CHAPTER 7. THE PREDICTABILITY OF DIFFUSE NITRATE CONTAMINATION OF GROUNDWATER

7.1. INTRODUCTION

7.1.1 General

Earlier components of this research have reported key outcomes regarding the source and origin of nitrate into groundwater in the study area. These findings have included:

(i) elevated groundwater concentrations in the centre of the study area are likely to be due to local point-sources; and,

(ii) definition of a previously reported 'plume' encompassing Coonawarra were misguided.

However, the work also identifies that there is evidence of elevated concentrations of nitrate in groundwater that cannot be easily explained by point sources (particularly in the western side of the study area). This outcome is supported by the landuse statistical method that found significant difference between nitrate concentrations under grazing and vineyards.

Quantification and modelling of non-diffuse nitrate sources has been previously undertaken in the South East region of South Australia (e.g. Dillon 1988, Richardson 1990), resulting in considerable knowledge of the risks and response measures that relate to these sources. The leaching of nitrate to groundwater has also been studied for non-point sources within the South East region, with considerable work having been completed regarding non-irrigated pasture, irrigated pastures, and effluent irrigation areas (Dillon 1989, Pakrou and Dillon 1995, Dillon et al. 1996, Pakrou 1997, Dillon, et al. 2000, Pakrou and Dillon 2000, 2004). These later studies have reported the magnitude (and in some cases) compared nitrate leaching under different landuses and land management systems.

This chapter investigates the modelled variability of leaching under a series of representative land use regimes within the study area to ascertain potential nitrate leaching contributions. Modelling was also used to provide detail regarding potential mechanisms controlling diffuse nitrate leaching to the unconfined aquifer.

There have been many nitrogen leaching models generated for a variety of environments and situations, and other references provide a summary of them (Canter 1997, Borah and Bera 2003). The key trade-off in determining an appropriate descriptive (modelling) approach is between the availability of information describing variability within the environment (e.g. soil, geology, land management), and the scale of the system being simulated (e.g. regional, catchment, paddock or in-laboratory profile).

The aim of this component of the research was to quantify nitrate leaching under specific land management regimes, with the intention that the assessment was applicable to other regional areas. The approach chosen was to apply the one-dimensional LEACHN model (Hutson 2003) to describe nitrogen cycling for specific landuse scenarios within the study area.

The LEACHN model was selected as it includes capacity to take into account the complex nitrogen cycling within soil and plant systems, while not necessitating resource intensive on-site data gathering (e.g. soil infiltration studies).

Application of the LEACHN model to specific land management scenarios within the study area examined variability between these systems, and provides a context for, and additional confidence in, the observations previously reported in this thesis.

7.1.2 The LEACHN model

LEACHN is a specialised nitrogen simulation module of the LEACHM model. The architecture and the operation of the model is described in detail by the programmers (Hutson and Wagenet 1991, Hutson 2003), however a brief overview is provided for context.

The LEACHN model simulates vertical nitrogen movement within the unsaturated soil profile. The model is constructed so that the soil profile is described as a series of discrete (horizontal) layers, each with its own physical characteristics, water retention (and content) characteristics, and nitrogen content (and form). The model allows the role of vegetation (water uptake and nutrient cycling) to be incorporated into the soil profile. Water inputs into the profile can be from rainfall and irrigation, and nutrient inputs can be from fertiliser application or other amendments (e.g. animal waste). The model incorporates water losses from the profile to runoff (negligible in this study), evapotranspiration and leaching. Nutrient losses can be through runoff, gaseous losses, plant removal or leaching.

In this study the model operates on a time-step of 0.1 days for the duration of the model. At each time-step, all fluxes are calculated using the model algorithms. For nitrogen, these fluxes include between pools (and sources and sinks) as well as between profile layers. A simplified conceptual representation of the model illustrating the main pools and fluxes for water and nitrogen calculated at each time step is presented in Figure 7.1.

The van Genuchten equation (van Genuchten 1980) was used to simulate water flow within the model. The parameters for this function were estimated from the percentage of clay, silt and sand in each layer using NeuroTheta (Minasny and McBratney 2003). For the purposes of modelling water flow, physical and chemical soil properties are important and so these are discussed in detail in this Chapter.

As indicated in Figure 7.1, the profile is modelled to the maximum depth reached by vegetation. Due to the shallow nature of the study area's soils and groundwater, it was considered that water (and dissolved nitrate ions) passing below the root zone would quickly enter the aquifer.

The nitrogen fluxes for the model are estimated for inputs and plant removal (leaching and gaseous losses are simulated by the model). Although not

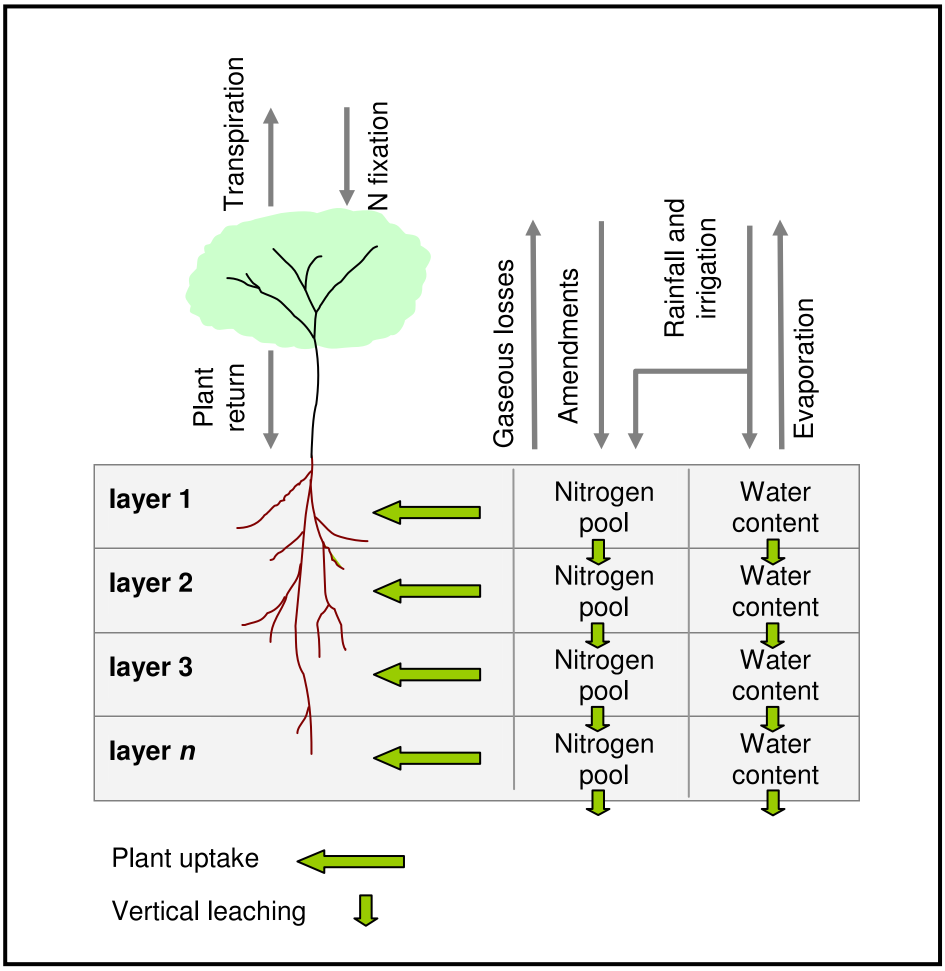


Figure 7.1: A simplified conceptual framework of the LEACHN model illustrating the main fluxes and pools for water and nitrogen

detailed in Figure 7.1, the fluxes between different forms of nitrogen (i.e. organic and inorganic forms) are also calculated at each time step.

Net plant uptake of nitrogen and water is estimated from existing literature, and the root distribution of the plants is also estimated so that the model can realistically simulate water (and nutrient) uptake from each layer. Plants can either be growing throughout a season, or be at maturity. Plant residues after harvest are returned to the soil, and for this study, the C:N ratio for plant residue was set (within the LEACHN code) to be 20:1.

The organic nitrogen pool within a soil profile is substantially influenced by the initial conditions that are estimated for organic carbon content in the soil. Given the importance of the flux quantities and the initial conditions to the model, considerable effort was made to determine realistic model inputs.

7.2. METHODOLOGY

While there are a variety of land systems and land management practices across the study area, a small number of 'typical' land management scenarios were identified to represent the main land management approaches. Point sources were not modelled. Modelling was targeted towards land systems already classified within this study.

Five scenarios were assessed; vineyards (irrigated and non-irrigated), improved pastures, leguminous crops and native vegetation. Nitrogen and water cycling (including leaching) for each of these scenarios was modelled from 1940 to 2005 using the LEACHN model.

All input files (describing the modelling environment) for each scenario are included in Appendix 8, however a description of each scenario and the origin of assumptions and data used in each assessment is provided below.

7.3. MODEL INPUTS

7.3.1 Scenarios

Irrigated vineyards

Landuse in the central part of the study area is dominated by irrigated vineyards. The soil type used in simulation of this land use was a terra rossa (shallow red loam over limestone). The Coonawarra grape growing area is characterised by this soil type (Blackburn 1983). The soil depth varies throughout its range, but is usually not more than 40 cm (Figure 2.8). The soil physical and (initial) chemical properties are shown in Table 7.1, and have been derived from soil profiles presented in Appendix 9 (as with all subsequent scenario soil information).

The underlying Tertiary and Quaternary limestones vary across the study area, and have layered sub-units. There is little information on the hydraulic conductivity of these geological units, with the available regional data having recently been reviewed by Mustafa and Lawson (2002). A specific yield (the quantity of water that will drain under gravity from the aquifer matrix, as a proportion of the volume of the aquifer) of 0.1 has been adopted as a general approximation for the Tertiary limestone aquifer at a regional scale (Walker, et al. 2001). Within the study area the hydraulic conductivity of the Gambier Limestone has been estimated to be 10 m/day (Williams 1978), and while this is an order of magnitude below other regional examples (Waterhouse 1977), it is above the estimated regional range of 0.9 to 3.9 m/day (Love, et al. 1994). To the west of the study area, a specific yield of 0.15 has been estimated for the Bridgewater Formation from water table fluctuations and geophysical logs (Mustafa et al. 2006). These flow estimates are usually measured through lateral pump tests, and therefore caution is needed when considering these in the context of anisotropic geological settings such as in the study area. To the north of the study area (Padthaway), van der Akker (2005) measured the gravimetric water content (θ_g) and soil water suction for profiles including the Bridgewater Formation for application of the LEACHN model.

For this and all other scenarios, the limestone units are considered to be uniform for the modelled profile depth. From available literature, the organic carbon content of these units are estimated to be <0.1% (Stace et al. 1972, DWLBC 2002). The clay component of limestone strata (Padthaway and Coomandook-Bridgewater Formations) is estimated to be 2%, with a silt percentage of 2% based upon interpretation of borehole logs (see Appendix 1). Due to the cemented nature of the limestone, a bulk density of 1.9 kg/L (Mee 2001) was included in determining the water retention curves.

The scenario assumed vegetation was an established vineyard with a mown alley management system. There was no root growth inclusion in the model reflecting the fact that vineyards in the centre of the study area are well established.

Table 7.1: Soil physical and chemical properties used for determining water retention curves for the vineyard scenario

Soil Group	Profile Depth(m)	Clay %	Silt %	Fine Sand %	Coarse Sand %	Nitrogen %	Carbon %
Terra rossa	0.0-0.1	20	5	30	45	0.2	2.5
Terra rossa	0.1-0.2	20	5	30	45	0.15	2.5
Terra rossa	0.2-0.4	50	5	20	25	0	2.5
Terra rossa	0.4-0.5	60	0	10	30	0	1
Terra rossa	0.5-0.6	70	0	10	20	0	1
Limestone substrate	0.6-2.0	2	2	30	66	0	0

The growing season has been selected to be from bud burst (start of September) to harvest (and start of winter dormancy) at the start of April (Hamilton and Coombe 1998, Wood 2000).

Irrigation included in the scenario was both frost irrigation (overhead) and growing season irrigation (dripper). The modelled frost irrigation is based upon the work of Pudney and her colleagues (Pudney et al. 2006, 2006, Pudney 2007). It assumes that September, October and November are the frost risk months, and that at any time during this period, frost irrigation is activated when the air temperature drops to 2^{o}C. A review of meteorological data for the study area identified that there was considerable variability in the number of ‘frost risk days’ between 1966 and 2005 (Figure 7.2). Prior to 1966, the meteorological data does not include daily values, and values for days prior to this date are interpolated from long term daily averages.

Pudney and her colleagues (2006) assumed that water application from frost sprinklers was 3.7 mm/hr for approximately seven hours for each occasion of 2^{o}C. This application rate is at the upper end of normal frost applications rates of between 2.5 and 3.5 mm/hr (McCarthy et al. 1998), although it is consistent with earlier estimated application rates within the study area of between 2.0 to 5.0 mm/hr (Harvey 1975). Based upon the information in Figure 7.2, there is considerable variability in the volume of frost irrigation

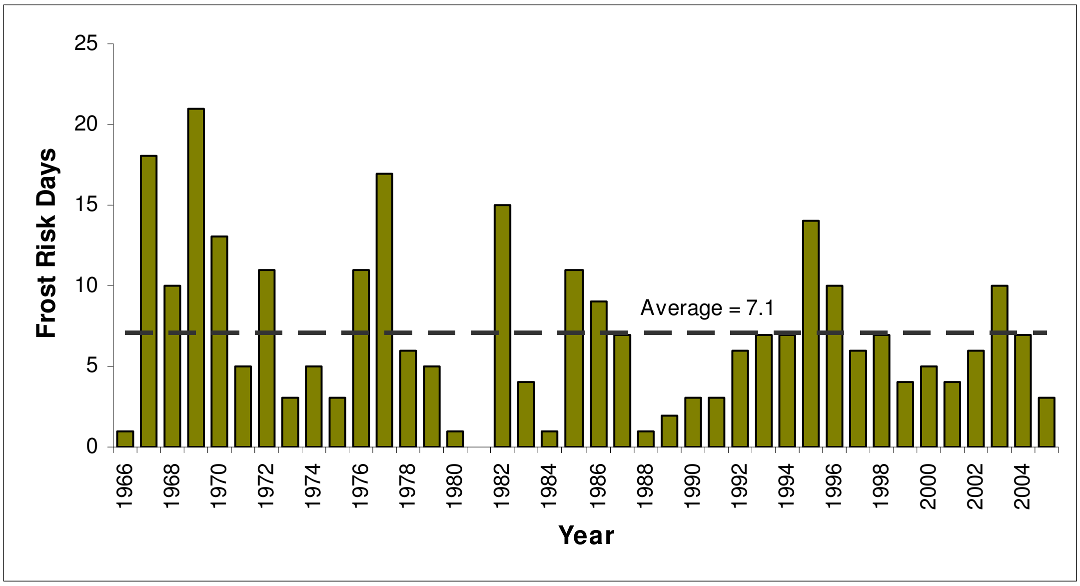


Figure 7.2: Number of frost risk days 1966-2005

water between years for this scenario; ranging from zero in 1981 to 525 mm/ha in 1969 (when there were 21 frost risk days).

Net growing season irrigation requirements for vineyards in the Coonawarra area have recently been reviewed for volumetric conversion of water allocations, and have been calculated as 189 mm/yr (climatic zone 3A);(Carruthers et al. 2006). This figure has been used in this scenario - applied at an even amount (7.2 mm) twice weekly from November to January (approximately to veraison); although it was not applied if frost irrigation also occurred on the same morning.

The vertical distribution of vine roots has been shown to vary based on factors such as site management and the vine cultivar (Smart et al. 2006). In the study area, the presence of the shallow Padthaway limestone is expected to control the root distribution, and therefore the distribution in this scenario is based upon that for shallow ploughed vineyards; namely 0-25cm (50%), 25-50cm, (31%) 50-75cm (12%), 75-100cm (6%) and 100-125cm (1%) (McCarthy, et al. 1998).

Nitrogen uptake during the growing season is estimated to be 21.2 kg N/ha, based upon an individual vine uptake of approximately 50 mg/day (Schreiner et al. 2006), a vine density of 2,000 vines/ha (Boehm and Coombe 1995), and a 212 day season. The vine crop cover ratio (proportion of ground surface covered by plant leaves) is estimated to be 0.6 with a maximum crop cover achieved in mid December.

The evapotranspiration scaling factor used for the irrigated vineyards (K_c) is 0.7 and is adopted from the scaling factor for the main growing period of wine grapes developed by the Food and Agriculture Organization of the United Nations (Allen et al. 1998).

Non-irrigated vineyards

Vegetation, soil and plant characteristics of a non-irrigated vineyard were taken as the same as an irrigated vineyard. The only difference is that the former is not subjected to any irrigation (frost or summer growing season irrigation).

Native vegetation/plantation forest

The native vegetation/plantation forest scenario models a significant land use on the eastern side of the study area. While water and nutrient cycling within these two landuses may differ, they are combined here as their variability was expected to be less than that between other landuses. In addition, research already reported within this study found that these landuses do not appear to be a significant source either of nitrate to the unconfined aquifer, or recharge to drive groundwater flows. These aspects suggested that differentiation of these two land uses was not warranted.

The soil type for this scenario is a highly leached sand. This soil is relatively uniform throughout its depth and depth to limestone is generally up to 10 metres (De Silva 1994, Bradley, et al. 1995). This soil often exhibits an organic-rich band at between 1 and 1.5 metres. The soil physical properties used in the scenario are shown in Table 7.2.

Table 7.2: Soil physical and chemical properties used for determining water retention curves for the native vegetation scenario

Soil Group	Profile Depth(m)	Clay %	Silt %	Fine Sand %	Coarse Sand %	Nitrogen %	Carbon %
Highly leached sand	0.0-1.1	1	1	30	68	0	0.1
Highly leached sand*	1.1-1.3	1	1	30	68	0.01	2
Highly leached sand	1.3-2.0	1	1	30	68	0	0.1

* An organic-rich layer

While there have been recent studies that have quantified water extraction by plantation forestry directly from the aquifer (Benyon and Doody 2004), this scenario considers that the depth to the water table is approximately 10 metres, and therefore there is no direct water extraction from the aquifer by tree roots.

Vegetation of this scenario is either an established woodland dominated by mature trees of *Eucalyptus camaldulensis* and *E. baxteri* or an established softwood plantation (Laut et al. 1977).

Crop canopy for this landuse was assumed to be 1.0 as a perennial vegetative cover. The FAO56 study reports an evapotranspiration scaling factor (K_c) for coniferous trees of 1.0 (Allen, et al. 1998). Benyon and Doody (2004) reported that the actual evapotranspiration from a young *E. globulus* plantation in South Australia's south east region was equal to rainfall (where the plantation did not access water directly from the aquifer) indicating that evapotranspiration in this study was limited by plant-available water, and that the measured amount was less than the potential evapotranspiration. Higher evapotranspiration rates were identified when plant-available water was not limited. A review of evapotranspiration from forests across the world (including from Australia) found scaling factors were between 1.0 to 2.0

(Zhang et al. 2001). Due to the water conserving nature of *Pinus* and *Eucalyptus sp.* a K_c of 1.0 was used in this scenario.

The root distribution for vegetation in this scenario was assumed to be 0-20 cm (40%), 20-50 cm, (30%) 50-100 cm (25%) and 100-200 cm (5%) based upon research into *Eucalyptus* species (Bouillet et al. 2002, Moroni et al. 2003, O'Grady et al. 2005).

There is little information on the nitrogen uptake from undisturbed native vegetation, with most of the available information relating to plantation systems. Depending upon the plant density, more than 300 kg N/ha/yr of nitrogen uptake has been reported for *E. moluccana* (Grey Box) under effluent irrigation over a two year period (MacDonald et al. 2004). Highly-productive established *Eucalyptus* plantations (not subject to effluent application) can have nitrogen uptake rates of approximately 200 kg N/ha/yr (Smethurst et al. 2004). South of the study area, nitrogen uptake from *Pinus radiata* plantations have been shown to vary from 26 to 171 kg N/ha/yr under different management and fertiliser regimes (Carlyle 1998). For the purposes of this scenario, it was assumed that there were no anthropogenic nitrogen applications, and therefore a reduced nitrogen uptake of 50 kg N/ha/yr was set. Although some Australian native plants are able to fix atmospheric nitrogen (Forrester et al. 2006) this is not included in this scenario.

This scenario did not incorporate any irrigation, grazing, harvesting or anthropogenic nitrogen application.

Improved pasture

In both the eastern and western parts of the study area, there are broad areas that remain as improved pastures for stock grazing. The soil type used in this scenario is a sandy loam over brown clay, with the initial physical and chemical properties shown in Table 7.3.

Vegetation in this scenario is a mixture common in the area (S Haase, pers. comm. 2007): 50% subterranean clover (*Trifolium subterranean*) and 50% perennial grasses (e.g. *Lolium sp., Phalaris sp., Dactylis sp.*). The growing

Table 7.3: Soil physical and chemical properties used for determining water retention curves for the grazing scenario

Soil Group	Profile Depth(m)	Clay %	Silt %	Fine Sand %	Coarse Sand %	Nitrogen %	Carbon %
Sandy loam over clay	0.0-0.2	5	5	30	60	0.05	0.5
Sandy loam over clay	0.2-0.3	20	5	20	55	0	0.5
Sandy loam over clay	0.3-0.4	30	5	20	45	0	0.5
Sandy loam over clay	0.4-2.0	50	5	20	25	0	0.1

season for the clover is from the end of April to the end of November (with flowering and maximum biomass during spring; the end of September). The maximum potential growth rate for grasses was assumed to be constant throughout the year.

Pasture establishment requires cultivation and pasture improvement can be undertaken by irregular shallow cultivation (at least once every five years). Biennial shallow cultivation was included in the scenario to a depth of 100 cm. Subterranean clover is capable of biological nitrogen fixation, and the mixture of clover species with perennial grasses allows nitrogen transfer to the grass species due to mineralisation of the senescing clover root system (Laidlaw et al. 1996, McNeill et al. 1998).

Anderson and his colleagues (1998) reported upon a range of studies within southern Australian that have shown that nitrogen fixation by subterranean clover can range from 2 to 238 kg N/ha/yr in mixed pastures. Bergersen and Turner (1983) also reported that the maximum fixation rate was approximately 4 kg N/ha/day in a mixed pasture system. The total nitrogen fixation (partitioned in above-ground and below-ground biomass) relevant to the study area is 29-162 kg N/ha/yr (*mean*=80, *n*=5);(Anderson, et al. 1998) and 123 kg N/ha/yr (McNeill, et al. 1998), respectively. Field studies have estimated that generally, subterranean clover obtains approximately 70-90% of its nitrogen requirements from nitrogen fixation (Bergersen and Turner 1983, Ledgard et al. 1985, Anderson, et al. 1998, Dear et al. 1999, Pakrou and Dillon 2000). These studies indicate that total nitrogen uptake by subterranean clover could range from 169 to 407 kg N/ha/yr. For this scenario, and recognising the proportion of subterranean clover in the pasture (50%), the total nitrogen uptake was based primarily on the estimates of Pakrou and Dillon (2000) where the total (non-irrigated) clover requirements were estimated to be 125 kg N/ha/yr, of which 80% (100 kg N/ha/yr) is sourced from nitrogen fixation.

Phalaris nitrogen uptake has been estimated to range from approximately 50 to 230 kg N/ha/yr depending upon plant density, with the majority of field sites requiring more than 100 kg N/ha/yr (Dear, et al. 1999). In this scenario, the nitrogen requirement of the perennial grasses was assumed to be 100 kg N/ha/yr. The scenario assumed that that no nitrogen fertiliser was applied.

Root density for the pasture scenario was taken to be 0-10cm (50%), 10-20cm (30%), 20-30 cm (10%), 30-40 cm (4%), 40-50 cm (3%), 50-60 cm (1%), 60-70 cm (1%), 70-80 cm (0.5%), 80-90 cm (0.5%) based upon *Lolium* and *Trifolium* sp. pasture studies (Mengel and Steffens 1985).

The evapotranspiration scaling factor (K_c) for grazing pasture is reported to range from 0.85 to 1.05 during the growing season (Allen, et al. 1998). In the volumetric conversion project for water allocation in the South East region Skewes (2006) provided a similar range of values, although he discriminated between high production pastures and lower production pastures. An improved pasture in central Victoria (*Lolium* and *Trifolium* sp.) was investigated between 1959 and 1966 and a K_c of approximately 1.08 was estimated (Dunin 1970). As the scenario tested here is for a lower level of production, a K_c of 0.85 was adopted.

For a dairy property south of Mt Gambier, Pakrou and Dillon (2000) assumed that grazing removed 70% of above-ground forage at a continuous stocking rate of three cows/ha; or 57 DSE (dry sheep equivalents) based upon lactating dairy cows (McLaren 1997). For the study area, an annual average stocking rate for beef grazing cattle is more likely to be 10-12 DSE (S Haase, pers. comm., 2007) or 1.4 – 1.7 cattle/ha (McLaren 1997). Within this scenario, the annual forage removal is assumed to be the same as that assumed by Pakrou and Dillon (2000), while the faeces and urine inputs are assumed to be proportionally reduced from those determined by them. Therefore the scenario assumed faeces and urine inputs of 15 kg N/ha/yr and 19.2 kg N/ha/yr respectively.

This scenario did not include irrigation, as irrigation of pastures is not common throughout the study area.

Legume cropping

A legume cropping scenario was developed to model the study area's western portion where cropping of Faba Beans (*Vicia faba*) is a significant broadacre cropping system. The soil type used is a shallow dark clay loam over limestone (groundwater rendzina), with a depth to limestone of reasonably uniform soil of around 0.8 m. The initial soil physical and chemical properties used for determining the water retention curve are shown in Table 7.4.

Table 7.4: Soil physical and chemical properties used for determining water retention curves for the legume cropping scenario

Soil Group	Profile Depth(m)	Clay %	Silt %	Fine Sand%	Coarse Sand%	Nitrogen %	Carbon %
Black cracking clay	0.0-0.2	55	10	20	15	0.2	1.5
Black cracking clay	0.2-0.3	55	10	20	15	0.1	1.5
Black cracking clay	0.3-0.4	55	10	20	15	0.05	0.5
Black cracking clay	0.4-0.8	55	10	20	15	0	0
Limestone substrate	0.8-2.0	2	2	30	66	0	0

A single cropping of *Vicia faba* sown at the start of June and harvested at the end of January was modelled. Cultivation occurs in mid April. The crop cover ratio achieves a maximum of 1.0 at the end of October, and reduces to 0.3 at harvest. The harvest proportion of forage is assumed to be 0.8 (either through burning or bailing), as there is a need to reduce forage load to not impede equipment for the following cultivation.

The legume *Vicia faba* is able to biologically fix atmospheric nitrogen. A range of field studies within Australia have reported that the crop obtains between 62 and 76% of its nitrogen requirements (61-171 kg N/ha/yr) from nitrogen fixation (Peoples et al. 1995). For this scenario, the total nitrogen requirement for the Faba Bean crop was estimated to be the mid range of these studies (116 kg/ha/yr), of which approximately 70% (81 kg/ha/yr) was sourced from biological nitrogen fixation. It was assumed in this scenario that there are no crops or grazing between harvest and subsequent sowing of the next year's crop.

Root density for the crop scenario was assumed to be 0-10cm (50%), 10-20cm (20%), 20-30 cm (12%), 30-40 cm (8%), 40-50 cm (6%), 50-60 cm (3%), 60-70 cm (1%) adopted from the relationship described by Reid and his colleagues (1984).

The evapotranspiration scaling factor (K_c) for Faba Beans is reported to be up to 1.15 during the growing season, reducing to 0.3 prior to harvest when the bean vine senesces (Allen, et al. 1998). Monthly crop coefficients are provided by Skewes (2006) for the growing season in the South East region, and based upon these values, a K_c for the whole growing season was taken to be 0.8.

This scenario does not include irrigation, grazing or fertiliser applications. It is acknowledged that inter-seasonal grazing may occur on some properties, however this was not incorporated in this scenario.

7.3.2 Precipitation and evapotranspiration

Rainfall and evaporation data for the model was sourced from SILO data provided by the Bureau of Meteorology for Coonawarra weather stations.

7.4. RESULTS

7.4.1 Nitrogen leaching under different scenarios

All of the scenarios were developed over the period from 1 January 1940 to 31 December 2005 using the available environmental data for this period, and assuming constant landuse.

These simulations allowed the major leaching fluxes within the scenarios to be calculated on an average annual basis (shown in Figure 7.3). Reported calculations exclude the fluxes modelled for the first 10 years of the simulation (i.e. 1940 to 1949). This approach was adopted as the initial physical and chemical properties of the soil established in the model did not set all characteristics (such as water content and oxidised nitrogen content in each layer). While these characteristics stabilised within the model after a short period, the earlier years tend to report unrealistic leaching and plant growth results.

The results for each scenario were generated by a single pass-through of the model over the years 1940-2005. The exception was for the native vegetation scenario. The simulation for this scenario was repeated using the meteorological data for five iterations during which the model reproduced a steady-state with respect to nutrient soil pools.

Table 7.5 shows mean annual fluxes but there is considerable variability in the reported components. The variability in the infiltrating rainfall shown in Table 7.5 is the result of runoff occurring (not infiltrating the profile) at high intensity rainfall events. In some scenarios runoff will occur when soil moisture content is high, and this is more likely in the irrigated scenario due

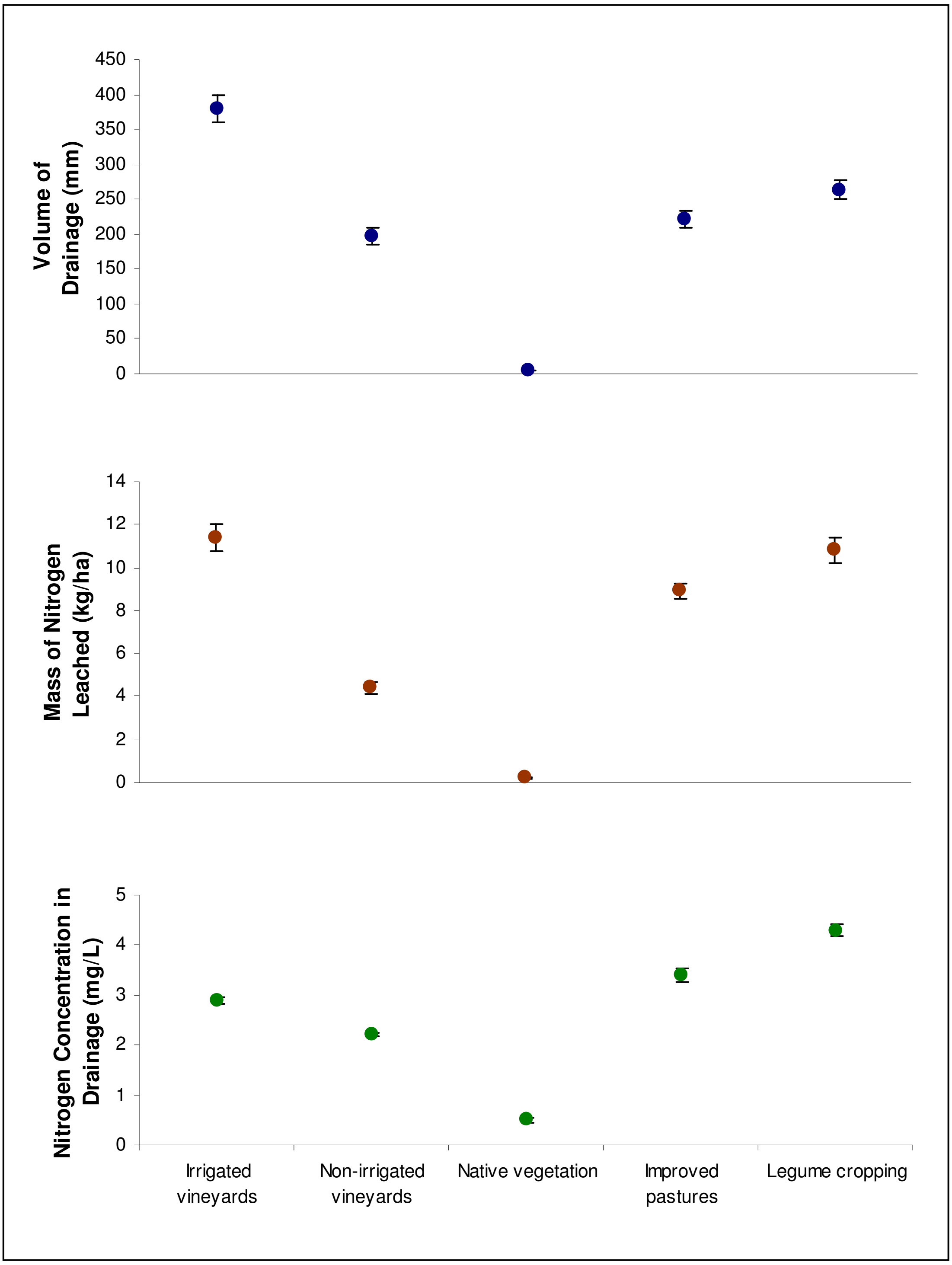


Figure 7.3: **The average annual drainage and nitrate leached below the root zone from the modelled scenarios (with standard errors)**

Table 7.5: The mean annual drainage and nitrogen leaching from the root zone for each of the five land use scenarios as calculated by the LEACHN model

Scenario	Infiltrating Rainfall (mm/yr)	Irrigation (mm/yr)	Drainage below the root zone (mm/yr)	Mass of nitrogen leached below the root zone (kg/ha/yr)	Concentration of nitrate in drainage water (mg/L)
Irrigated vineyards	599.2 (*n=56*, *SD=108.26*)	273.0 (*n=56*, *SD=168.70*)	380.1 (*n=56*, *SD=147.17*)	11.4 (*n=56*, *SD=4.95*)	2.9 (*n=56*, *SD=0.48*)
Non-irrigated vineyards	604.0 (*n=56*, *SD=108.37*)	-	197.2 (*n=56*, *SD=95.86*)	4.4 (*n=56*, *SD=2.1*)	2.2 (*n=56*, *SD=0.29*)
Native vegetation	614.7 (*n=280*, *SD=110.62*)	-	4.9 (*n=280*, *SD=10.29*)	0.2 (*n=150*, *SD=0.28*)	0.5 (*n=280*, *SD=0.73*)
Improved pastures	616.4 (*n=56*, *SD=110.48*)	-	221.5 (*n=56*, *SD=96.59*)	8.9 (*n=56*, *SD=2.79*)	3.4 (*n=56*, *SD=0.91*)
Legume cropping	616.4 (*n=56*, *SD=110.48*)	-	263.9 (*n=56*, *SD=97.45*)	10.8 (*n=56*, *SD=4.25*)	4.3 (*n=56*, *SD=0.79*)

to its higher soil moisture content. In the irrigated vineyard scenario, mean annual runoff is approximately the difference between it's infiltrating rainfall and that for cropping and improved pasture (where no runoff occurred). This equated to an estimated annual nitrogen runoff of approximately 0.1 kg N/ha, which is considered to be insignificant when compared to the other calculated fluxes.

While illustrating that there are significant differences between leaching of nitrogen under the different scenarios, Figure 7.3 shows that nitrogen leaching is not simply driven by the drainage volume through the profile. This is apparent for the three scenarios of non-irrigated vineyards, improved pasture and legume cropping. While these three scenarios have similar annual drainage below the root zone, the mass of nitrogen leached is markedly different. This indicates that nitrogen cycling within the soil is substantially impacting on the concentration of nitrogen being leached.

7.4.2 Simplified nitrogen budgets under different scenarios

The drainage and leaching results report on the mean fluxes from the environmental scenarios over the term of the model. To provide further information of the behaviour of nitrogen within these scenarios, a simplified nitrogen budget was developed for each scenario.

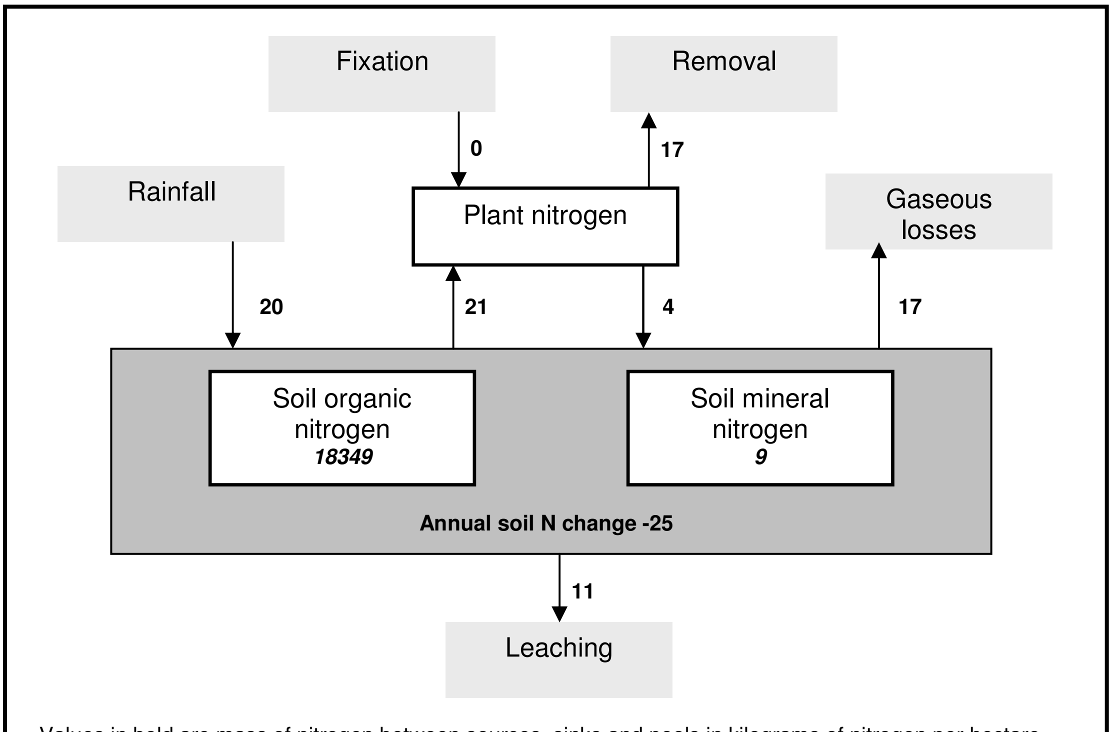
The budget is presented in the same format for each scenario: but not all inputs and losses are relevant for each scenario. Each simplified nitrogen budget reports the following:

Inputs-	***Rainfall***	Nitrogen content in rainfall
	Fixation	The conversion of atmospheric nitrogen to ammonium ions
	Plant residues	The return of organic nitrogen to the soil nitrogen pool
Losses-	***Leaching***	The loss of mineral nitrogen beyond the soil profile
	Gaseous losses	The loss of nitrogen in gaseous form from the volatilisation of ammonia, and the denitrification of nitrate
	Removal	The net change in nitrogen in plants (primarily the loss of organic nitrogen from harvesting or grazing)
Storages-	***Soil organic nitrogen***	Organic bound nitrogen within the soil
	Soil mineral nitrogen	Ammonia and nitrate (and nitrite) within the soil
	Plant nitrogen	Nitrogen within living plants

As discussed above, the loss of water and nitrogen in runoff is considered negligible and therefore is not reported in the nitrogen budgets. Also, as reported in Chapter 6, atmospheric depositions (or windblown transport) of nitrogen are also consider to be small. Both of these external fluxes are considered to be within the uncertainty of the modelled data. Fertilisation applications are not included in any of the scenarios, and therefore this is also not included in any of the reported nitrogen budgets.

Irrigated vineyards

The nitrogen budget and annual fluxes for the irrigated vineyard scenario are presented in Figure 7.4.



Values in bold are mass of nitrogen between sources, sinks and pools in kilograms of nitrogen per hectare per year. Values in bold-italics are soil storages. Annual removal and soil change is calculated by difference.

Figure 7.4: A quantified nitrogen budget for irrigated vineyards using LEACHN

The irrigated vineyard nitrogen budget incorporates the additions of nitrogen in irrigation water within the 'Rainfall' source. The contribution of nitrogen (as nitrate) through irrigation is based on the volumes calculated in the methodology section, and assumes that the nitrate concentration in irrigation water was 5 mg/L. The concentration of nitrate in irrigation water is estimated from the groundwater concentrations within the central part of the study area (where the majority of viticulture is undertaken). The contribution estimated using these assumptions is 17 kg N/ha/yr which, in net terms, is the majority of the viticultural crop usage. While there are proportionally higher leaching loads from this scenario compared to some of the other scenarios, the modelling predicts that irrigation would contribute to a reduction in the concentration and mass of nitrate in groundwater (assuming the irrigation water is being removed from the unconfined aquifer below the landuse). From Table 7.5 the model predicts that each year 273 mm of groundwater was extracted (with a nitrate concentration of 5 mg/L) and replaced by 380.1 mm drainage having a nitrate concentration of 2.9 mg/L.

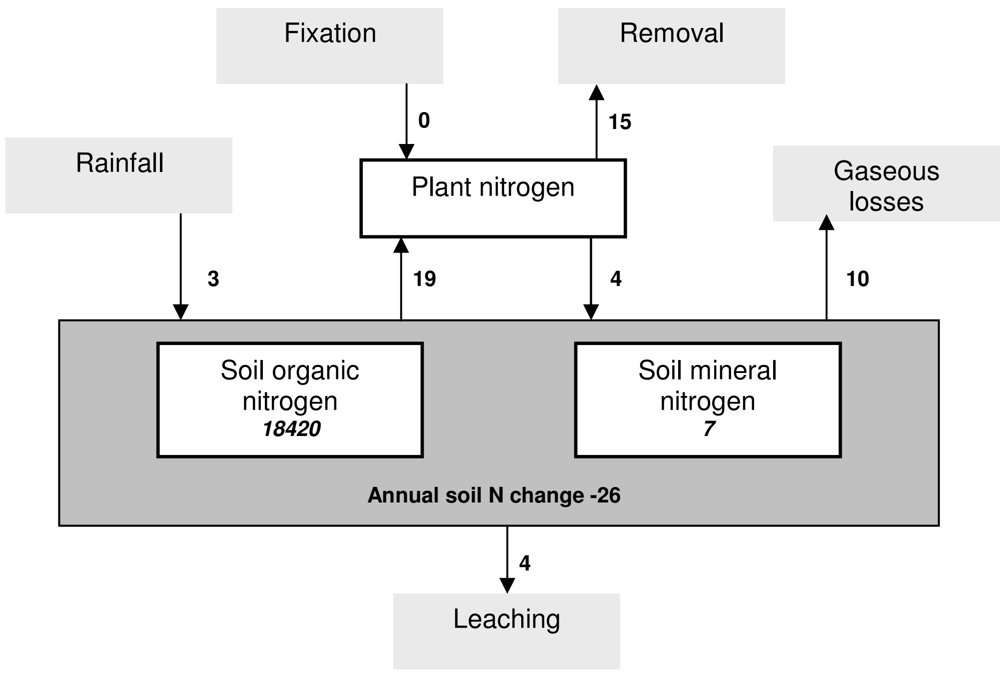
Chapter 2 reported that groundwater levels in the surface aquifer are decreasing in recent years, which appears to contradict this model. However, there are other influences on groundwater within the study area, such as surface drainage, various other groundwater extractions within the study area, and, more recently reductions in annual rainfall. The drainage results are higher than the indicative estimate of 235±8mm reported by Pudney (2007) in assessing the recharge under frost irrigation within the study area. While Pudney outlined that their model may not be appropriate for the vineyard growing season, their more detailed research suggests that the modelled recharge reported by this scenario simulation is high.

Non-irrigated vineyards

The modelled nitrogen budget and annual fluxes for the non-irrigated vineyard scenario are presented in Figure 7.5.

The non-irrigated vineyard scenario was constructed so that its only difference to the irrigated vineyard scenario was that there is no irrigation water. The resulting impact on the predicted nitrogen budget was a reduction in the fluxes between nitrogen pools. This is driven by reduced plant vigour, due to reduced water availability, as well as a reduction in the application of plant-available nitrogen from the irrigation water.

In the model, the reduction in available water also reduces the net mineralisation rate and therefore the reduction in soil organic nitrogen is less than for the irrigated vineyard scenario.



Values in bold are mass of nitrogen between sources, sinks and pools in kilograms of nitrogen per hectare per year. Values in bold-italics are soil storages. Annual removal and soil change is calculated by difference.

Figure 7.5: A quantified nitrogen budget for non-irrigated vineyards using LEACHN

Native vegetation

Figure 7.6 presents the nitrogen budget and annual fluxes for the native vegetation scenario.

In this scenario the losses from the system are comparatively small, and reflect the nutrient conservative nature of native vegetation in Australia (Adams and Attiwill 1984, Mulligan and Sands 1988, Attiwill and Adams 1993). The major flux shown in the nitrogen budget is the return of plant

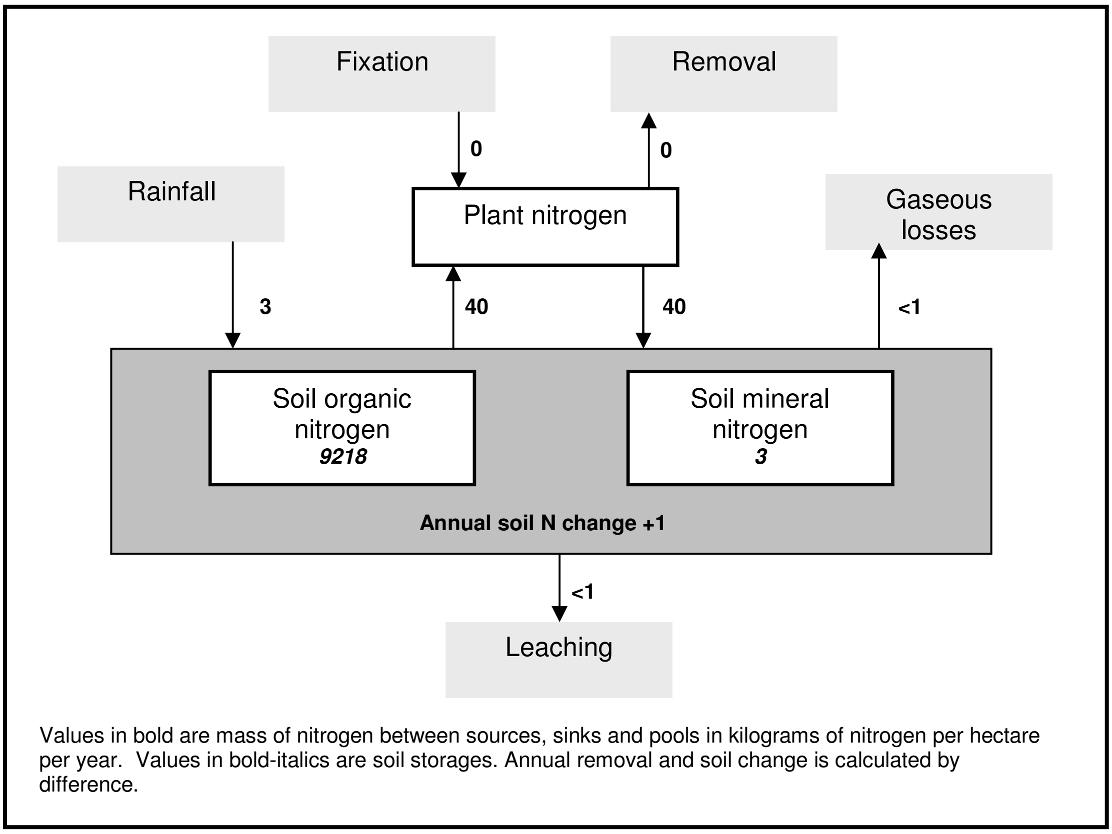


Figure 7.6: A quantified nitrogen budget for native vegetation using LEACHN

residue to the soil organic pool, and this reflects an annual return of plant material through litterfall. Litterfall plays a major role in recycling of nutrients and maintaining soil fertility in native vegetation systems (Grierson and Adams 1999, Lindsay and French 2005). The litterfall values are at the lower end of the range reported for *Eucalyptus* forests in Victoria (38-84 kg N/ha/yr);(Polglase et al. 1992).

In this scenario litterfall is a significant contributor to nitrogen cycling, and because the native vegetation is not being harvested, the harvest component is zero. Under undisturbed conditions it would thus be predicted that native vegetation would increase the soil organic nitrogen pool and not quickly reach a steady state (Turner and Lambert 2002).

The main input of nitrogen into Australian forests is expected to be through fixation (Adams and Attiwill 1984), however this scenario did not include fixation. The model simulated a steady state due to the conservative nitrogen recycling (minimal leaching or gaseous losses) and the moderate soil organic pool. The exclusion of nitrogen fixation may not be appropriate for higher production forests or where the soil is depleted in nitrogen.

Pasture grazing

This is a more complex scenario that incorporated the growth of a mixed species pasture, and included nitrogen fixation and grazing impacts. The calculated nitrogen budget and annual fluxes for the pasture grazing scenario are presented in Figure 7.7.

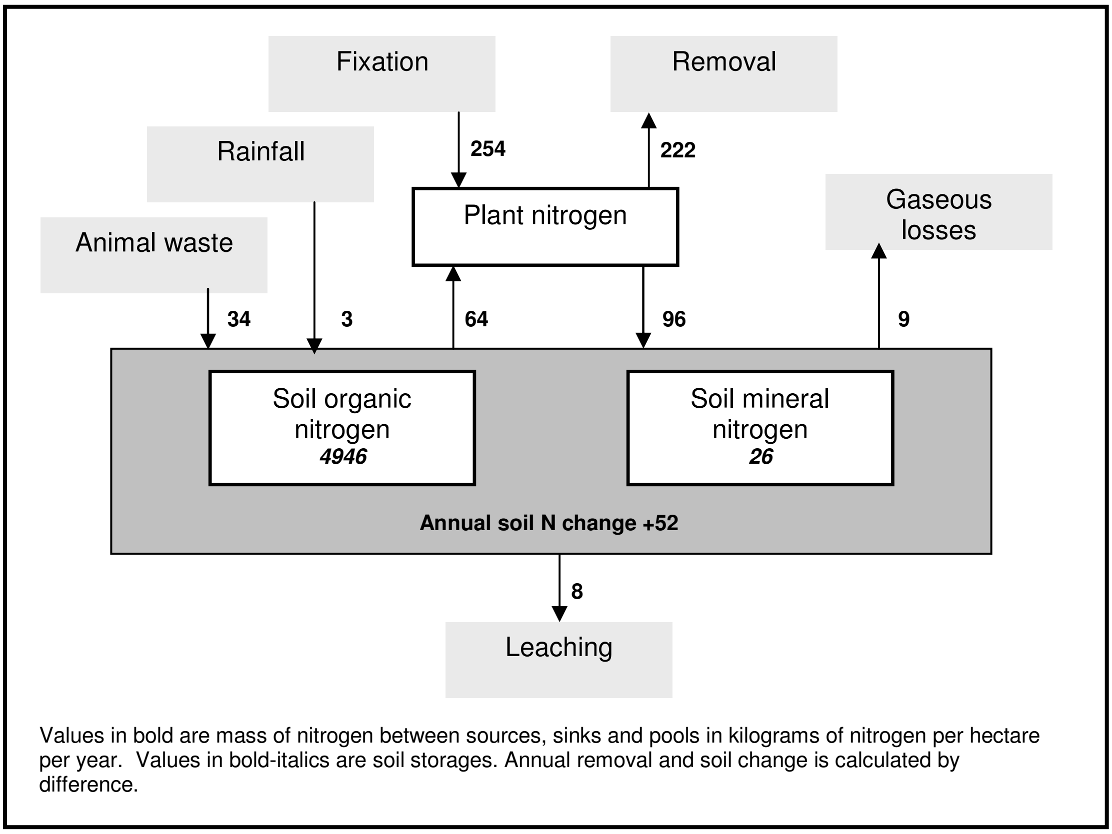


Figure 7.7: A quantified nitrogen budget for pasture grazing using LEACHN

The modelled scenario reports relatively high fixation rates (254 kg N/ha/yr) when compared to other non-irrigated mixed pasture studies (100 kg N/ha/yr);(Pakrou and Dillon 2000).

Figure 7.7 also incorporates the addition of urine and manure during stock grazing. The application is, in effect, applied evenly over the profile. Pakrou and Dillon (1995) showed that high leaching rates occurred under urine patches, due to the high localised applications. This scenario produced similar leaching results to their work as the urine-sourced nitrogen was strongly leached.

A high application of urine is reflected in the relatively high concentration of inorganic nitrogen in the soil. This elevated level of inorganic nitrogen increases the risk of leaching of nitrogen beyond the root zone.

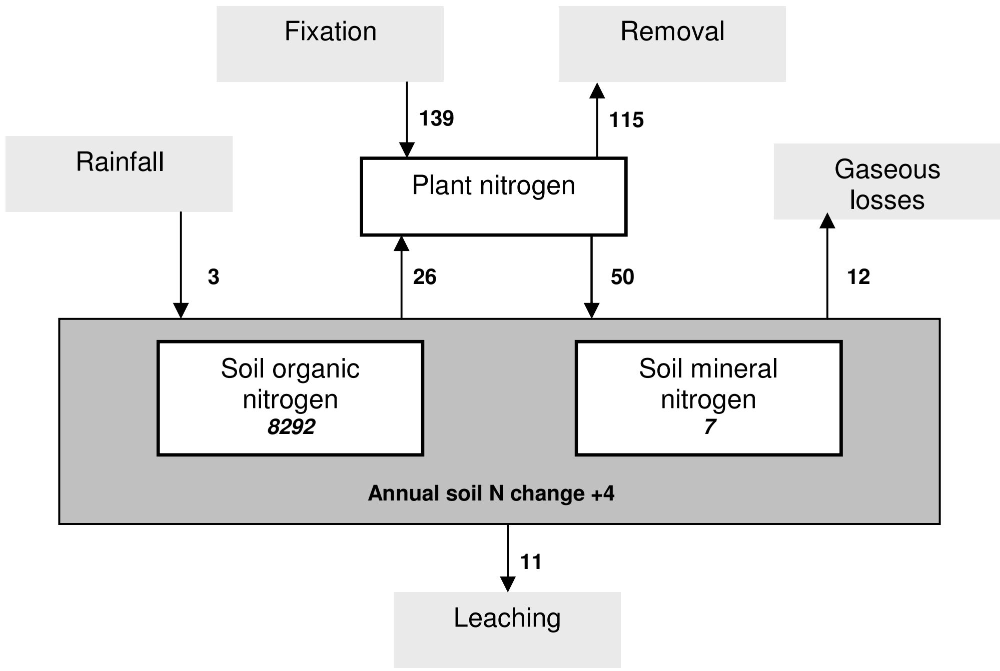
The model predicted volume recharge rates that were above those predicted in Chapter 5 (221.5 mm/yr), but which were consistent with other Australian studies (White et al. 2003).

When compared to other Australian studies, the mass of nitrogen leached from the pasture is low compare to that of Pakrou and Dillon (2000) (i.e. 81 kg N/ha/yr), but close to those reported by Anderson and his colleagues (1998) for leaching under lucerne pastures (12 kg N/ha/yr), and within the range of Melland and his colleagues (2008) for sheep grazed pastures (3.2-10.6 kg N/ha/yr).

Legume cropping

The fifth scenario modelled was leguminous cropping, and the nitrogen budget and annual fluxes for this scenario are presented in Figure 7.8.

This scenario predicts that there is an increase in the nitrogen pool – a result expected due to the fixation capacity of the crop. The degree of increase is small, and an explanation for this may be the proportionally high removal of plant nitrogen on an annual basis: it is common that the majority of the above-ground plant material is either harvested or completely burnt after harvest. The root fraction of the crop is maintained (hence plant residue being reported in Figure 7.8), and the mineralisation of these roots after



Values in bold are mass of nitrogen between sources, sinks and pools in kilograms of nitrogen per hectare per year. Values in bold-italics are soil storages. Annual removal and soil change is calculated by difference.

Figure 7.8: A quantified nitrogen budget for legume cropping using LEACHN

harvest and prior to the next year's crop allows leaching of some of the inorganic nitrogen.

The higher drainage predicted in this scenario (and potentially the higher leaching of nitrogen) may be the result of the assumption that after cropping there is no vegetation cover before the next planting. It is usually the case that there is no planted crop, but grass and broadleaf pasture species (and weeds) grow during this period. Given that the model assumed no vegetation over this period, it may overestimate drainage between January and May, however the predicted drainage is consistent with other Australian studies on drainage under various agricultural cropping regimes (White, et al. 2003).

7.4.3 Mineralisation rates in soil

The rate of mineralisation of organic nitrogen in the soil is an important component of the nitrogen cycle and determines rates of soil nitrogen loss (particularly leaching). Table 7.6 presents the average annual mineralisation calculated for the five scenarios.

The modelled results reflect that the mineralisation of plant residue is preferred over humus (as humus is not as easily mineralised; Hutson 2003). High mineralisation modelled in the pasture and cropping scenarios reflect the annual die-off of the clover and Faba Beans respectively, and the mineralisation of their root-bound nitrogen. Studies have indicated that more than half the residue from these types of legumes will mineralise within six months (Agehara and Warncke 2005).

Table 7.6: The net organic nitrogen mineralisation (N kg/ha/yr) predicted for each scenario

Scenario	Annual mineralisation of plant residual organic nitrogen	Annual mineralisation of humic organic nitrogen	Annual mineralisation of manure organic nitrogen	Annual total mineralisation of organic nitrogen
Irrigated vineyards	4	32	0	36
Non-irrigated vineyards	4	27	0	31
Native vegetation	28	10	0	38
Pasture grazing	96	7	15	118
Legume cropping	50	13	0	63

The moderately high mineralisation for the native vegetation scenario is within the reported range of other studies (Attiwill and Adams 1993).

The slightly higher mineralisation for the irrigated compared to the non-irrigated vineyard scenario is consistent with the understanding that mineralisation of soil organic nitrogen can be limited by soil moisture (Paul et al. 2003), however the comparison of the relative values should be undertaken with caution given the accuracy of the model. The lower returns (mass) of plant residues to soil in these scenarios are reflected in the lower mineralisation rates.

7.4.4 Predicted groundwater nitrate profiles

The time-series leaching of nitrate below the root zone was modelled for the five scenarios to investigate whether the nitrate concentrations in drainage water could impact upon groundwater nitrate concentration profiles. As discussed in Chapter 4, the upper portion of the unconfined aquifer is dominated by vertical drainage. The predicted groundwater nitrate profiles (Figure 7.9) under each of the scenarios were calculated from the volume of drainage and the mass of nitrate leached.

The groundwater nitrate profiles were modelled using annual summaries from 1950 to 2005, and assumed a saturated aquifer porosity of 40%. Figure 7.9 illustrates that there is greater inter-annual variations in those scenarios where annual vegetation is present (i.e. cropping and grazing). The modelled profiles for the vineyard scenarios display similar results, with both showing little variability.

The predicted groundwater nitrate profile for the native vegetation scenario is markedly different, and is substantially influenced by the small recharge volumes reaching the aquifer. The proportionally smaller recharge rates were predicted to only contribute 0.7 m to the aquifer over the 56 year period (compared to approximately 53 m of recharge for the irrigated vineyard scenario).

The groundwater nitrate profiles are theoretical profiles presented to describe the variability of groundwater nitrate that may occur under the different modelled scenarios. The profiles are a simplified representation, and are not

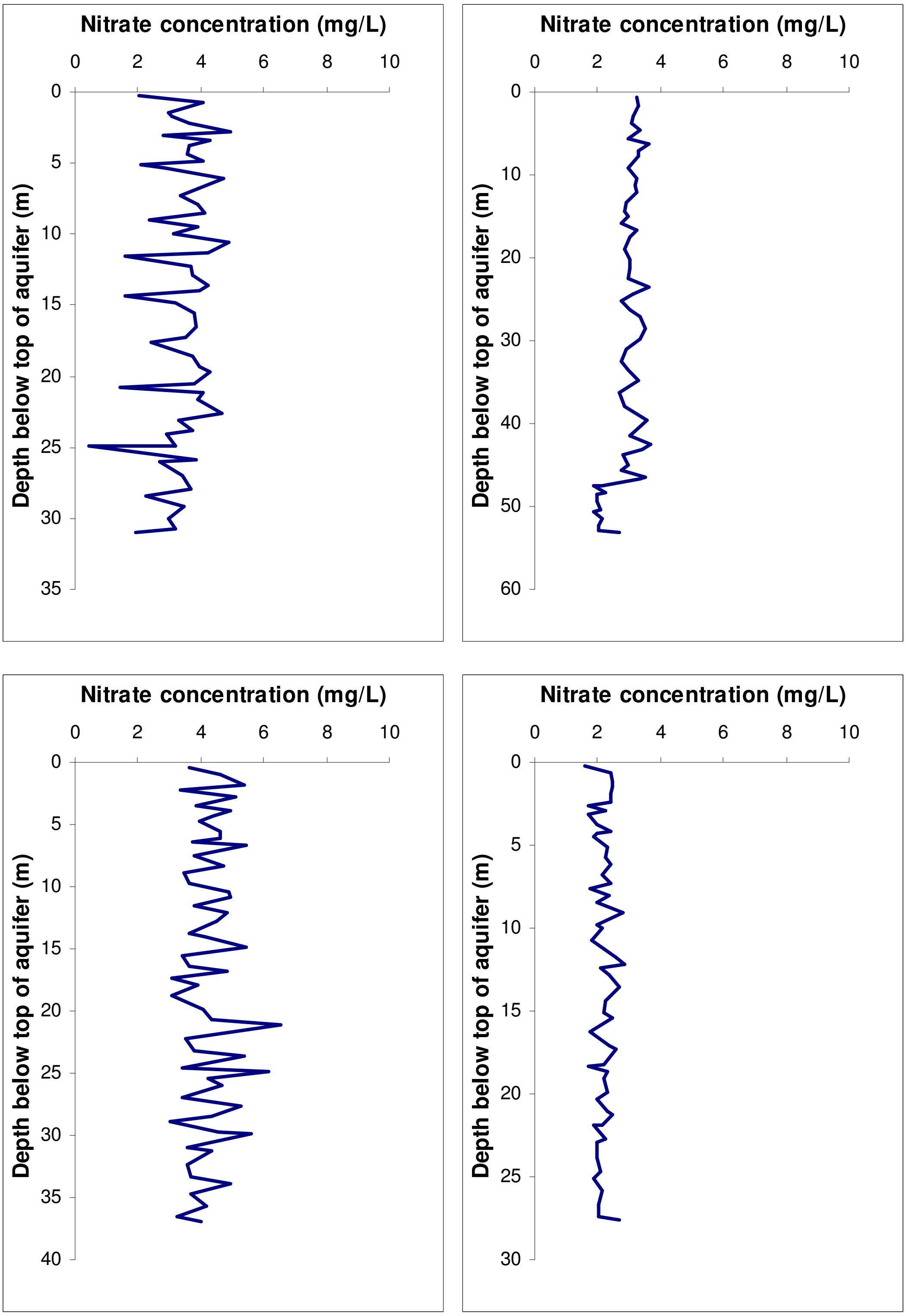


Figure 7.9: Modelled groundwater nitrate profiles; Grazing (top-left), Irrigated vineyards (top-right), cropping (lower-left), non-irrigated vineyards (lower-right)

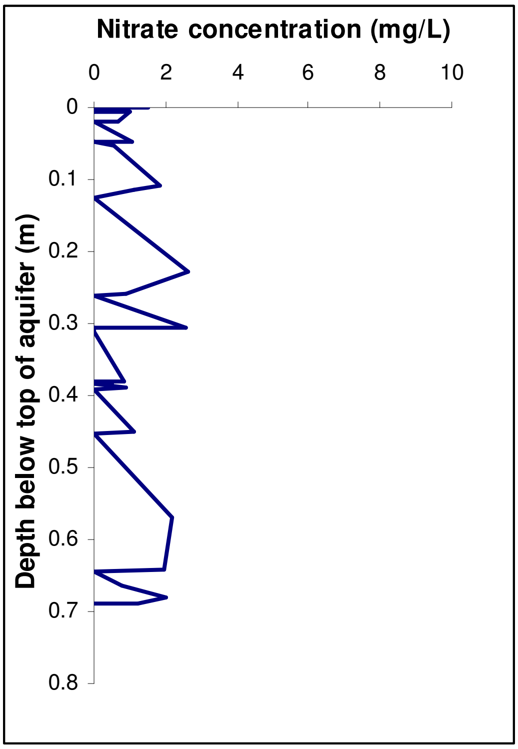


Figure 7.9: (cont.) Modelled groundwater nitrate profiles; Native vegetation

intended to incorporate the effects of denitrification, dismissivity or diffusion within the aquifer.

7.5. DISCUSSION

7.5.1 Consideration of modelled scenarios

As has been observed "All models are wrong but some are useful" (Box 1979). Any consideration of the reliability of modelling should recognise the necessity for the assumptions and simplifications that models bring to complex and variable environments. The present modelling results should be considered in this context. For instance, while the models have produced systems approaching a steady-state and reduced variability in results as shown by the standard error bars in Figure 7.3, these do not necessarily indicate they accurately replicate the environmental system. The accuracy of the model can be evaluated through comparisons to other studies. A summary of the predictions of the modelled scenarios against other information is presented below.

The water balance presented by the model shows that there is considerable variability between the different landuse scenarios. The volumes of recharge estimated are substantially higher than those predicted using the tritium isotope technique (with the exception of the native vegetation scenario). This is most notable in the vineyard areas, although as suggested within Chapter 5 the tritium technique is unlikely to be appropriate where considerable irrigation recycling of groundwater is occurring.

The volume of recharge is comparatively high (up to 380 mm/yr) compared to other studies in which annual recharge across or near the study area is estimated to be between 20 - 120 mm (De Silva 1994, Bradley, et al. 1995, Cobb and Brown 2000, SECWMB 2001). They are more comparable with the range of 50 – 270 mm/yr reported for south of the study area (Allison and Hughes 1978). The model showed that recharge rates have decreased in recent years (i.e. they are 10% lower over the last 20 years of the model) reflecting the lower rainfall. This dry period may be reflected in lower recharge for some of the studies above. While the model predictions are relatively high, there is evidence of relatively rapid recharge in the study area; Figure 2.12 shows an increase in groundwater height of approximately 3 metres during winter. While some of the increase at this site may be recovery from summer extraction (the well is a windmill for a stock trough), the degree of recovery indicates that recharge rates are locally high within the study area.

As previously discussed, modelling of nitrogen cycling under the landuse scenarios has produced results that are considered to be realistic for the study area. Particularly related to the pasture and grazing scenarios, the predicted rates of nitrate leaching are consistent with other Australian studies (Ridley et al. 2001, McCaskill et al. 2003, White, et al. 2003, Melland, et al. 2008). In considering predictions of the model, the following limitations must be acknowledged in addition to the assumptions already discussed.

Point sources

The modelled scenarios are representative of broad-scale landscape activities within the study area, and therefore do not incorporate effects of any nitrogen point sources. At a local scale, point sources may dominate the diffuse nitrogen sources to groundwater, however their impact on the overall nitrogen budget for the study area has not been investigated.

Disturbance

There are a range of disturbances that can impact on landscapes, and the modelling did not attempt to investigate these. Anthropogenic disturbances (e.g. irregular heavy cultivation, fertiliser application, pasture regeneration) could significantly impact on mineralisation and/or soil nitrogen pools. Further, natural disturbances can also substantially impact on nitrogen cycling within native forests (Attiwill 1994). In a study of subalpine *Eucalyptus* forests it was estimated that the wildfire combustion of litter and understorey resulted in nitrogen loss (via volatilisation) of 74-109 kg N/ha (O'Connell 1989).

Due to the irregular and (generally) unpredictable nature of these disturbance events, it is difficult to incorporate them into a model. It is acknowledged however that they may result in alterations to the nitrogen cycle.

Variability of land management

The model assumes that the land management practices defined in each scenario model are consistently applied by land owners. Based on field observations land management practices vary substantially across the study area. For example, irrigation practices (and application methods) vary for vineyards, and frost fans rather than irrigation are being increasingly used for frost protection. Further, the inter-row management of vineyards varies considerably, ranging from leaving them fallow, to application of straw-mulch and growing cover crops (Figures 7.10-7.13).

Nevertheless, each scenario presented reflects a generic land management regime that is intended to represent the most common practice for the landsystem. Variations in land management practices are likely to impact on water and nutrient cycling.

Figure 7.10: Examples of variations in vineyard management; (1) a cover crop and frost fans

Figure 7.11: Examples of variations in vineyard management; (2) a mown/herbicide-sprayed cover crop

Figure 7.12: Examples of variations in vineyard management; (3) no inter-row crop

Figure 7.13: Examples of variations in vineyard management; (4) stock grazing part-cover crops

7.5.2 Implications of model outputs

While recognising the assumptions and limitations of the model, the simulations produce results that support earlier conclusions in this thesis concerning the sources of nitrate to groundwater.

In all five land system scenarios, the leaching of nitrogen to groundwater was generally low (less than 12 kg N/ha) and the concentration of nitrogen in recharge water was below 5 mg/L (Table 7.5). At the study area scale, these inputs alone would not result in the high groundwater concentrations reported in Chapter 3.

Chapter 5 reported that there was little correlation between measured nitrate concentrations in groundwater and the broad landuses in the study area (although vineyards did show some relationship). However, there did appear to be a relationship relating to proximity to point sources. Modelling of the five scenarios supports this by suggesting that the landuses are not significant sources of nitrogen to groundwater. The scenarios indicate that even vineyards are not a significant source, and it may be that the relationship reported between the vineyards and elevated nitrate concentrations is not a cause-effect relationship.

The land uses do not provide a significant source of nitrogen to the groundwater as they occur in generally nitrogen limited environments (although not in the case of legume cropping when during January to May there is no crop). In the model there is no nitrogenous fertiliser application to these systems, and the only application of nitrogen is rainfall (minimal), irrigation and nitrogen fixation. Nitrogen fixation is a major input into the pasture and crop regime, however this is primarily due to the deficit between plant-available nitrogen in the soil and plant requirements. Other studies have demonstrated that nitrogen fixation in pasture and cropping will be inhibited where there is a readily available source of soil nitrate (Guo et al. 1992, Peoples, et al. 1995). The high fixation rates modelled support the suggestion of nitrogen-limited plant environments. While the manure and urine applications in the pasture scenario can be considered recycling, the urine is highly leachable, and in the simulation contributes significantly to the leached nitrogen. This further suggests that plant available nitrogen is limited for the modelled scenarios.

In the vineyard scenarios it is reported that there is a reduction in the soil organic nitrogen pool. In these instances the mineralisation of soil organic nitrogen is likely to be a significant source of nitrogen for plant uptake and nitrogen leaching.

In all scenarios there is considerable cycling of nitrogen within the soil profile, with mineralisation rates of 31-118 kg N/ha/yr. This rate of mineralisation of inorganic nitrogen in the soil profile is comparable to recommended fertiliser application rates for pastures (S Haase, pers. comm. 2007). The outcome of the scenario modelling indicates that mineralisation is likely to be a significant source for diffuse nitrate leaching to groundwater. This is primarily due to this source of inorganic nitrogen in the soil profile being dominant over all other sources (e.g. rainfall, irrigation).

The model predictions are consistent with the results of the nitrogen isotope studies (Chapter 6) which indicated that the isotopic signatures for nitrate in some samples were consistent with them being sourced from mineralisation of soil organic nitrogen.

7.6. CONCLUSIONS

Modelling of five 'generic' landsystem scenarios appropriate to the study area using the LEACHN one-dimensional leaching model and literature information has allowed comparison of their soil-nitrogen cycles.

A challenge in applying this, or any other model, is the environmental variability present in the system being studied (e.g. soil properties, land management practices). This limitation is potentially off-set by the ability to undertake modelling without having to first undertake resource and time intensive field testing and sampling.

While recognising its limitations, the results of LEACHN modelling support the earlier findings of this study that diffuse sources do not appear to be a significant source of nitrogen to groundwater in the study area.

Nitrogen fixation in pasture and legume cropping is expected to be the main nitrogen input into these landsystems, given that application of nitrogenous fertilisers is not wide spread.

In all simulations, the organic nitrogen pool dominated the inorganic nitrogen pool. The mineralisation of this organic nitrogen (from plant residue and soil organic matter) results in appreciable generation of soil inorganic nitrogen in all scenarios. While the mass of inorganic nitrogen within the soil is often low (orders of magnitude less than the organic nitrogen pools), this is a continuous source of inorganic nitrogen to plants (and for leaching).

Modelling suggests that, where leaching of nitrogen is occurring within these landsystems, it is likely that it is sourced from the mineralisation of soil organic matter. This conclusion is consistent with the assessment of nitrogen isotopes reported in Chapter 6.