 stems/ha without effluent applied, produced 28 odt/ha.yr for 6 year old trees (Nicholas *et al.* 1997).

In our study, trees improved effluent treatment because higher evapotranspiration rates reduced the volume of leachate passing beyond the root zone. Uptake of nitrogen by the trees further reduced the quantities of nitrogen available for leaching. In this study *S. kinuyanagi* was the most suitable of the 4 treatments evaluated for land treatment. The low leachate nitrogen concentration is the key criterion in determining suitability of a tree species for land treatment of effluent. The biomass production and tendency for nitrogen accumulation are secondary criteria.

Acknowledgements

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References

Bond, W. J. (1998). Effluent irrigation – an environmental challenge for soil science. *Australian Journal of Soil Research* **36,** 543-55.

Bremner, J.M., and Mulvaney, C.S. (1982). Nitrogen – Total. *In* 'Methods of soil analysis, Part 2'. (Ed A.L. Page) pp. 595-624. Am. Soc. Agron., Madison, WI, USA.

Clothier, B.E., Scotter, D.R. and Kerr, J.P. (1977). Water retention in soil underlain by a coarse-textured layer: theory and a field application. *Soil Science* **123**, 392-99.

Dunin, F.X. and Aston, A.R. (1984). The development and proving of models of large scale evapotranspiration: An Australian study. *Agricultural Water Management* **8**, 305-23.

Hewitt, A. E. (1993). New Zealand Soil Classification. Landcare Research New Zealand Ltd: Lincoln , New Zealand.

Hickey, C. W., Quinn, J. M., and Davis-Colley, J. (1989). Effluent characteristics of dairy shed oxidation ponds and their potential impacts on rivers. *New Zealand Journal of Marine and Freshwater Research* **23,** 569-84.

Hopmans, P., Stewart, H. T. L., Flinn, D. W., and Hillman, T. J. (1990). Growth, biomass production and nutrient accumulation by seven tree species irrigated with municipal effluent at Wodonga, Australia. *Forest Ecology and Management* **30,** 203-11.

Markus, D.K., McKinnon, J.P., and Buccafuri, A.F. (1985). Automated analysis of nitrite, nitrate and ammonium nitrogen in soils. *Soil Science Society of America Journal* **49,** 1208-15.

Ministry of Health (1995). Drinking water standards for New Zealand. Ministry of Health Wellington, pp. 1-87.

Myers, B. J., Bond, W. J., Falkiner, R. A., O'Brien, N . D., Polglase, P. J., Smith, C. J., Theiveyanathan, S. (1994). Wagga effluent plantation project - Technical Report. CSIRO Division of Forestry, User Series Number 17, 123p.

Myers, B. J., Theiveyanathan, S., O'Brien, N.D., and Bond, W. J. (1996). Growth and water use of effluent-irrigated *Eucalyptus grandis* and *Pinus radiata* plantations. *Tree Physiology* **16**, 211-19

Nicholas, I. D. (1997). Hardwoods in land treatment schemes: Current status of New Zealand and Australian research. *In* 'Land Application of Biosolids'. New Zealand Land Treatment Collective Technical Session No 15. (Eds H. Wang and J.M. Carnus). Nelson, April 1997. (NZ Forest Research Institute: Rotorua, New Zealand). pp. 34-5.

Nicholas, I. D., Carnus, J. M., and Oliver, G. R. (1997). Comparative performance of tree species in New Zealand wastewater irrigation systems. *In* 'Proceedings of Land Treatment and Wetland Workshop'. (Eds H. Wang and J. M. Carnus). (New Zealand Waste Water Association and New Zealand Land Treatment Collective, NZ Forest Research Institute Rotorua, New Zealand). pp. 45-52.

Roberts, A. H. C., O'Connor, M. B., and Longhurst, R. D. (1992). Wastes as plant nutrient sources: issues and options. *In* 'The Use of Wastes and By-products and Fertilisers as Soil Amendments for Pastures and Crops' (Eds P. E. H. Gregg and L. D. Currie.) pp. 44-55. Occasional Report No 6, (Fertiliser and Lime Research Centre, Massey University., Palmerston North, New Zealand).

Tungcul, R. O., Sims, R. E. H., and Mason, I. G. (1996). Response of short rotation forestry to dairy-farm pond effluent irrigation. *In* 'Proceedings of the 7th National Bioenergy Conference' (South Eastern USA Regional Biomass Program. 15-20 September, Nashville, Tennessee, USA). pp. 793-99.

Weatherburn, M. W. (1967) Phenol-hypoclorite reaction for determination of nitrogen. *Analytical Chemistry* **39,** 971-74.

Young, H. E., and Carpenter, P. N. (1975). Sampling variation of nutrient element content within and among trees of the same species. In 'Oslo Biomass Studies International Union of Forest Research Organisations S4.01 Mensuration, Growth and Yield. Working Party on the Mensuration of the Forest Biomass'. College of Life Sciences and Agriculture University of Maine at Orono, USA.

Table 1. Water and nitrogen balance from the lysimeter study. ET, evapotranspiration, L_v, leachate volume, and M_L, mass of nitrogen leached. Within each row, means followed by the same letter are not significantly different at P=0.05.

Process	Units	Month	Bare soil	E. nitens	E. saligna	S. kinuyanagi
ET	mm/day	winter	0.68 a	2.49 b	1.93 b,c	1.10 a,c
ET	mm/day	summer	1.86 e	5.51 f	5.71 f	4.59 f
L_v	mm/day	winter	2.75 g	0.66 h	1.06 h	2.27 i
L_v	mm/day	summer	3.29 j	0.00 k	0.02 k	0.00 k
M_L	kg N/ha/day	winter	1.11 l	0.26 m	0.43 m	0.21 m
M_L	kg N/ha/day	summer	2.70 n	0.0 o	0.02 o	0.0 o

Figure captions.

Figure 1. Inputs and outputs of water and nitrogen in the lysimeter treatments.

Figure 2. Nitrogen concentrations of the effluent pond (NH_4^+) and the leachates (NO_3^-) compared to the maximum permissible level (MPL) (a) winter with no effluent; (b) summer with effluent.

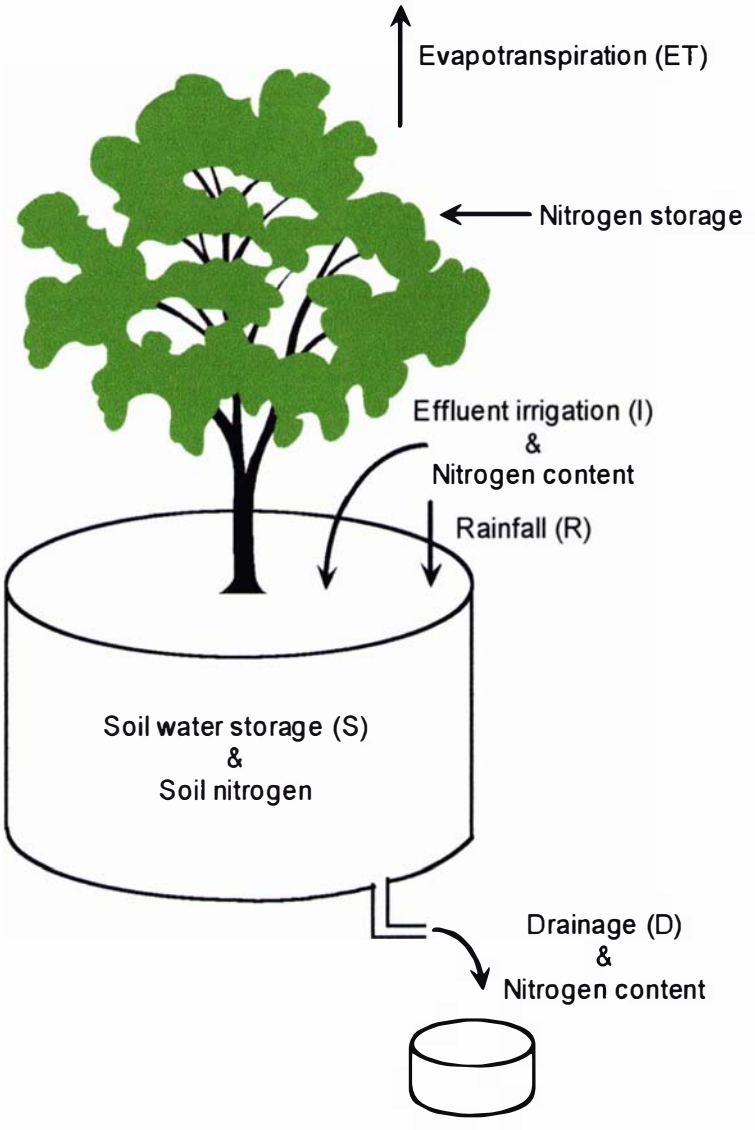

Figure 1. Inputs and outputs of water and nitrogen in the lysimeter treatments.

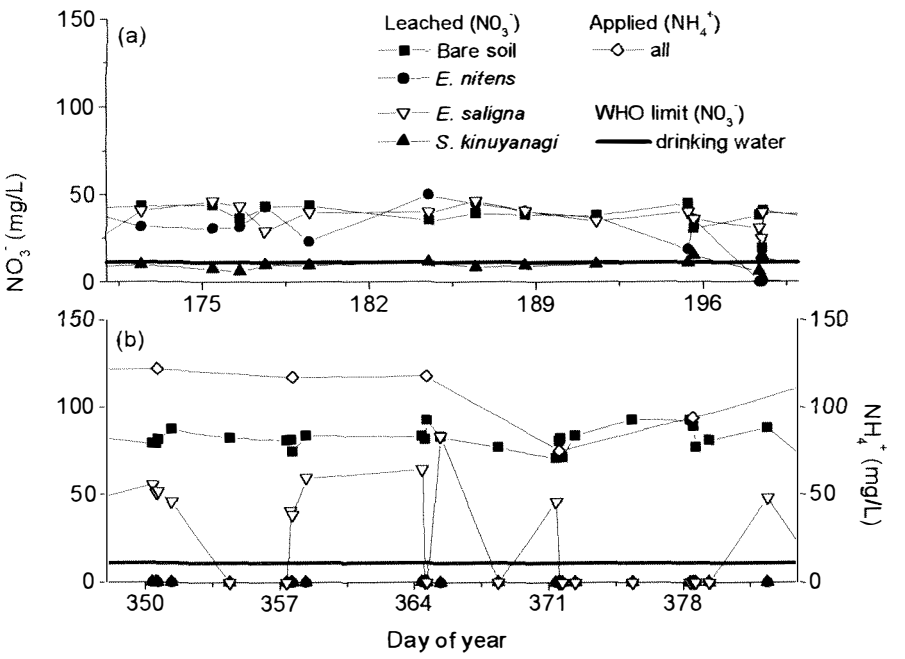

Figure 2. Nitrogen concentrations of the effluent pond (NH_4^+) and the leachates (NO_3^-) compared to the maximum permissible level (MPL) (a) winter with no effluent; (b) summer with effluent.

Addendum

Notes of clarification

General:

The lysimeters were arranged in a block design. Lysimeters were grouped into 3 replicates. Each replicate contained one lysimeter of each treatment. Treatments were *Eucalyptus nitens*, *E. saligna*, *Salix kinuyanagi*, and bare-soil. The bare-soil treatment provided a measurement of the 'treatment capacity' of soil alone. This allowed the role of the tree in the system to be clearly defined.

Guard trees nearby the respective treatments were of similar species to those in the lysimeters. Guard trees were planted at 4000 stems per hectare. The spacing of the trees reflected the size of ground area available to the single trees planted in the lysimeters i.e. 2.5 m^2. Trees outside the lysimeters were not irrigated. The lysimeter trees appeared more productive than the trees outside the lysimeters. Although biomass production of the surrounding trees was not quantified in the experiment. Canopy closure of the trees occurred in November of 1996. Tree height was not a good reflection of biomass production due to the production of multiple stems by some trees.

The lysimeters were repacked to simulate the natural soil profile of the Manawatu fine sandy loam. Effluent irrigation frequency and rationale are discussed in Section 3.2.2 on Page 38 and 39. Rainfall interception measurements were attempted during March and April of 1997. The measurements however were limited in use due to the lack of replication. Literature methods of estimating rainfall interception were thought to be a more accurate estimate (as described on Page 41).

Statistics for the field experiment, such as means and standard deviations, were calculated from the measurement of the 3 replicates of each treatment. The exception being *Eucalyptus saligna* which was represented by only two replicates. For example average root length densities were calculated by the following method. Four cores were collected from each replicate of the three tree treatments. Cores were divided into 0.1 m sections. The root length density of each of these cores was determined. The mean and standard deviations were then calculated from the cores for each treatment for each depth.

Weighing lysimeter data was collected however it was sporadic and unreliable. Problems in collecting the weighed lysimeter data included; loss of power to the data logger; small range of measurement and measurement of only change in weight. Thus following recalibration into the range for measurement the relationship to the previous collected data was unknown. For some periods the lysimeters were resting on the housing making measurements unreliable. Ceramic cup measurements were collected frequently (several times per week) during the experiment. Having measured leachate concentrations the data from ceramic cups were of limited use for this study.

Specific:

Page 46 2nd Paragraph "Some drainage could not be recorded"
Some drainage events exceeded the capacity of the collection buckets thus total volume could not be measured. This required some calculation of the drainage volume as outlined in Section 3.2.6.

Page 60 1st Paragraph "125 mm"
125 mm is an arbitrarily defined number that represents a low water content in the soil below which water stress is thought to occur.

Page 70 and throughout "Plant nitrogen accumulation"
Plant nitrogen accumulation was defined as the amount of nitrogen in the biomass on the day of harvest for above-ground biomass and on the day of root mass estimation for below-ground biomass. This excludes uptake that had been shed through senescence. Effectively it is the stored nitrogen on the day of sampling.

Page 76 Figure 4.4 "Organic soil nitrogen with depth"
Two soil cores were collected from each lysimeter for estimation of soil N content. Total organic nitrogen content was only chemically analysed for one core of each treatment. Thus no variability data is available for soil organic nitrogen with depth. The focus was on mineral nitrogen content with depth, as this was to be used for calculation of the nitrogen balance.

Page 129 3rd Paragraph "modelling tree root biomass changes following harvest"
Roots are not harvested or removed from the site. Following harvest, growth is low and the proportion of tree roots adjust to the proportion of above ground biomass through the senescence of roots being greater than the growth of roots.

Page 137 2nd Paragraph "Rainfall interception in the model"
The model used a single estimate of rainfall interception. It was not possible to adjust the rainfall interception through the growth of the biomass. Having demonstrated the sensitivity of this parameter, this is a possible area for future improvement of the model.

Page 163 "Reducing N concentration in leachate by dilution of effluent"
Other possible options for diluting the N concentration in the leachate include further effluent treatment prior to application to the tree system. Options range from increasing pond management e.g. cleaning, aerating or further treatment such as zeolite and bark filters.

Further such dilution would require an assessment of plant nitrogen requirements and water requirements of the plants. Indications of the model suggest the plants require around 110 kg of N for uptake (Table 5.4 Page 126 and Table 5.5 page 147). Water requirements should be around the rate of potential evaporation in summer (assuming a crop coefficient of 1.0). However the model could easily be utilised to provide more accurate assessments of the water and nitrogen requirements. It is recommended that determination of these requirements be completed using the model.