Quantifying and reducing nitrogen leaching under intensive vegetable production in Horowhenua

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Abstract

Vegetable growers often need to apply relatively large quantities of nitrogen (N) fertiliser to successfully grow their crops, and this can lead to large losses of N via runoff and/or leaching to receiving water. Growers are under increasing pressure to reduce N leaching. Therefore, practices that increase N use efficiency and mitigate N losses are receiving increasing attention from growers, regulators, and scientists both in New Zealand and worldwide. For instance, high N levels have been observed in the Arawhata stream and Lake Horowhenua in the Arawhata catchment, where most of the fresh vegetables consumed in the lower North Island of New Zealand are grown.

There is currently little research data available which quantifies N leaching from different practices of vegetable production systems in New Zealand, including for the Arawhata catchment. Therefore, the first objective of this study was to identify the likely range of N leaching rates under vegetable production in the Arawhata catchment. Nitrate-N in soil and N losses were measured under a series of crops at two field trial sites which were established on two local farms: a site with a potato-onion rotation (Potatoes site) and a site under a beetroot-Pak choi rotation (Green vegetables site). Soil nitrate-N was sampled at depths of 0-30 and 30-60 cm at the Potatoes site, and at 0-20 cm at the Green vegetables site. Mean soil nitrate-N in the topsoil (0-20 and 0-30 cm) ranged between 9.3 and 18.3 ppm, while mean soil nitrate-N in the subsoil (30-60 cm) ranged between 7.3 to 9.6 ppm. N leaching was measured using suction cups and lysimeters. Mean nitrate-N concentrations in soil water ranged between 16.9 to 61.9 mg L^{-1}, and N leaching ranged from 95 to 225 kg N ha^{-1}. These field measurements of N leaching were used to calibrate and validate the APSIM Next Generation model. APSIM simulates N dynamics, including leaching, under crops at the paddock scale. APSIM was then used to investigate N leaching rates under a wider range of climate conditions (31 years), the effect of soil type on leaching, and to quantify the ability of different mitigation strategies to reduce N losses under intensive vegetable farming in the region. The mitigation scenarios at the Potatoes site included: potato harvest at maturity and ryegrass cultivation during the drainage season, and potato harvest at maturity and cultivation of winter catch crops. The mitigation scenarios at the Green vegetables site included: potential of drainage management, a range of alternative cover crops and catch crops instead of ryegrass, and other crop rotations. These mitigation scenarios resulted in an N leaching reduction of 7 to 47% in the Potatoes site, and of 6 to 52% in the Green vegetables site. The final objective of this research was to evaluate the impact of field-scale mitigation measures on the Arawhata stream by coupling the N losses simulated in APSIM with a catchment-scale model (SWAT+). This simulation exercise suggested that by considering crop rotations that use less N fertiliser and frequently grow cover crops, N loads in the Arawhata stream could be reduced by 17% to 21%.

Keywords: nitrate losses, water quality, diffuse pollution, vegetable farming, mitigation, APSIM, crop rotations, cover crop, N fertiliser, SWAT

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List of Abbreviations

APSIM	Agricultural production systems simulator
ANOVA	Analysis of variances
BPM	Best management practices
C	Carbon
CRF	Controlled release fertiliser
DON	Dissolved organic nitrogen
EMM	Estimated marginal means
GAMLSS	Generalised additive model for location, scale and shape
GIS	Geographic information system
GMP	Good management practices
N	Nitrogen
N_{org}	Organic nitrogen
NO_x	Nitrogen oxides
NO_3-N	Nitrate nitrogen
NUE	Nitrogen use efficiency
NSE	Nash-Sutcliffe efficiency
SON	Soluble organic nitrogen
SRF	Slow released fertilisers

SWAT	Soil and water assessment tool

Chapter 1: Introduction

1.1 Background

As losses from agricultural fields are often of the order of 50% of applied nutrients, agriculture has been identified as the main diffuse source of the pollutants that degrade aquifers and water bodies (Fageria & Baligar, 2005; Stewart & Lal, 2017; Wang & Li, 2019). Therefore, increasing nitrogen (N) use efficiency and mitigating N losses have become key challenges to sustainable agriculture. Farmers need practices that, simultaneously, maintain or increase food production, and protect water quality. This is particularly the case for vegetable production systems where nitrogen fertiliser management and nitrogen leaching are problematic.

In New Zealand, N fertiliser use has become widespread since 1990, with artificial fertiliser increasingly replacing much of the nitrogen fixed by clover in pastoral soils (Di & Cameron, 2002; Ledgard et al., 1990). New Zealand has now become the world's 29th largest user of N fertiliser for agricultural purposes (Food and Agriculture Organization of the United Nations, 2020), which is surprising given the country's relatively small size. In fact, N fertiliser application increased 627% between 1991 and 2019, with urea as the most imported and applied fertiliser (Statistics NZ, 2021). The Ministry for the Environment (2015) estimated that the total N leached from soils countrywide increased 22% between 1990 and 2012.

There is growing concern about freshwater pollution and water quality in New Zealand. According to available official statistics, 46% of the country's monitored lakes had poor or very poor median Trophic Lake Index (TLI) values –a nutrient enrichment parameter– between 2016 and 2020 (Statistics NZ, 2022a). Rivers are undergoing a similar degradation process. Around 69% of monitored rivers showed median concentrations of nitrogen above the expected natural conditions between 2016 and 2020 (Statistics NZ, 2022b), which jeopardizes natural biodiversity. Several documents highlight the deteriorating state of the quality and quantity of national water systems (Ministry for the Environment, 2007; Morton, 2017; Statistics NZ, 2022a). A sound understanding of New

Zealand's water quality and nutrient leaching, along with quantitative information, is highly valuable for regulatory frameworks and identifying the management practices required to avoid further environmental damage.

Vegetable production systems have particularly intensive fertiliser and water inputs because of their intrinsic high N demand and high growth rates. Consequently, vegetable production may result in low nutrient efficiencies and large nutrient losses to the environment (Agneessens et al., 2014; Cameira & Mota, 2017). Vegetables are normally harvested while they are still growing and N uptake is ongoing, leaving a significant amount of mineral N in the soil profile (Cameira & Mota, 2017). Moreover, the situation can be worsened if vegetable crop residues are incorporated, due to their high N mineralisation rate (Chaves et al., 2007; De Neve & Hofman, 1998).

However, actual leaching losses of N from intensive vegetable production systems in New Zealand are not yet well understood due to the limited amount of data available. More studies are needed to quantify the magnitude of N losses from intensive vegetable production systems, and to identify and develop mitigation practices that reduce nitrogen leaching rates.

1.2 Rationale of study

Lake Horowhenua (40.6100°S, 175.2547°E) is a 290-hectare shallow coastal located near Levin in the lower North Island, New Zealand (Figure 1.1). The lake is fed by several tributaries, most notably the Arawhata stream. There are also several small water courses and marshes, and one main outlet stream, the Hokio stream (Gibbs, 2011; White, 1998). It has always been in Muaūpoko iwi (tribe) ownership, and alongside Lake Papaitonga, it has immense importance in their world view (White, 1998; Winiata, 2009). The Lake Horowhenua has historically been used for fishing and recreational purposes, and a range of species have been harvested by local people (White, 1998; Winiata, 2009).

Figure 1.1. Map of the location of Lake Horowhenua and Lake Papaitonga

Nowadays, Lake Horowhenua is one of the most degraded lakes in New Zealand. Land Air Water Aotearoa (2022) reported a mean total N concentration of 3.5 g N m^{-3} in 2021, which does not compare well with the national bottom line (0.8 g N m^{-3}) (Ministry for the Environment & New Zealand Government, 2022). Some studies suggest that this lake is currently in a supertrophic state (Statistics NZ, 2022a), while others suggest that it could even be hypertrophic (Gibbs, 2011). In 2010, Lake Horowhenua had the seventh worst water quality out of 112 surveyed lakes in New Zealand, and it is frequently closed to the public because of the presence of toxic cyanobacteria (Horizons Regional Council, 2017).

Even though numerous efforts have been made to clean up the lake, more information and interventions are needed (Roygard et al., 2015). In the past, the catchment of Lake Horowhenua has been subjected to native tree removal, drainage of swamplands, intensive farming, and about 25 years (1962 - 1987) of direct sewage discharge from the Levin, the largest town in the Horowhenua District, which is adjacent to the lake. There is much debate about what is exactly affecting today's water quality in the lake, but the most likely cause is land use intensification surrounding the Arawhata stream (Gibbs, 2011; Horizons Regional Council, 2017).

For several decades, the Arawhata catchment has supported horticulture, market gardens and dairy farming, often with high rates of fertiliser application (Gibbs, 2011; Horizons Regional Council, 2017). However, there is currently limited knowledge and information in terms of the critical flow pathways, or magnitude, of nutrient losses from surrounding farm land including horticulture and market gardens to the Arawhata stream and the Lake Horowhenua.

To improve water quality in the Manawatu-Whanganui Region, Horizons Regional Council established its 'One Plan', which seeks substantial reductions in nitrogen leaching (Horizons Regional Council, 2014). It has been a particularly important issue for vegetable growers in the Arawhata catchment because it is often very difficult for them to minimise leaching losses. Hence it is imperative to identify strategies and initiatives that will help growers to mitigate nutrient losses under vegetable crops and reduce the amount of N reaching the Arawhata stream and Lake Horowhenua.

A robust identification and evaluation of potential mitigation practices requires a combination of field trials and modelling tools to predict potential N leaching under different practices of vegetable production in the area. Agricultural simulation models have improved over recent time, and they can be used to estimate N leaching from vegetable crops under a wide range of practices and climatic conditions. The APSIM model (section 2.5) has been developed with the intention to simulate ecological and economic consequences of management practices (Keating et al., 2003). As such, APSIM has been extensively calibrated and validated under different crop management scenarios and conditions (Carberry et al., 2002; Probert et al., 1998a; Probert et al., 1998b; Robertson et al., 2002). Nonetheless, it has not been tested or calibrated thoroughly in vegetable agricultural systems in New Zealand conditions (Sharp et al., 2011a; Sharp et al., 2011b). Given APSIM's reliability in a range of countries and conditions, following calibration, it is likely to be able to simulate N losses from vegetable systems at paddock-scale in the Arawhata catchment.

The calibrated and validated, field-scale APSIM model is then coupled with the Soil and Water Assessment Tool Plus model (SWAT+) to explore the impacts of in-field mitigation strategies on catchment-scale in reducing the quantity of N entering the Arawhata stream and the Lake Horowhenua.

Figure 1.2 presents the robust two-pronged approach adopted in this study. Firstly, field trials and modelling were used to quantify N losses under intensive vegetable production in the Arawhata stream and the Lake Horowhenua, and to investigate the effect of a range mitigation practices on N leaching losses from the soil profile. Secondly, a catchment scale model was developed that simulated the likely N load from farm land in the catchment to the Arawhata stream. This model was used to assess the potential of mitigation options to reduce the N leaching to the Arawhata stream.

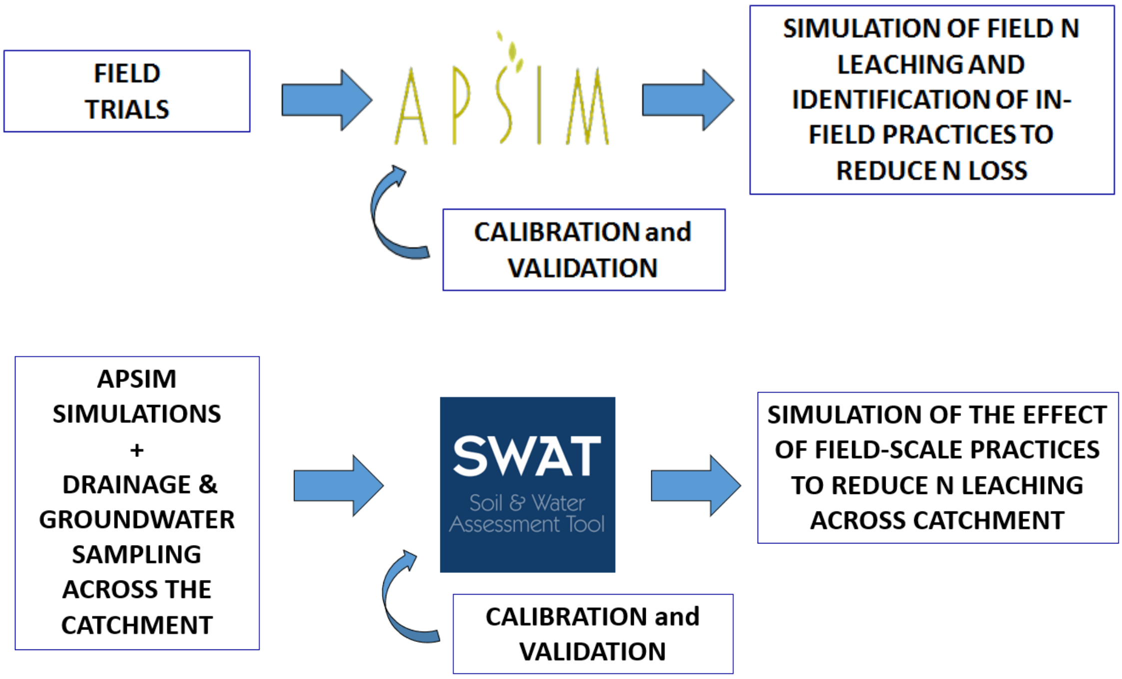

Figure 1.2. Schematic overview of the research approach.

1.3 Research objectives and hypothesis

The main objective of this study was:

- To identify and assess the ability of a range of in-field strategies to reduce N leaching from intensive vegetable farming in the Arawhata catchment.

In order to achieve the main goal, specific objectives include:

- Quantify the amount of N leached from the rootzone in a selection of representative Arawhata catchment paddocks and crops.
- Adapt the APSIM Next Generation model for use in the Arawhata catchment using the results from these field trials (i.e., to parameterise and evaluate this model).
- Investigate a range of mitigation options to reduce N leaching from intensive vegetable production using APSIM.
- Develop a catchment-scale model that can predict the effect of vegetable production on N loads to the Arawhata stream (i.e., by coupling together APSIM, Overseer and SWAT+).
- Investigate the effect of some mitigation measures on N loads to the Arawhata stream.

The main hypothesis for this research is:

- It is possible to measure and model N losses under intensive vegetable production in the Arawhata catchment, and to identify mitigation measures that will reduce N leaching from the soil root zone and N loads to the Arawhata stream.

Chapter 2: Literature Review

2.1 Nitrogen forms, sources, and losses

While atmospheric N is the most abundant form of nitrogen, it is inert and not accessible to most living organisms (Bernhard, 2010; Canfield et al., 2010; Stevenson & Cole, 1999; Xia et al., 2018). In contrast, many different forms of N can be found in soils that are accessible to plants and animals (Table 2.1).

Table 2.1. Forms of nitrogen in soil (Krishna, 2002; Savant & De Datta, 1982).

N pool	**N form**
Organic N	Insoluble humin-N
	Amino acid-N
	Alkali insoluble humin-N
	Hydrolysed ammonia-N
	Amino sugar-N
Inorganic N	Nitrogen and nitrous oxide in soil, air and water
	Nitrite-N reduced in soil
	Nitrate-N
	Exchangeable and fixed ammonium-N in clay

A major fraction of soil N is organic N (N_{org}), constituting about 90% of N in the topsoil of most soils (Stevenson, 1982). Even so, several authors have concluded that this pool has not been properly documented, especially when it comes to bonding to other particles (Jones et al., 2004; Krishna, 2002; Murphy et al., 2000). Proportions of the sub-fractions of N_{org} in the soil may change according to local conditions, but generally amino acid N is more abundant, constituting 30-45% of soil N (Krishna, 2002; Stevenson, 1982). Typically, some N_{org} originates from the breakdown of more complex molecules from deceased organisms or organisms' wastes in a process known as aminization (Bolan et al., 2004; Prasad & Power, 1997). From a reactive standpoint, organic N can be found as soluble organic nitrogen (SON) in soils or as dissolved organic nitrogen (DON) in soil solution and drainage water (Murphy et al., 2000).

Inorganic N, on the other hand, has several forms of N, such as ammonia-N, ammonium-N, nitrate-N, nitrite-N, and nitrous oxide, and most of it is adsorbed to clays, organic colloids or contained in soil air and soil solution (Krishna, 2002). Despite mineral N comprising only a small amount of soil N, it cycles faster than organic N and is essential for its assimilation by organisms (Hadas et al., 1992; Jarvis et al., 1996; Jones et al., 2004; Murphy et al., 2000). As ammonia tends to be retained in soils, dissolved inorganic nitrogen (DIN) is mainly composed by nitrate and, to a minor extent, by nitrite (Hood et al., 2003; Kröger et al., 2007).

Additions of N in soil include the biological fixation of gaseous N, atmospheric deposition (as rainwater or thunderstorms) and fertilisation, whilst removals include plant uptake, leaching, runoff, denitrification, and volatilisation (Murphy et al., 2000; Stevenson & Cole, 1999).

2.2 N transformations and translocations

The N cycle has been relatively well described over the past decades. N transformations are determined by microorganisms and are primarily dependant on reduction-oxidation conditions (Canfield et al., 2010; Falkowski, 1997; Stevenson & Cole, 1999; Xia et al., 2018). Nearly 13 enzymes of different microorganisms are involved in the N cycle which act in different oxidation states (from -III to V), both in aerobic and anaerobic environments (Canfield et al., 2010). These processes are often very dynamic and fast, all or some of them occurring one at a time or simultaneously, or none of them occurring at all (Stevenson & Cole, 1999).

N fixation

N fixation is defined as the integration of gaseous N into the biosphere by microorganisms, normally prokaryotes, and is the main source of N in soils in natural conditions (Bernhard, 2010; Stevenson & Cole, 1999). Microorganisms break down covalent triple bounds of N_2 in an exergonic reaction, reducing atmospheric N to ammonium (Bernhard, 2010; Stevenson & Cole, 1999). Several organisms can fix gaseous N in a wide range of environments, such as cyanobacteria, free-living bacteria and other microorganisms living in symbiosis with non-leguminous plants, but most of the atmospheric N is fixed by nodules of legumes, in which symbiosis take place with bacteria from the genera *Rhizobium* and *Bradyrrhizobium* (Bernhard, 2010; Stevenson & Cole, 1999). The specific elements and mechanisms behind each one of these processes are outside of the scope of this study, but essentially, most N-fixing microorganisms make use of enzyme nitrogenase to carry on subsequent reactions (Bernhard, 2010). Other ways in which N can also be fixed include lightning, atmospheric depositions, and some industrial processes (for instance, burning fossil fuels) (Bernhard, 2010).

Mineralisation-immobilisation

Mineralisation can be defined as the conversion of organic N to inorganic N, whilst immobilisation is the opposite conversion (Delwiche, 1981; Fageria & Baligar, 2005;

Hauck, 1981; Jansson & Persson, 1982; Stevenson & Cole, 1999; Xia et al., 2018). After aminization, N_{org} is converted to ammonia through ammonification process, where heterotrophic microorganisms use organic compounds as an energy source (Fageria & Baligar, 2005; Jansson & Persson, 1982). In the immobilisation process, microbes use inorganic N to build N_{org} as part of their biomass (Jansson & Persson, 1982). The difference between both processes is called net mineralisation or net immobilisation and is essential for nutrient uptake and availability for plants (Jansson & Persson, 1982).

Different factors might influence the microbial activity during mineralisation and immobilisation, but soil moisture, temperature and C/N ratio of organic matter are the most significant (Doran & Smith, 1991; Paul & Juma, 1981; Stevenson & Cole, 1999; Vagstad et al., 1997). Generally, microbial activity starts at >0 °C and increases lineally until plateauing at 25 and 35°C, and later decreasing lineally until reaching no activity at 60 °C (Doran & Smith, 1991; Paul & Juma, 1981; Stevenson & Cole, 1999). For mineralisation, optimum C/N ratios are <20, whilst for immobilisation, optimum C/N ratios are >40 (Doran & Smith, 1991; Paul & Juma, 1981; Stevenson & Cole, 1999). Further discussion about the influence of soil moisture on these processes will be given later in this chapter.

Nitrification

In this predominantly aerobic process, prokaryotes oxidise the ammonia to nitrate in two phases; 1) the oxidation of ammonia to nitrite, and then 2) the oxidation of nitrite to nitrate (Canfield et al., 2010; Krishna, 2002). The first phase is mainly performed by a single species of bacteria, *Nitrosomonas europaea*, however, bacteria from some other genera may also execute it, such as genera Nitrosospira and Nitrosococcus (Bernhard, 2010; Stevenson & Cole, 1999). Similarly, the second phase is mainly carried out by *Nitrobacter winogradskyi*, but genera Nitrospira, Nitrococcus, and Nitrospina may also be involved (Bernhard, 2010; Stevenson & Cole, 1999).

Factors affecting nitrification are temperature, moisture, pH, and concentrations of the substrates (Stevenson & Cole, 1999). When the nitrification process is not constrained, oxidation to nitrate-N can be as quick as ammonification, hence, nitrate-N is the main form of N available for plants in oxidized soils (Fageria & Baligar, 2005; Kaboneka et al., 1997; B. C. Liang & MacKenzie, 1994; Schmidt, 1982).

Biological denitrification

Biological denitrification can be defined as the anaerobic nitrate reduction carried out by microorganisms using organic carbon as the electron donor and producing gaseous N and/or nitrogen oxides (NO_x) (Bernhard, 2010; Canfield et al., 2010; Delwiche, 1981; Fowler et al., 2013; Neeteson & Carton, 2001). Denitrifiers comprise organisms of numerous genera of prokaryotes and some eukaryotes (Bernhard, 2010; Canfield et al., 2010). The main factors that affect performance of biological denitrification include pH, temperature, moisture, redox conditions, nitrate supply and organic carbon content (Krishna, 2002; Neeteson & Carton, 2001). It generally occurs in the first ten centimetres of soil (Neeteson & Carton, 2001). Recently, some mitigation strategies have been focused on promoting this process in order to decrease nitrate concentrations in drainage, but ensuring denitrification does not lead to higher emissions of NO_x is important, as nitrous oxide is considered to be a potent greenhouse gas (Di & Cameron, 2002).

For mineralisation, nitrification and denitrification of N, soil moisture (and consequently, redox conditions) is a key driver. Mineralisation and nitrification processes increase proportionally with water content in soil pores until reaching a maximum activity at 60% of water filled pore space. From this point, these processes will decrease with greater water, but denitrification activity will start, finding its maximum activity at 90% of water filled pore space (Power, 1990).

Processes within the N cycles are constantly interacting with each other, therefore, for example, if more N is being fixed, then probably N mineralization and nitrification will be enhanced, which could also favour leaching or denitrification depending on previously discussed factors (Jansson & Persson, 1982). Other processes such as assimilatory nitrate reduction, DNRA, chemo denitrification and anammox are also described by different authors but the characterisation of these processes is outside of the scope of this review.

Nitrogen leaching losses

Nitrogen movement in soil is mainly through mass flow and diffusion, the former being predominant for nitrate and the latter for ammonia (Krishna, 2002). Nitrate is the most abundant and mobile form of nitrogen in soil, therefore, factors influencing mass flow will also have an impact in nitrate transport (Table 2.2). Consequently, nitrate movement is highly dependent on hydrological processes (Salazar et al., 2015). In a comprehensive review, Lu et al. (2011) explained that the greatest changes in the ecosystem N cycle caused by N additions are soil inorganic N leaching and soil nitrate-N concentrations. The main N losses are denitrification and leaching (Wang et al., 2019). N will enter water systems as nitrate mainly through runoff and leaching (Wang et al., 2019).

Table 2.2. Factors affecting leaching losses of nitrogen during crop production (Krishna, 2002).

Factor	**Less leaching of N**	**Higher leaching of N**
Crop	Vigorous crop, established permanent crops, such as plantation fruit crops, grass lands or forestry	Poor crop stand, follow simple subsistence level cropping, sparse plant density
Soil	Fine-textured soils, poor drainage	Coarse-textured soils, good drainage, excessive seepage
Time of N application	Basal dosage or when applied at rapid growth stage	Off season application, autumn application
Quantity of N	Application at appropiate recommended levels allow better N-use efficiency	High rates, not timed properly
Climate (precipitation)	Low rainfall zones, moisture deficit conditions	Rapid surface flow, no moisture deficit, larger seepage

The role of fertilisation is still a matter of debate. In their study, Vagstad et al. (1997) highlighted that most of nitrate-N leached is from mineralised N_{org} from organic matter rather than from inorganic forms applied as fertilisers. On the other hand, Canfield et al. (2010) stressed the importance of rapid nitrification of ammonia from N based fertilisers. Finally, Jégo et al. (2008) pointed out that fertilisation might influence nitrate leaching only depending on application rate and time.

Consequences of nitrogen leaching

N overload in aquatic systems carries numerous health and environmental outcomes. In rivers and lakes may cause eutrophication, increasing algal growth exponentially and decreasing oxygen-dependant life, biodiversity and recreational value (Cameron et al., 2013). Another effect of nitrogen losses is the generation of nitrous oxide to the atmosphere, a greenhouse gas with a global warming potential 296 times the carbon dioxide potential (in a 100-year time horizon evaluation) (Folland et al., 2001; Kim et al., 2015). At the same time, nitrate leaching from agricultural fields might be worsened as intense precipitation events become more frequent with climate change, which could ultimately result in more nitrous oxide (Wang & Li, 2019).

High nitrate concentrations in drinking water have been closely related to methemoglobinaemia in babies and high risk of cancer and heart disease (Cameron et al., 2013; Grizzetti et al., 2011; Schullehner et al., 2018). Other health issues have been associated with high nitrate concentrations in drinking water, such as spontaneous abortion, intrauterine growth restriction, birth defects (Centers for Disease Control and Prevention, 1996; Manassaram et al., 2006; Nolan, 1996), Alzheimer's disease, diabetes mellitus and Parkinson's disease, among others (De La Monte et al., 2009; Tohgi et al., 1998).

2.3 Nitrogen in vegetables systems in New Zealand

Vegetable production in New Zealand constitutes a small proportion of the total area of national farms, representing about 0.3% (Horticulture New Zealand & Plant and Food Research, 2020; Stats NZ, 2021). Millner & Roskruge (2013) classified vegetable production into two categories, processed crops and fresh crops. The most important processed vegetable crops in New Zealand are potatoes, peas and sweetcorn, whilst major fresh vegetables crops are onions and buttercup squash. Comprehensive recommendations for nutrient management of vegetables systems in New Zealand are given by Reid & Morton (2019). They provide information on potential yields, essential nutrient demand, critical nutrient concentrations in plants and other factors for the major vegetables grown in New Zealand.

Intensive vegetable production systems normally require a large input of nutrients, and fertiliser N recovery by crops is normally low (approximately 11% in leafy greens to 50% in tuber crops), which results in significant N losses (Cameron et al., 2002; Crush et al., 1997; Williams & Tregurtha, 2003). Other related factors that may play an important role in increasing N leaching risks in these systems are the short period of plant growth, shallow rooted plants, frequent cultivation, and high N content of crop residues (Cameron et al., 2013). In addition, plant growth in winter is often slow and rainfall more intense, which increases the risk of N leaching losses.

Nitrogen leaching in market gardens in New Zealand has not been widely researched or well documented. According to Parfitt et al. (2006), nitrate-N leaching losses from vegetables may be around 60 kg N ha^{-1} yr^{-1}, while Di & Cameron (2002) suggest that normal values range from 70 to 180 kg N ha^{-1} yr^{-1}. Observations of nitrogen leaching in literature may vary from <5 up to 429 kg N ha^{-1} in a crop season, but the majority of the values are around 120 kg N ha^{-1} yr^{-1} in New Zealand (Table 2.3). In general, most studies show a positive relationship between the amount of fertiliser applied and N leaching values.

Crush et al. (1997) carried out a comparison of leachable N between different land uses for the Pukekohe area, concluding that the highest potential for leaching was from early potatoes > winter cabbage, winter lettuce and squash > dairying, kiwifruit, summer cabbage and summer lettuce > pumpkins, onions and main potatoes > dry stock farming.

Table 2.3. Nitrogen leaching losses studies in vegetable farming in New Zealand

Reference	System	Fertilisation	N Leaching averages^	
		-kg N ha^{-1}-	-kg NO_3-N ha^{-1}-	-kg N ha^{-1}-
Adams (1981), Leathers et al. (1983)	Pea	n/d	90 yr^{-1}	
Adams and Pattinson (1985)	White clover-pea-wheat-wheat-pasture	no fertiliser	5-90 yr^{-1}	
Crush et al. (1997)	Several vegetable crops	212-543	92-429 leachable	
Spiers et al. (1996); Di & Cameron (2002)	Vegetable crops	n/d	-	300 yr^{-1}
Martin et al. (2001)	Potato	no fertiliser	82	
		242-472	167-208	
Francis et al. (2003)	Winter potatoes	481		114
	Winter greens (spinach, cauliflower or cabbage)	166		110
Craighead & Martin (2003)	Potato	0-480		82-208
	Potato	0-480		24-134
Williams & Tregurtha (2003)	Winter broccoli	59-218		11-180
	Winter cabbage	150		178
	Winter spinach	400		246
	Winter potatoes	111-469		25-171
Williams et al. (2003)	Winter spinach	0		13-144
		200		80-236
		400		119-292
Sharp et al. (2011)	Potato-winter fallow-spring pea-winter fallow-potato	n/d		0-125
Herath et al. (2014)	Potato	120		67.7
Fenemor et al. (2015)	Cabbage-lettuce	400		6-45 yr^{-1}*
	Lettuce-lettuce	450		38-90 yr^{-1}*
	Pumpkin-lettuce	282		5-35 yr^{-1}*
Norris et al. (2017)	Fallow-lettuce-spinach-fallow-lettuce-cabbage-fallow	n/d		120-210 yr^{-1}
	Ryegrass-carrots-ryegrass-peas-ryegrass	n/d		<5 yr^{-1}
	Ryegrass-sweet corn-ryegrass-tomato-ryegrass	n/d		<5-37 yr^{-1}
	Ryegrass-potato-fallow-ryegrass-potato-fallow-onion	n/d		12.5-30 yr^{-1}

^Losses estimated for the crop growing period

*Estimated through SPASMO model

According to Cameron et al. (2002), market gardens in New Zealand contribute to high nitrate-N concentrations in groundwater, particularly shallow aquifers, citing concentrations >10 mg nitrate-N L^{-1}. Quin & Burden (1979) found concentrations of 8-12 nitrate-N L^{-1} in groundwater under irrigated land, consisting of pasture and crops in Canterbury. McLay et al. (2001) measured groundwater nitrate concentrations under different dominant land uses and concluded that, of the five land uses that they surveyed, market gardens had the greatest mean values (5-8 mg nitrate-N L^{-1}), although they found no strict correlation between land use and groundwater nitrate-N concentrations.

2.4 Mitigation efforts in vegetable farming

In New Zealand, Good Management Practices (GMP) is a suite of practices defined by the Land and Water Forum (2010) as "*industry-led programmes promoting practice changes to improve industry performance against water related objectives*". A document released by several national organisations including Crown Research Institutes (Crown entities charged with conducting scientific research) and primary sector organisations details a number of different GMPs focused on water quality improvement (MGM Governance Group, 2015). Therefore, GMPs might be a potential tool to use on farms to help increase water quality and make progress in meeting the One Plan requirements (Burbery et al., 2020). An important GMP described by the aforementioned document is developing nutrient balances in farms to minimise nutrient losses. The Code of Practice for Nutrient Management (Fertiliser Association, 2013) emphasises the GMPs on fertiliser use and details a number of factors to take into account, such as fertiliser type, application and timing.

In-field mitigation strategies

In this section, a selection of in-field mitigation strategies that deal with the way N is delivered to plants will be characterised. These mitigation practices have been selected because they have not been well studied in New Zealand.

Y-drop application

As part of the GMP suite, fertiliser timing and placement can enhance N use efficiency. According to the Fertiliser Association (2013), the fertiliser application method will affect the nutrient accessibility to plants and the soil-fertiliser interaction.

Y-drop is a relatively new technology, developed by the 360 Yield Center, which involves the precision placement of fertiliser (Bernhard & Below, n.d.). It consists of an attachment to a sprayer boom that dribbles liquid fertiliser right next to the crop row, thus increasing yields and reducing rates (360 Yield Center, 2015; Bernhard & Below, n.d.; Holland, 2014; Livesay, n.d.; Nafziger, 2019). A limited number of studies regarding this topic have been undertaken, and the available ones have been carried on only in the U.S. and specifically in maize. Firle (2018) compared three different application methods and concluded that, under the same conditions, coulter application, a type of side-dressing, produced higher maize yields with lower costs than a Y-drop application. Results reported by Purucker & Steinke (2020) indicated that the use of Y-drop did not produce different corn grain yields than the use of coulter injection, however, these authors suggested that the use of Y-drop may be more beneficial when soil moisture is not a limiting factor or when N needs to be applied in the late season. Similar findings were reported by Griesheim et al. (2022), who indicated that N uptake efficiency was comparable in corn and corn-soybean rotations between Y-drop and subsurface liquid application, indicating that Y-drop is less efficient when N is lost through volatilisation. Contrarily, Holland (2014) and The Ohio State Digital Ag Team (2018) obtained significantly higher yields with the same N fertiliser rates when using Y-drop application than coulter injection in maize. In their work, Bernhard and Below (n.d.) had also found considerably higher yield results in split Y-drop application compared to split broadcast sprayer, suggesting that this significant difference is due to better use of N by the plant.

Limited research in Y-drop application has been done in New Zealand. (Posthuma et al., 2020) have compared broccoli crop yields when applying dry or liquid fertiliser as bands alongside plants, indicating no significant differences between both methods.

Controlled release fertilisers

Controlled release fertilisers (CRF) or slow-release fertilisers (SRF) are a fertiliser technology to enhance N use efficiency (NUE), by coating encapsulated fertiliser with an organic or inorganic material, ensuring the release of small amounts of N over time (Bahar et al., 2019; Shaviv, 2000).

CRF have been widely studied in recent years and constitute a common GMP in the Code of Practice (Fertiliser Association, 2013). In a meta-analysis of 13 research articles performed by Quemada et al., (2013), CRF were found to reduce nitrate leaching between

20 and 45%. Though CRF might have a negative effect on crop yield by 0-10%, the nitrate leaching scaled-yield, that is, the crop yield divided by N leached, is still significantly higher in CRF treatments compared to rapidly available N fertilisers (Quemada et al., 2013).

While CRF have proved to be a cost-effective strategy, growers are still unwilling to use them mainly because of their higher cost (Shaviv, 2000). Furthermore, it is difficult for growers to predict the rate at which N is released over time with the highly intricate formulas available (Trolove et al., 2019a). Nutrient release in soil is directly related to environmental variables, such as soil temperature, humidity, or structure (Bishop, 2010). An extensive characterisation of the advantages of CRF, numerous types of CRF, the different release mechanisms (modes, length and pattern) as well as their model functions are described by Bahar et al. (2019), Bishop (2010), and Shaviv (2000). Shaviv, (2000) emphasised that controlled release fertilisers should match the nutrient demand and maintain nutrient availability over crop season in order to increase NUE.

In New Zealand, Bishop et al. (2008) measured higher yields of Italian ryegrass per kg N applied with polyurethane coated urea (PCU) in comparison with normal urea, and also registered lower N leaching losses in the PCU treatment. Research regarding CRF in New Zealand is limited and has been mainly focused on pastoral systems (Bishop, 2010; Bishop et al., 2008; Dawson et al., 2019; Edmeades, 2015; Pollock, 1989; Pollock et al., 1994; Trolove et al., 2019a). Thus, CRF use in New Zealand conditions has not been adequately researched, especially regarding its evaluation in other agricultural systems such as vegetable production.

Chicken manure

Although organic amendments are well known to improve various chemical, physical and biological soil properties (Flores et al., 2005), the effect of organic matter on nitrate leaching in soils is still a matter of controversy. In their extensive review, Wang et al. (2018) concluded that manure only or manure mixed with fertiliser reduced nitrate leaching in comparison with conventional fertilisation. Other researchers also stressed the importance of manure in reducing nitrate leaching when compared with conventional mineral N fertilisers. For example; Maynard (1993) in intensive vegetable production with intense rainfall events, Malone et al. (2007) in a long-term maize-soybean rotation, Kramer et al. (2006) in an apple orchard, and Sanchez-Martin et al. (2010) in a fallow-onion crop rotation. Kramer et al. (2006) emphasised the remarkable influence of organic matter in promoting denitrification processes and obtained 4.4 to 5.6 times more N leached from conventional farms than from organic farms, with the same amount of nutrients applied. However, in a comprehensive review, Kirchmann & Bergström (2001) suggested that organic farming usually present slightly less nitrate leaching mainly due to lesser inputs, and if N levels are the same, similar N amounts are also leached.

In fact, various researchers emphasise that manure can lead to significant nitrate leaching loads, in particular if variables such as crop N uptake, amounts, timing (season) or mineralisation are not well considered when applying (Daza-Torres et al., 2018; Farneselli et al., 2018; Flores et al., 2005; Malone et al., 2007; Thangarajan et al., 2015; Yan et al., 2002). For instance, Kirchmann & Bergström (2001) argue that organic amendments can release N through mineralisation when no uptake is taking place, favouring N leaching losses. Furthermore, Thangarajan et al. (2015) highlight that addition of organic N can increase ammonia oxidiser bacteria population, promoting nitrification, and therefore, soil temperature and microbial activity play a major role in N release from organic amendments that must be acknowledged above all other variables when applying.

Information about the effect of organic matter on water holding capacity in soils is also often contradictory, regardless of soil intrinsic properties. While Bouyoucos (1939) determined there is a significant effect of organic matter on available water in soils, a more recent study comprising 60 different studies suggest this effect is negligible (Minasny & McBratney, 2018).

The basis for using organic amendments, like chicken manure, as an alternative to conventional fertiliser is that they slowly and steadily release N in time (Daza-Torres et al., 2018; Hadas et al., 2004; Yan et al., 2002; J. Yang et al., 2018). According to some researchers, as organic amendments improve soil structure promoting infiltration and percolation, higher volumes of drainage with low nitrate concentrations might be expected (Daza-Torres et al., 2018; Sanchez-Martin et al., 2010).

In some cases, particularly in intensive vegetable production, N requirements from crops are not met by organic amendments; therefore, researchers often use complementary mineral N fertilisers. Yang et al. (2018) demonstrated that adding controlled-release fertiliser and chicken manure to conventional fertiliser is highly effective in reducing nitrate leaching losses in comparison with only conventional fertiliser, whilst reducing costs and improving yields.

2.5 Simulation models

Impacts of non-point source pollution have led to development of different tools to increase N use efficiency and assess mitigation measures. Simulation models based on soil-crop-atmosphere relationships make it possible to understand local hydrology, estimate nutrient leaching, assess management practices and provide a guide to select N management strategies in a long term, realistic and cost-effective way to reduce nutrient losses (Cameira & Mota, 2017; Neeteson & Carton, 2001; Whitmore, 1996). For these reasons, simulation models are a key component to develop public policies and address environmental and economic issues (Bezlepkina et al., 2010).

APSIM

Of the numerous farming simulation models, one of the most suitable for the Arawhata catchment conditions is the APSIM model. The Agricultural Production Systems sIMulator (APSIM) (Holzworth et al., 2014; Keating et al., 2003) is a farming simulation model framework with several modules that integrate environmental factors, such as climate, soil, and water, with management factors, such as crop plant species, rotations, fertiliser application, etc. (Gaydon et al., 2017; Sharp et al., 2011a; Sharp et al., 2011b). One of the special characteristics of APSIM compared to other models is that the soil is one of the main components in the model, hence, the simulation focus is pointed towards resources stock instead of demand (Gaydon et al., 2017; Holzworth et al., 2014; McCown et al., 1996; Probert et al., 1998b). This allows observation of long-term changes in soil and assessing sustainability, as well as economic and ecologic consequences from agricultural practices (Gaydon et al., 2017; Keating et al., 2003). A highly flexible manager module is also one of the key advantages of this model, making agricultural practices customisable (Holzworth et al., 2014). The history, origins, modules, coding languages and studies of APSIM have been described extensively by (Holzworth et al., 2014; Keating et al., 2003; McCown et al., 1996). Documentation for APSIM's parameterisation are also given by the official website (https://www.apsim.info/documentation/model-documentation/soil-modules-documentation/).

APSIM has been parameterised, calibrated and validated in a number of different countries. The bulk of these studies have been done in Australia, where APSIM originated (Asseng et al., 1998; Brilli et al., 2017; Holzworth et al., 2014; Huth et al., 2009; Keating et al., 2003; O'Leary et al., 2016; Probert et al., 1998a; Soufizadeh et al., 2018; Zhao et al., 2014). Gaydon et al. (2017) pinpointed several works done all over the Asian continent, while some other studies have been also conducted in Africa (Borus et al., 2016; Cheeroo-Nayamuth et al., 2000; Inman-Bamber et al., 1999; O'Leary, 2000; Van der Laan & Franke, 2019; Whitbread et al., 2010) and Europe (Asseng et al., 2000; Wessolek & Asseng, 2006). A complete list of hundreds of studies around the globe can be accessed on APSIM's official website (https://www.apsim.info/apsim-model/publication-metrics/).

APSIM was recently re-written into a new version, APSIM Next Generation, with the intention to provide a more efficient and robust model, and the ability to be run in several platforms, such as clouds or mobile devices (Holzworth et al., 2018). Little research has been done in this recently readjusted version of APSIM.

Most research carried on in New Zealand in APSIM has been focused on pastures, with a ryegrass-white clover rotation foremost, while only about 7% of the studies have been focused on vegetable farming (Figure 2.1). Waikato and Canterbury have been the main areas where these studies have been performed, with only a few made in Manawatu-Whanganui Region. However, many of these studies include a nitrate leaching simulation and some even include leaching mitigation strategies related to pasture grazing. In the Canterbury area, Sharp et al. (2011a) and Sharp et al. (2011b) compared APSIM simulated N values with field experiment data in a vegetable farming system, finding that overall model performance was successful, but adjustments must be made to obtain more accurate predictions of nitrate leaching and nitrogen mineralisation. In a more recent study, Khaembah et al. (2015) compared simulated and observed nitrate-N leaching losses obtained from ceramic suction cups under vegetable crops in Canterbury, finding that APSIM explained 67% of the variation in measured data and that there was generally good agreement between both datasets.

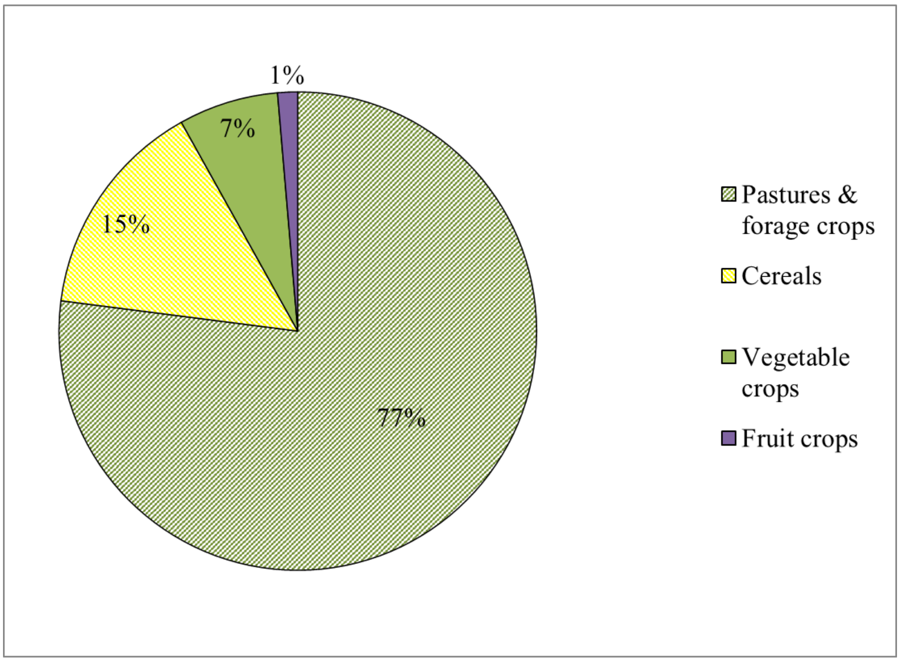

Figure 2.1. APSIM studies in New Zealand (details in Table A1 in Appendix)

Even though it has been extensively studied, and most studies indicate a successful simulation, APSIM is constantly evolving and improving. Hendy et al. (2018) noted APSIM's ability to cover different crop systems and processes at different scales, but also warned about execution time, out of date documentation and the need for more model validation. Bloomer et al. (2019) particularly emphasize the high potential of APSIM to simulate horticultural systems due to its daily time step. However, Brilli et al. (2017) related some weaknesses of APSIM relating to N cycle and soil properties (such as underestimation of soil N supply or overestimation of leaching).

SWAT and SWAT+

The SWAT model (Soil and Water Assessment Tool) is a river basin scale model that works on a daily-step time basis and can predict several outcomes from land use on water, nutrient and sediments in large basins with heterogeneous conditions (Arnold et al., 2012; Gassman et al., 2007; Morcom, 2013). Watersheds can be subdivided in sub-watersheds and then into hydrologic response units (HRUs), which are defined as units with homogeneous attributes (soil, land use, management, and topography) (Arnold et al., 2012). Similar to APSIM, a complete peer-reviewed literature regarding the SWAT model can be found in the official website (https://www.card.iastate.edu/swat_articles/). Parameterisation and calibration of the model have been described by Arnold et al. (2012) and other documents that can be found in the official website (https://swatplus.gitbook.io/docs/user/io). Some authors underline that SWAT requires a large amount of input data (Ekanayake & Davie, 2005; Morcom, 2013; Neitsch et al., 2002). The Soil and Water Assessment Tool Plus (SWAT+) is a completely restructured version of the SWAT model, which integrates with QSWAT+, a more user-friendly interface developed for QGIS software (Bieger et al., 2017; Kakarndee & Kositsakulchai, 2020). The SWAT+ model has been developed to provide more flexibility to the model and enhance the spatial representation of transportation processes in a catchment (Bieger et al., 2017). However, research has not been done extensively yet for this newly restructured version.

Most international studies have evaluated the ability of SWAT and SWAT+ to estimate flow rate and water balance, with normally satisfactory results, but fewer have evaluated nitrate-N loads and concentrations, with less consistent outcomes. Van der Laan & Franke (2019) evaluated SWAT in the Middle Olifants catchment in South Africa, concluding that the model performed well for flow rate based on the R^2 indicator, but poorly when considering other statistical measurements, and for nutrient loss prediction. Similar findings were reported by Samarinas et al. (2020) for flow rate and nitrate-N loads in Greece; however, they also suggested that SWAT's estimations improved significantly when land use data was updated annually. Akhavan et al. (2010) aimed to represent the flow rate and nitrate-N concentrations in an intensively cultivated catchment in Iran (with predominantly wheat and potato crop cultivation). SWAT's ability to predict discharge at the outlet of the catchment was deemed as good to very good (NSE=0.7-0.77; R^2=0.7-0.83), and a qualitative validation of the nitrate-N concentrations estimations with groundwater data deemed as satisfactory (Akhavan et al., 2010). Schilling & Wolter (2009) assessed the use of SWAT to represent flow rate and nitrate-N loads of the Des Moines River in the U.S. The SWAT model achieved very good performance, with an R^2 and NSE for flow rate of 0.8 and 0.79, respectively, and an R^2 and NSE for monthly nitrate-N loads of 0.74 and 0.77, respectively (Schilling & Wolter, 2009).

Recent studies show that, once calibrated and validated, SWAT has been able to satisfactorily simulate watersheds with several crops, and different in-field and edge-of-field BMP scenarios, altogether and in different combinations, helping in deciding the most effective one in reducing N loads (Himanshu et al., 2019; López-Ballesteros et al., 2019; Malagó et al., 2019; Merriman et al., 2018; Motsinger et al., 2016). Schilling & Wolter (2009) found the SWAT model a helpful tool to identify where and how to begin implementing nitrate-N reduction practices most efficiently. Gitau et al. (2008) demonstrated quantitatively the SWAT model's suitability to assess BMPs impacts, with observed data pre- and post- BMP installation of phosphorous at watershed-scale level in the U.S., however, a BMP tool was necessary to provide detailed evaluations of individual BMP impacts. On the other hand, Gassman et al. (2007) underlined the model's high flexibility, which makes it suitable for BMP assessment, but it might show some limitations when it comes to BMP placements due to insufficient HRU spatial definition, especially in filter strips, riparian buffer zones, etc.

For simulating BMP placements, most studies complement the use of SWAT with an optimization tool, such as a genetic algorithm (GA) or a computational framework to find the BMP placement with best cost-effective ratio (Chiang et al., 2014; Geng et al., 2019; Gitau et al., 2004, 2006; Liu et al., 2019; Noor et al., 2017; Panagopoulos et al., 2011; Pyo et al., 2017; Qin et al., 2018). Genetic algorithms are described by Gitau et al. (2004) as a multi-tool that "*combines initial pollutant loadings from SWAT with literature-based pollution reduction efficiencies from the BMP tool and with BMP costs to determine cost-effective watershed scenarios*".

According to Fu et al. (2019), SWAT is one of most studied hydrology models in the world, but only a handful of these studies have been undertaken in New Zealand (Ekanayake & Davie, 2005; Me et al., 2015, 2018; Morcom, 2013). The SWAT model has not been used extensively in New Zealand conditions probably because other hydrologic models have been created for local conditions, such as the CLUES model, or existing ones –that require a smaller number of inputs– have been adapted, such as GLEAMS or SPARROW models (Morcom, 2013; Parshotam & Robertson, 2018). In their study, Ekanayake & Davie (2005) did not have satisfactory predictions of nitrate concentrations for the Motueka Catchment, arguing that SWAT was unable to account for N in groundwater, underestimating background concentrations. Also in the Motueka

Catchment, Cao et al. (2009) found a satisfactory performance of SWAT to predict flow rate of the main river, but variable (poor to good) performance was found in tributaries. Me et al. (2018), on the other hand, reported adequate results when simulating nutrient loads and comparing with observed data. In an additional study, Me et al. (2015) evaluated the SWAT model's performance in the Puarenga Stream in Rotorua, concluding that flow rates were estimated accurately, but total N concentrations could not be estimated correctly because of non-steady state groundwater concentrations due to anthropogenic N applications to the land. They also highlighted the importance of high-frequency event-based monitoring to not underestimate quick flow fluxes that can lead to increased model error (Me et al., 2015). Hoang (2019) indicated that, particularly for monthly time-steps, SWAT had very good performance to reproduce flow rate in the Toenepi Catchment, and reasonably good for nitrogen loads and concentrations.

Chapter 3: Quantification of N losses under intensive vegetable production systems

3.1 Abstract

Nitrate-N concentrations and losses in soil and drainage water were monitored between 2020 and 2022 in two vegetable crop sites near Levin, Manawatu-Whanganui. One site consisted of a potato-fallow-potato-fallow-onion rotation with treatments: control (C), standard practice (STD), split liquid (LF), controlled release (CR), good practice (GP) and excess (EXC) fertiliser applications. The second site consisted of a beetroot-fallow-Pak choi-ryegrass rotation with treatments: control (C), standard practice (STD), reduced fertiliser programme (RF), chicken manure (CM) and excess (EXC) fertiliser applications. High levels of mean nitrate-N concentrations in topsoil (0-20 and 0-30 cm) (from 9.3 to 18.3 ppm), subsoil (30-60 cm) (from 7.3 to 9.6 ppm), and drainage water (from 16.9 to 61.9 mg L^{-1}), and nitrate-N leaching losses (95 to 225 kg N ha^{-1}) were observed under the usual practice in these systems during the drainage season, particularly at fallow periods. In-field management practices that aim to change the way N is released did not produce equal or larger crop yields than current practices but could potentially help in reducing nitrate-N concentrations (mean reduction of 45% in the topsoil) and losses if N demand by the crop is addressed correctly. When excessive amounts of N fertiliser were applied, crop yields increased marginally and did not have a significant difference from the normal practice of growers. However, excessive N fertiliser applications had significant impacts on mean nitrate-N concentrations in soil (up to 129% greater in the topsoil and 53% greater in the subsoil) and soil water (up to 63% larger) compared to the normal practice of growers. Quantitative observations are key to the calibration and validation of field-scale models such as APSIM Next Generation to predict nitrate-N losses under different cropping and fertiliser scenarios in the study area.

3.2 Introduction

Vegetable crop production is important to the economy of the Horowhenua district in the Manawatu-Whanganui region, particularly in the Arawhata catchment. Horowhenua has been identified as one of the key areas of vegetable crop production in New Zealand, particularly for broccoli, cauliflower, cabbage, and carrots (Horticulture New Zealand, 2017). In addition, businesses associated with outdoor vegetable production are one of the main employers in the region (Infometrics, 2021). However, vegetable cropping is considered to be one of the land-uses responsible for most of the nitrate-N leaching losses to the environment (Clothier et al., 2007). In an attempt to characterise nitrate-N leaching losses of 40 different green vegetables species, Zemek et al. (2020) indicated that broccoli, cauliflower and cabbage, –the most commonly grown crops in Horowhenua– have a very high leaching potential (>250 kg nitrate-N ha^{-1}).

The vulnerability of vegetable systems to nitrate-N leaching has been attributed to the use of large amounts of N fertiliser, low N use efficiency, and large amounts of residual N (Di & Cameron, 2002). Moreover, residues of vegetable crops often have low C:N ratios which result in rapid N mineralisation and accumulation in the soil (Chaves et al., 2007; De Neve & Hofman, 1998).

Essentially, the risk of nitrate-N leaching will be proportional to the accumulation of nitrate-N in the soil that coincides with drainage events (Di & Cameron, 2002). In this regard, studies have found that some practices might help to reduce nitrate-N leaching from these systems. Three important practices are; optimising the timing of N fertiliser applications to meet plant N demand, the use of catch crops, and managing crop residues adequately (Zemek et al., 2020). Other in-field practices have also shown promising results in decreasing N losses, such as the use of controlled release fertilisers (Quemada et al., 2013), fertilisers that release small and steady amounts of N in time, and the use of liquid fertiliser (Holland, 2014; The Ohio State Digital Ag Team, 2018).

In New Zealand, there has been very little research into nutrient leaching under vegetable crops. In part, this is due to the difficulties of measuring leaching losses in free draining soils. Essentially, there are three techniques that have been used, namely: suction cups, lysimeters, and fluxmeters. Each of these techniques has its advantages and disadvantages (Weihermüller et al., 2007), but as they sample relatively small drainage volumes, all of them are limited in their ability to quantify variability. All these techniques involve a degree of soil disturbance during installation. Suction cups do not measure drainage volume independently and are unsuitable for use in coarse-textured soils or soils with preferential flow (Q. Wang et al., 2012). Repacked lysimeters are disruptive to the soil profile and may create conditions conducive to denitrification if suction is not applied at the bottom of the lysimeter (Weihermüller et al., 2007). While it is important to be aware of these shortcomings, these are the only techniques available to measure leaching losses in permeable soils.

The limited research conducted in New Zealand using suction cups and fluxmeters has suggested that there are large amounts of nitrate-N leached under vegetable production systems, ranging from 63 to 292 kg N ha^{-1} yr^{-1} (Francis et al., 2003; Herath et al., 2014; Norris et al., 2017; Williams et al., 2003). Some findings suggested that the amount mineral of N in the soil is closely associated with the amount of N in drainage water (Norris et al., 2017).

Existing research appears limited in terms of a robust quantification of nutrient losses and its critical flow pathways in vegetable crop production of the Arawhata catchment surrounding Lake Horowhenua. Increased understanding of leaching rates under vegetable systems is essential in order to assist central Government and Regional Councils to make realistic and effective policy decisions, and to identify and evaluate the impact of mitigation practices on N leaching. Hence, the aims of this study were:

- To quantify the amount of N leached to groundwater and surface drains in representative Arawhata sub-catchment paddocks and crops.

- To assess the performance of selected in-field strategies to reduce N leaching from intensive vegetable farming in the Arawhata sub-catchment.

3.3 Materials and Methods

3.3.1 Study area

Two different research sites were established in the Arawhata catchment, Horowhenua District in the Manawatu-Whanganui region (Figure 3.1). The trial was conducted between December 2019 and February 2022. The selected sites were considered representative of the vegetable growing soils and climate in this region.

Figure 3.1. Arawhata catchment experimental sites ("Greens" for green vegetables and "Potatoes" for potatoes).

Green vegetables were grown at the experimental site located in the lower section of the catchment while potatoes were grown at the other site in the middle of the catchment.

The climate in Levin is temperate (Garr & Fitzharris, 1991). The 30-year average annual rainfall is 1163 mm (Chappell, 2015). Some growers may supplement rainfall with irrigation to obtain higher yields. Levin has mild summers (21-22°C mean daily maximum temperature) and winters (5-6°C mean daily minimum temperature), resulting in annual average temperatures of 13-14 °C (Chappell, 2015).

3.3.1.1 Soil types

According to the 'New Zealand Soil Classification', the most common soils in the area are Allophanic Brown Soils, Brown Soils and Perch-gley Pallic Soils (Landcare Research, 2018).

The soil at both experimental sites was inspected in a recent survey (A. Palmer, personal communication, May 2022). At the Green vegetables site, the soil was classified as a Shannon silt loam. The soil at the Potatoes site was classified as a Waitohu silty clay loam, which has better drainage than the Shannon series (Table 3.1).

Table 3.1. Description of soils at the experimental sites (Palmer & Wilde, 2007).

Variable/Site	Greens site	Potatoes site
Soil taxonomy	Aquic Dystrochrept	Typic Dystrochrept
Family	fine-silty mixed mesic	fine-silty mixed mesic
Series	Shannon	Waitohu
Texture group	Silt loam	Silt loam
Profile texture	Silt loam	Silt loam/Silty clay loam
Depth to gravels	~1.5 m	~0.9 m
Landform genesis	River terrace	River terrace
Soil parent material	Quartzo-feldespathic loess over gravel/sand	Quartzo-feldespathic loess over gravel
Derived rock	Greywacke, argillite	Greywacke, argillite
Physiographic position	Flat (0-3°)	Flat (0-3°)
Microrelief	-	Flat
Perched water	Yes	Yes – deeper
Drainage class	Imperfectly drained	Moderately well drained
Permeability class	Slow	Moderate-slow

Some impervious layers (pans), such as plough pans in the Waitohu soil and densipans in the Shannon soil have been reported. Both soils become stiff and harder to penetrate between 40-60 cm depth.

3.3.1.2 Cultivation practices

Management practices and crop rotations for both sites for the study period are summarised in Table 3.2. Fertiliser application practices are described in the treatment section of each site.

Table 3.2. Cultivation practices at each site for the study period.

Practice/Site	Green vegetables site	Potatoes site
Crop rotation[a]	Beetroot - ryegrass – Pak choi - ryegrass - cabbage/broccoli - maize - lettuce	Potato (cv. Nadine) (2 years)-Onion (2 years)-Ryegrass (4-5 months)
Crops during study[b]	Beetroot – fallow – Pak choi – ryegrass	Potato (cv. Nadine) (2 years)-Onion
Irrigation	No	Boom sprayer, as needed
Groundwork	Rotary hoe and bed building at sowing	Mouldboard ploughing and bed building at sowing. Mounding up at ~5 weeks from sowing
Herbicide	As needed	As needed
Insecticide	As needed	As needed

[a] Normal crop rotation for this field

[b] Crop rotation during the study period

Tillage, seed density, depth of sowing and other management practices on the plots followed the grower's normal practices.

3.3.2 Potatoes site

3.3.2.1 Treatments and experimental design

The experiment started with mouldboard ploughing on 15 December 2019, and potato (*Solanum tuberosum* L. cv. 'Nadine') sowing on 24 December 2019, and was finished on 18 February 2022. The study included two potato seasons and one onion season with different treatments for each season. A potato planter was used to sow the potatoes and beds were formed with a power harrow machine. After 5 weeks, potatoes were mounded up using the same machines. These machines were also used for the onion cultivation.

The treatments for the first potato season (from December 2019 to October 2020) were: control (C); standard practice (STD); split liquid (LF) and controlled release (CR) fertiliser applications (Table 3.3). The treatments for the second potato season (from November 2020 to August 2021) were: control (C), standard practice (STD); good practice (GP) and excess (EXC) fertiliser applications (Table 3.4). With only two treatments, the onion study was smaller in scale. As the onion crop was sown into the plots established for the second potato season, the treatments for the onion season (from September 2021 to February 2022) were: control-standard practice (C-STD) and standard practice-standard practice (STD-STD) fertiliser applications (Table 3.5).

Granular fertilisers were applied as a banded side-dressing at sowing, while later applications were broadcast. Split liquid fertiliser treatment was applied with Y-drop technology in each emitter of the sprayer boom. All treatments had sufficient P and K for optimum crop growth.

The 'standard' treatment was chosen to quantify N losses under potatoes at the site i.e., using the farmers routine fertiliser regime (the farmer's standard practice). The 'control' treatment was an unfertilised treatment but these plots, obviously, had a history of cropping. Treatments 'split liquid fertiliser' and 'controlled release fertiliser' were chosen mainly due to their potential to reduce nitrogen leaching without having major impacts on crop yield. For the second season, the treatment 'good practice' was selected to quantify N losses associated with the recommended N fertiliser practice for potato crop (Reid and Morton, 2019). It is not clear why the farmer was applying less fertiliser than 'good' practice for the second potato growing season. The 'excess' treatment aimed to test APSIM's ability to quantify N losses under fertiliser rates that are greater than the recommended fertiliser programme (Chapter 4). All of the above treatments would also help in parameterising and evaluating APSIM's ability to predict; crop yields, nitrate-N concentrations in soil and water, and soil moisture under field conditions (Chapter 4).

Table 3.3. First potato season (from December 2019 to October 2020) fertiliser programme treatments at the Potatoes site.

Treatment	N in fertiliser (kg ha^{-1})	Type of fertiliser	Timing
Control (C)	0	n.d.	Sowing
Standard practice (STD)	180	Yara Mila Complex ®	Sowing
Split liquid fertiliser (LF)	0	n.d.	Sowing
	60	Urea	4 weeks after sowing
	120	Urea	6 weeks after sowing
Controlled release fertiliser (CR)	193	SmartFert ®	Sowing

Table 3.4. Second potato season (from November 2020 to August 2021) fertiliser programme treatments at the Potatoes site.

Treatment	N in fertiliser (kg ha^{-1})	Type of fertiliser	Timing
Control (C)	0	n.d.	Sowing
Standard practice (STD)	180	Yara Mila Complex ®	Sowing
Good practice (GP)	180	Yara Mila Complex ®	Sowing
	60	Urea	6 weeks after sowing
Excess (EXC)	180	Yara Mila Complex ®	Sowing
	120	Urea	6 weeks after sowing

Table 3.5. Onion season (from September 2021 to February 2022) fertiliser programme treatments at the Potatoes site.

Treatment	N in fertiliser* (kg ha^{-1})	Timing
Control - Standard practice	36	63 days after sowing
(C-STD)	30	81 days after sowing
	30	105 days after sowing
	30	129 days after sowing
Standard practice - Standard practice	36	63 days after sowing
(STD-STD)	30	81 days after sowing
	30	105 days after sowing
	30	129 days after sowing

*All fertiliser applied corresponded to Yara Complex®

The paddock had previously grown ryegrass for five months, thus beginning the potato-onion-ryegrass rotation described in Table 3.2.

The experimental design in the first season consisted of four replicates of three fertiliser treatments and the control in four blocks which were arranged in a randomised complete block design (Figure 3.2). Each of the in 15 plots was 3.5 m wide x 16 m long. The control treatment (C1) in Block 1 in the first potato growing season was lost due an error in application. Liquid fertiliser was applied to the appropriate plot in Block 1 and, inadvertently, to the control plot in this block as well. The liquid fertiliser was applied in four rows (3.5) along the whole paddock, excluding the area where controlled release fertiliser (CR) and control (C) treatments were to be placed. The rest of the field (i.e., outside of these four rows) had the standard practice of the farmer (Figure 3.2) and one plot per block was chosen for monitoring.

For the second season, the experimental design was four replicates of three fertiliser treatments and the control in four blocks which were arranged in a randomised complete block design (Figure 3.3), resulting in 16 plots, each of which was 3.5 m wide x 16 m long. The control treatment was not in line with other treatments to avoid contamination when applying the second dose of fertiliser in treatments GP and EXC.

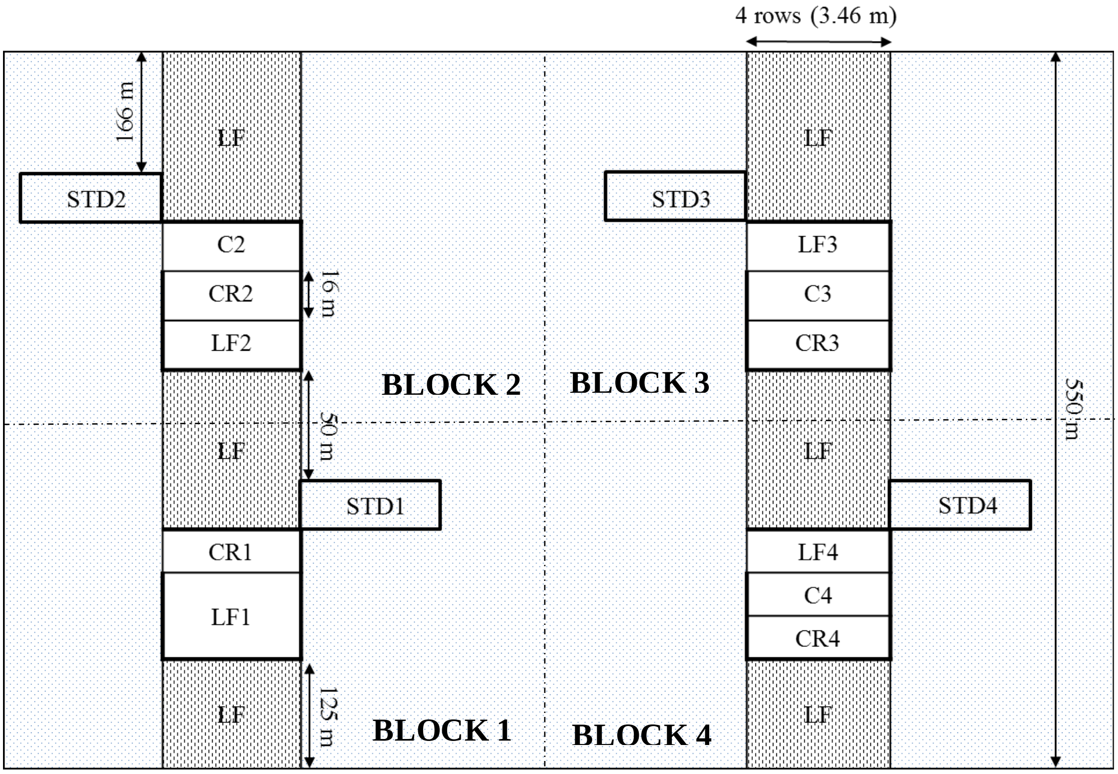

Figure 3.2. Potato field treatments design for the first season (C for control treatment; STD for standard practice treatment; LF for split liquid fertiliser treatment and CR for controlled release fertiliser).

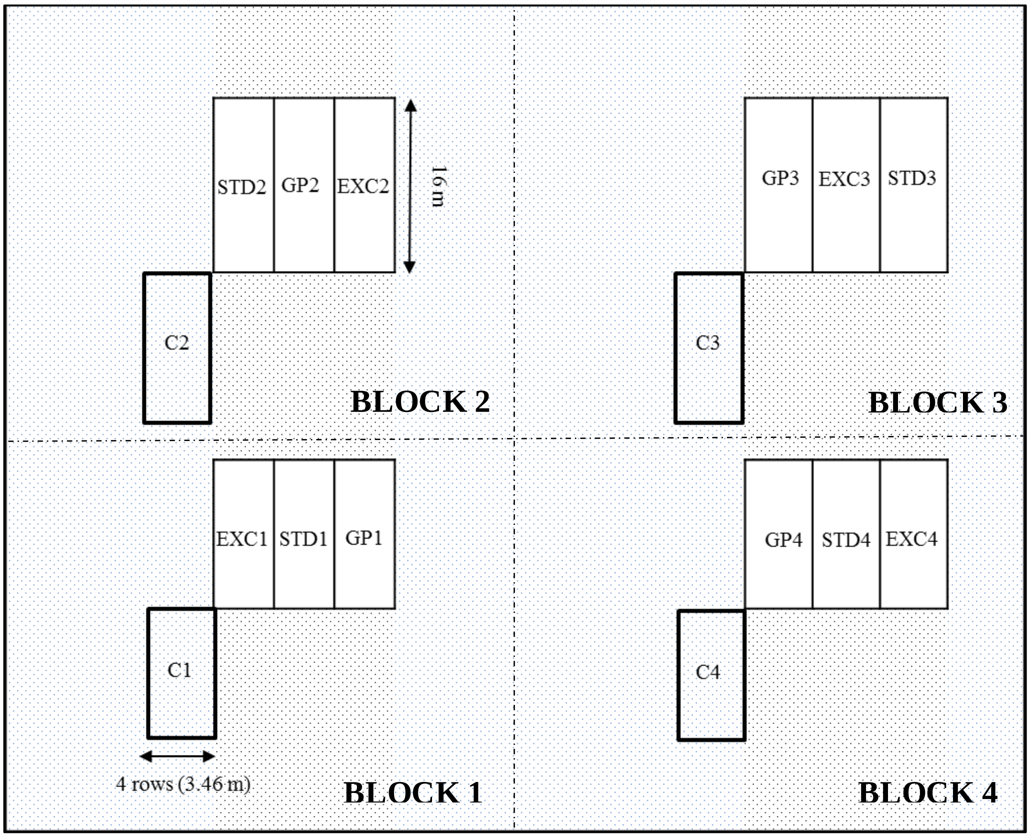

Figure 3.3. Potato field treatments design for the second season (C for control treatment; STD for standard practice treatment; GP for good practice treatment and EXC for excess treatment).

3.3.2.2 Methodology

Samples were collected for measurement of gravimetric moisture content of the topsoil (0-30 cm) and subsoil (30-60 cm). For the first two months of the first crop season, soil samples were collected weekly, after which samples were taken once a month. In April and May 2020, samples could not be collected due to a lockdown period of the COVID-19 pandemic. Two samples of soil from the two depths (0-30 and 30-60 cm) were collected in the centre of the potato ridge from each plot. In the second crop season, samples were collected from three locations in the bed (Figure 3.4) for both depths to obtain better represent conditions in the bed and furrow soil. The surface and subsoil samples were also sent to a laboratory for analysis of ammonia-N and nitrate-N (details in Table B1 in Appendix B).

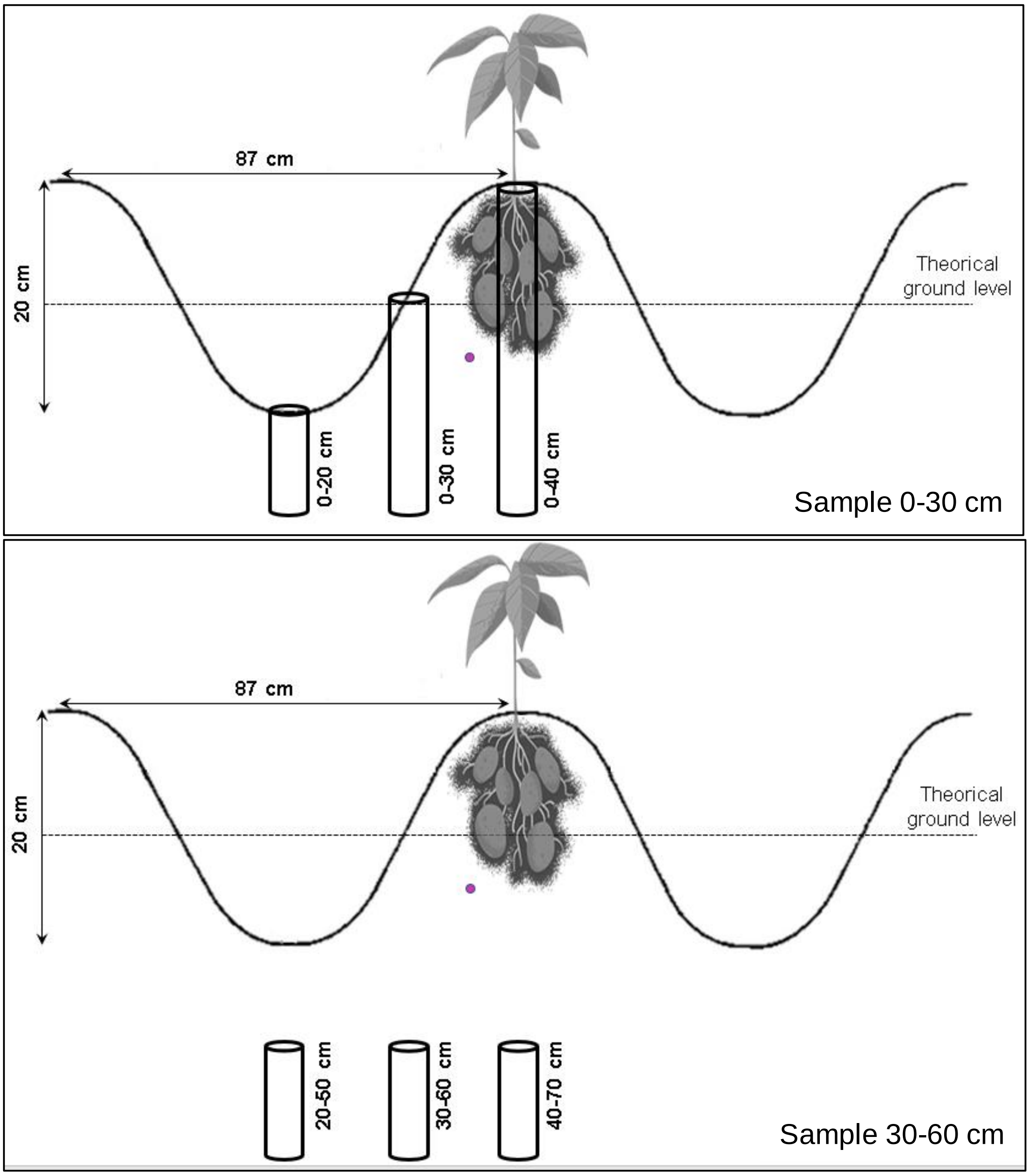

Figure 3.4. The soil sampling scheme employed in the second potato growing season. The three sampling positions are shown.

Soil bulk density and the soil water retention curve were determined using samples collected on 14 July 2020 in the standard practice treatment (STD) plots. Soil samples were collected in intervals of 10 cm depth for the first 40 cm, and every 20 cm for the 40-100 cm depth. Soil bulk density was measured following the procedures described by Klute (1986), while the soil water retention curve was measured using pressure plate apparatus following the Loveday (1974) method.

Suction cups were installed at 60 cm depth in the mid-point between ridge and furrow to help quantify nitrate-N leaching (Figure 3.5). To install the suction cups, a 60 cm hole was augered, 10 g of sand silica and 10 mL of water were added to the bottom of this hole. The suction cups were inserted and the gap around the cup was backfilled with soil before a cap of bentonite was placed on top (to avoid bypass flow). Three suction cups were installed in each plot resulting in 48 suction cups, and soil solution samples were collected the day after intense rainfall events (approximately >15 mm rain).

Figure 3.5. Porous suction cup in the Potatoes site.

N concentrations from suction cups samples were analysed using a QuikChem 8500 flow injection analyser, following the methods described in Table 3.6.

The nitrate-N leaching flux was calculated using the modelled drainage from APSIM (Chapter 4).

Table 3.6. Nitrogen forms analysis in water samples from suction cups in the Potatoes site.

Sample	N form	Method
Filtered	Nitrate-N	Reduction via copperised cadmium, NEDD coupling and colorimetry at 520 nm (Wendt, 2000).
	Ammonium-N	Reaction with sodium hydroxide, potassium tartrate, buffer solution, working buffer solution, sodium nitroprusside, sodium hypochlorite and a saline diluent. Use of colorimetry at 660 nm (Wendt, 2000).

As soil morphology (low chroma mottles) suggests that the water table sometimes rises to within 60 cm of the soil surface, one-metre-long piezometers (entire length screened) were installed in the field to monitor the presence of a perched water table. However, no water was found during the monitoring period.

Treatment plots were harvested when the grower considered appropriate to harvest his field. At harvesting time, lineal metre crop yields were estimated from 2 m of the two central rows of each treatment plot.

Normality of the residual errors and homoscedasticity of variances assumptions were checked on every dataset in order to fit a linear mixed model. If the normality assumption was not met, results were fitted to a Generalised Additive Model for Location, Scale and Shape (GAMLSS), a more extensive form of generalised linear mixed model, to identify whether there was any statistical difference between two or more treatments. GAMLSS has a broader option for data distributions to fit in the model and the most suitable distribution was selected depending on the dataset. The different replicates were deemed as a random effect variable in the model, and time and treatment were deemed as fixed effects. Subsequently, estimated marginal means from the model were then compared to search for statistical differences between treatments, following the Tukey method for adjusting the p-value. GAMLSS outputs were reported in the results section whenever an interaction between treatment and time was statistically significant, but if no significant interactions were found, only the comparison of the means is shown.

3.3.3 Green vegetables site

3.3.3.1 Treatments and experimental design

A bank of 20 lysimeters was constructed inside the grower's field in December 2019. A crop of beetroot was grown in the lysimeters in the first season (2019), and this was followed by Pak choi in the second season (2020). The following N fertiliser treatments were applied in this period: control (C); standard practice (STD); chicken manure (CM); reduced fertiliser programme (RF); and excess (EXC) (Tables 3.7 and 3.8).

Table 3.7. Fertiliser treatments for the first growing season (beetroot) at the Green vegetables site.

Treatment	**N in fertiliser** (kg ha^{-1})	**Type of fertiliser**	**Timing**
Control (C)	0	n.d.	Sowing
Standard practice (STD)	36	Nitrophoska®	Sowing
	81	Calcium ammonium nitrate	4 weeks after sowing
Chicken manure (CM)	117	Manure	Sowing
Reduced fertiliser programme (RF)	72	Nitrophoska®	Sowing

Table 3.8. Fertiliser treatments for the second growing season (Pak choi) at the Green vegetables site.

Treatment	**N in fertiliser** (kg ha^{-1})	**Type of fertiliser**	**Timing**
Control (C)	0	n.d.	Sowing
Standard practice (STD)	50	Calcium ammonium nitrate	Sowing
Chicken manure (CM)	100	Manure	Sowing
Excess (EXC)	100	Calcium ammonium nitrate	Sowing

The ‘standard’ treatment was chosen to quantify N losses under green vegetables at the site (the farmer’s standard practice). The ‘control’ treatment was an unfertilised treatment, but the soil had a history of cropping. The ‘chicken manure’ and ‘reduced fertiliser programme’ were chosen for their potential to reduce nitrogen leaching without having major impacts on crop yield. In the second season, the ‘excess’ treatment was added to test APSIM’s ability to quantify N losses higher than the recommended programme for soluble N (Chapter 4).

The lysimeters area, which was approx. 10 m long and 2 m wide, contained 20 lysimeters placed along the walls of a trench which allowed access for sampling purposes (Figure 3.6).

Figure 3.6. Installation of lysimeters at the Green vegetables site.

A trench (10 m long by 1 m wide by 1 m deep) was excavated and shored up to house the lysimeters. During excavation, the subsoil and topsoil were carefully separated to refill lysimeters. The lysimeters were 75 L in volume, with a radius of 0.2 m and height of 0.6 m (Figure 3.7). A piece of filtering fabric was set at the bottom of each lysimeter to act as a wick and provide suction at the bottom of the lysimeter. The lysimeters were then placed with 1-2 cm separation between each other.

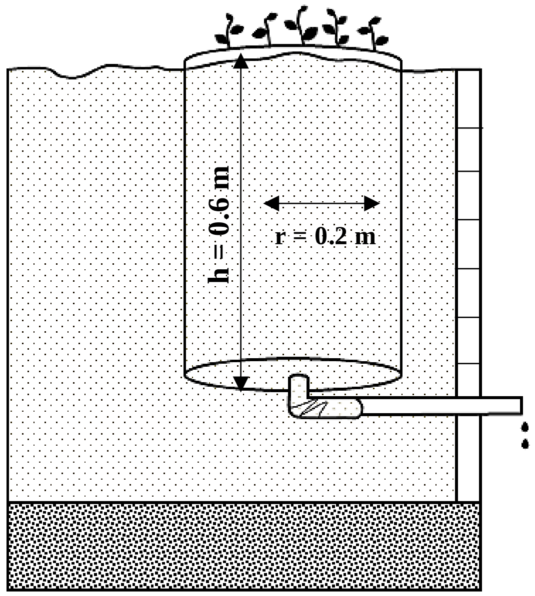

Figure 3.7. Lysimeter design.

Once the lysimeters were placed and refilled, groundwork, seeds, and fertiliser treatments were applied manually. The establishment of treatments on the lysimeters followed the procedures used for the surrounding field as closely as possible (i.e., seed density and ploughing).

The crop rotation started with a crop of beetroot sown on 16 January 2020 and harvested on 27 April 2020. The lysimeters lay fallow until a crop of Pak choi was sown on 15 January 2021. The Pak choi was harvested on 06 April 2021 and was followed by ryegrass which was sown on 26 July (Table 3.2).

In a related experiment in the field surrounding the lysimeters, treatments were applied in 40 m long by 1 m wide beds, with two replicate plots per treatment. In 2020, C, STD and RF treatments were applied using the same fertiliser rates as the lysimeters. In 2021, treatments C, STD and STD2 were applied, where the STD treatment had 50 kg N ha^{-1} applied at sowing and 50 kg N ha^{-1} applied seven weeks after sowing, and STD2, which replaced the previous RF treatment, had 100 kg N ha^{-1} applied upfront at sowing. Soil nitrate-N samples (0-20 cm) and beetroot tuber yields were measured on the field plots to compare with the lysimeter results and to help validate the APSIM model (Chapter 4).

3.3.3.2 Methodology

Soil samples (0-10, 10-20, 20-30, 30-40 and 40-55 cm of soil depth) were taken at the beginning and end of the experiment period to provide soil physical parameters in the APSIM simulations (Chapter 4). Bulk density and saturated hydraulic conductivity were analysed following procedures described by Klute (1986), while the soil water retention curve was measured using a pressure plate apparatus following the Loveday (1974) method. Ammonia-N and nitrate-N in the soil (0-20 cm soil depth) were quantified using the methods given by Blakemore et al. (1987).

All the plant biomass was harvested, and fresh weight and dry weight were measured.

Drainage water samples from lysimeters were stored in a 5 L container at the outlet pipe. Drain water samples from accumulated drainage were collected in 1 L bottles after intense rainfall events. The collected drain water samples from lysimeters were analysed for nitrate-N concentrations using the procedures described in the Methodology of the Potatoes site section (Table 3.6).

Four piezometers were installed to depths of 1 m (x2), 3 m and 5 m from ground level in the field to monitor the level and quality of shallow groundwater (Figure 3.8 and Figure 3.9). The 5-m-depth piezometer was screened (perforations) at the bottom 2 m, the 3-m-depth piezometer had 1 m screening at the bottom, and the remaining two piezometers were screened for the entire 1 m length. The space between the piezometers and surrounding soil was refilled with washed sand, and a layer of bentonite was then placed at the surface to seal the surroundings of the top of the piezometers and prevent the entrance of surface water, such as runoff or rainfall. A cap was also placed on the piezometers for the same purpose.

Once installed and operating, groundwater from piezometers was purged at least 3 times its volume with a peristaltic pump before sampling (U.S. Environmental Protection Agency, 2013). Water was then collected in 100 mL containers and nitrate-N was analysed with a TriOS OPUS spectral sensor.

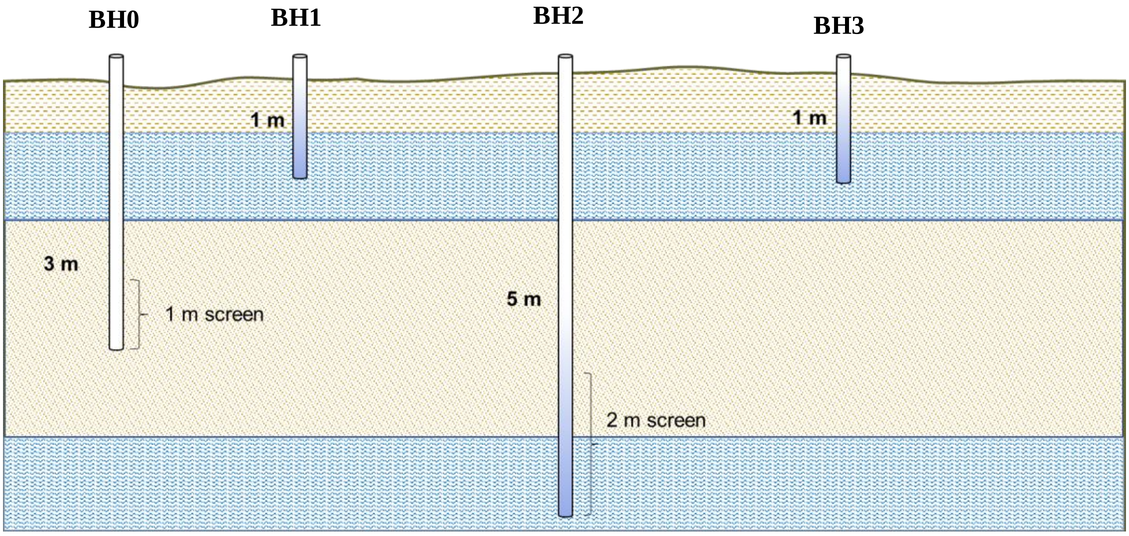

Figure 3.8. Piezometers placement at the Green vegetables site. Note that the piezometers at 1 m captured shallow (perched) groundwater, there was no water in the 3 m piezometer, and the 5 m piezometer captured deeper ground water (BH1 for piezometers 1 (1 m depth), BH2 for piezometer 2 (5 m depth) and BH3 for piezometer 3 (1 m depth)).

Figure 3.9. Location of piezometers in the field at the Green vegetables site (BH1 for piezometers 1 (1 m depth), BH2 for piezometer 2 (5 m depth) and BH3 for piezometer 3 (1 m depth)).

3.4 Results

Annual rainfall for the study area was 1056 mm in 2020 and 1270 mm in 2021. The largest rainfall during the sampling period occurred in the month of December in both 2020 and 2021 (Figure 3.10). Most of the rainfall in the year occurred during winter-early spring (June to September).

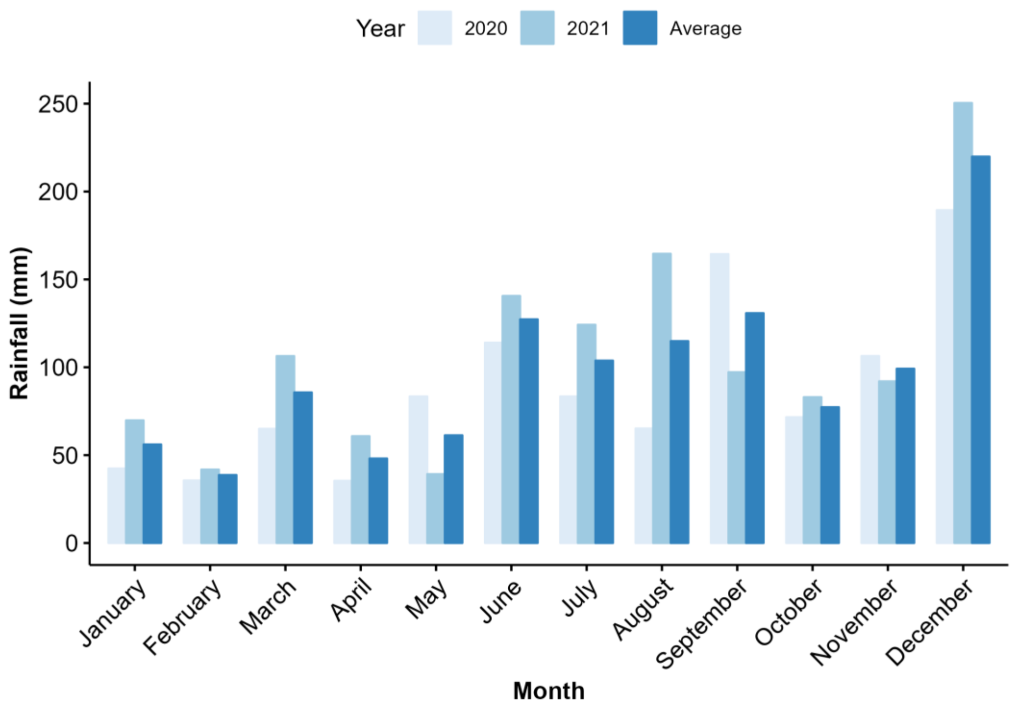

Figure 3.10. Rainfall during the study period in the Arawhata catchment (light blue bars for year 2020, darker blue bars for year 2021 and the darkest blue bars for average)

3.4.1 Potato site

3.4.1.1 Soil characterisation

Soil physical properties described here are similar to those reported by Palmer & Wilde (2007) for a Waitohu silt loam soil (Table 3.9). Dry bulk density is lowest in the first 20 cm of soil profile due to tillage and ridge formation for potatoes. The density increases in the 60-110 cm soil depth due to natural compaction.

Table 3.9. Mean bulk density, porosity and volumetric moisture content of the Waitohu silty clay loam at field capacity and permanent wilting point. Standard deviation given in brackets.

Depth (cm)	Mean bulk density (g cm^{-3})	Mean porosity	Mean volumetric moisture at Field Capacity [-10 kPa]	Mean volumetric moisture at Wilting Point [-1500 kPa]	Plant Available Water (mm)	Initial mineral N (kg ha^{-1})
0-10	0.87 (0.08)	0.67 (0.03)	0.31 (0.03)	0.19 (0.01)	11.5	8.5
10-20	0.93 (0.07)	0.65 (0.03)	0.33 (0.02)	0.23 (0.02)	11	8.5
20-30	1.03 (0.04)	0.61 (0.02)	0.37 (0.02)	0.27 (0.01)	10.7	8.5
30-40	1.11 (0.12)	0.58 (0.04)	0.41 (0.02)	0.28 (0.01)	12.8	8.1
40-60	1.17 (0.05)	0.56 (0.02)	0.44 (0.01)	0.31 (0.04)	13.3	8.1
60-80	1.49 (0.12)	0.44 (0.04)	0.41 (0.05)	0.30 (0.10)	22.8	8.1
80-100	1.47 (0.11)	0.45 (0.04)	0.41 (0.04)	0.30 (0.01)	22.4	-
100-110	1.51 (0.04)	0.43 (0.02)	0.39 (0.02)	0.26 (0.05)	12.4	-

Plant available water values are low for a silt loam soil, with 72.6 mm in the first 60 cm soil depth and 130.2 mm in the soil profile (Foundation for Arable Research, 2010). In contrast, silt loam soils usually have at least 93 mm in the 0-60 cm of soil, and 155 mm in the 0-100 cm of soil (Foundation for Arable Research, 2010). Plant available water values reported here may correspond better to those of clay loam soils, which usually have between 110 to 150 mm in the soil profile.

3.4.1.2 First potato growing season

Topsoil (0-30 cm) moisture content

A GAMLSS model with Weibull distribution indicated no statistically significant differences in mean topsoil (0 – 30 cm) moisture content between treatments over the sampling period (Table 3.10). Mean values of all treatments increased over time (i.e., from March to September) because of increased rainfall and lower evapotranspiration over the winter season.

The measured topsoil moisture content values fitted within the permanent wilting point and saturation (porosity) ranges given in Table 3.9, and during winter the topsoil was often at field capacity.

Table 3.10. Mean volumetric moisture content in the 0-30 cm soil depth for each treatment on each sampling date (C for control; CR for controlled release fertiliser; LF for split liquid fertiliser; and STD for standard practice). Standard deviation given in brackets.

Treatment/Date	**12/03/2020**	**18/05/2020**	**25/06/2020**	**2/08/2020**	**9/09/2020**
C	20.4 (1.6)	29.0 (3.0)	29.4 (0.4)	32.8 (2.5)	35.0 (2.9)
CR	25.5 (2.0)	29.9 (1.0)	29.9 (0.6)	32.7 (1.2)	34.8 (3.1)
LF	22.7 (2.9)	29.9 (0.2)	31.2 (2.5)	32.7 (1.8)	33.7 (1.7)
STD	22.3 (2.9)	27.8 (2.6)	29.6 (1.7)	32.6 (2.0)	33.7 (2.2)

Subsoil (30 – 60 cm) moisture content

Unlike the topsoil moisture contents, subsoil moisture contents had high within-treatment variability in all treatments (Table 3.11). An analysis of variances (ANOVA) indicated no statistically significant difference between the means of subsoil (30-60 cm) moisture content of treatments. There was a general trend of soil moisture content increasing over time.

Table 3.11. Mean volumetric moisture content in the 30-60 cm soil depth for each treatment on each sampling date (C for control; CR for controlled release fertiliser; LF for split liquid fertiliser; and STD for standard practice). Standard deviation given in brackets.

Treatment/Date	**12/03/2020**	**18/05/2020**	**25/06/2020**	**2/08/2020**	**9/09/2020**
C	30.8 (5.4)	39.5 (7.3)	38.7 (1.5)	41.4 (3.0)	41.0 (9.3)
CR	36.2 (2.2)	40.4 (3.6)	38.9 (3.7)	38.7 (10.0)	42.3 (1.9)
LF	33.5 (7.0)	38.9 (1.7)	40.4 (2.3)	41.9 (4.9)	38.6 (6.9)
STD	31.0 (2.4)	35.9 (8.3)	38.6 (3.2)	41.2 (6.4)	37.1 (4.9)

Topsoil (0-30 cm) nitrate-N concentrations

The topsoil nitrate-N values for the control treatment (C) were relatively consistent during the whole sampling period and this was the treatment with the lowest concentrations (Figure 3.11). Amongst the fertilised treatments, the control release fertiliser treatment (CR) had relatively low topsoil nitrate-N concentrations and the standard practice (STD) had the highest values in the early part of the season.

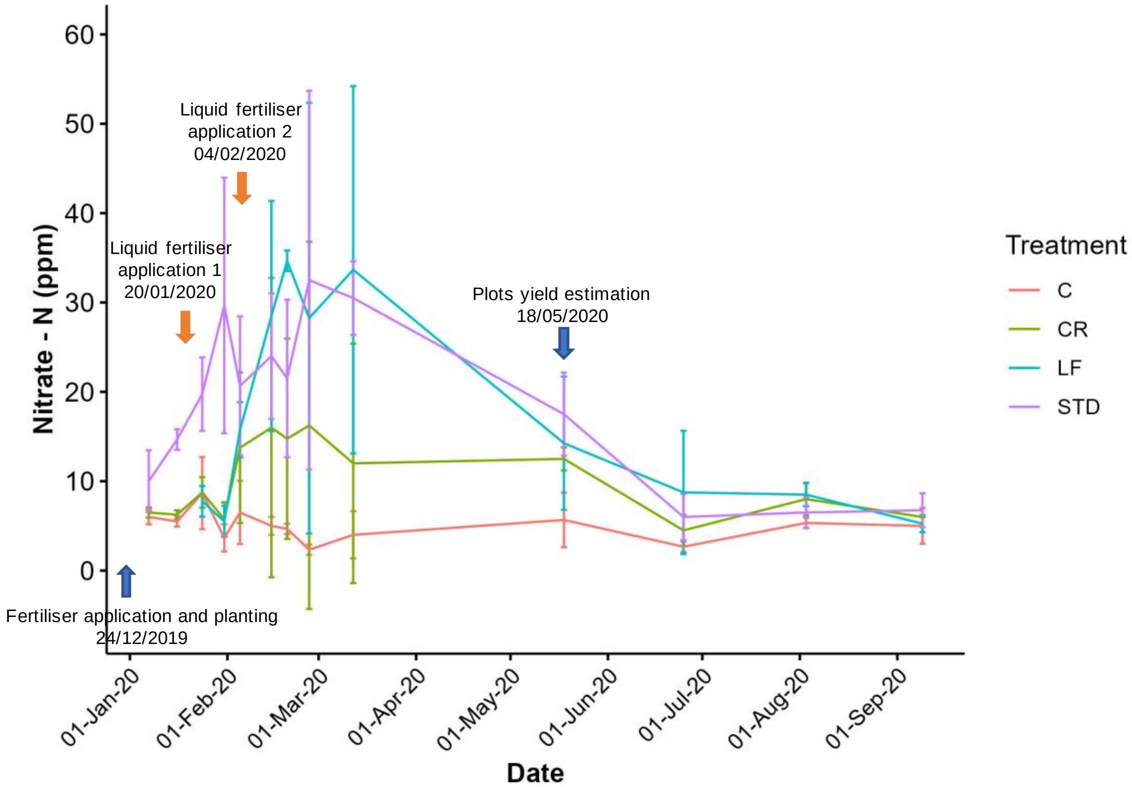

Figure 3.11. Mean nitrate-N concentrations in the 0-30 cm of soil from each treatment. Bars indicate standard deviations for each treatment (C for control; CR for controlled release fertiliser; LF for split liquid fertiliser; and STD for standard practice).

These results were fitted into a GAMLSS with log-normal distribution to estimate whether there was a substantial effect of time in the variability of the treatments (Table 3.12). The time variable was incorporated in the model as "days from sowing". The GAMLSS showed that there was a statistically significant difference between at least two treatments ($p<0.05$) (identified in the next paragraph) and there was a statistically significant effect of time in the STD treatment (p=0.00725), and some effect on the LF treatment (p=0.07435).

Table 3.12. GAMLSS for topsoil nitrate-N concentrations (CR for controlled release fertiliser; LF for split liquid fertiliser; and STD for standard practice).

Variable/Parameter	Estimate	Std. Error	t value	Pr(>|t|)	Significance^
(Intercept)	1.58	0.14	11.07	< 2e-16	***
TreatmentCR	0.62	0.19	3.22	0.00	**
TreatmentLF	1.31	0.21	6.23	3.40E-09	***
TreatmentSTD	1.62	0.19	8.27	3.26E-14	***
Days.from.sowing	-0.00	0.00	-0.71	0.48	
TreatmentCR:Days.from.sowing	-0.00	0.00	-0.17	0.86	
TreatmentLF:Days.from.sowing	-0.00	0.00	-1.79	0.07	
TreatmentSTD:Days.from.sowing	-0.00	0.00	-2.72	0.00	**

^Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05

A pairwise comparison between estimated marginal means from the model indicated statistically significant differences between treatments (Table 3.13). The C treatment had the lowest mean topsoil nitrate-N concentration, followed by the CR treatment (10.0 ppm) and finally both the LF treatment (16.6 ppm) and the STD treatment (18.3 ppm) which were not significantly different to each other. Notice that the mean soil nitrate-N concentration was 45% lower in the CR treatment than in the STD treatment.

Table 3.13. Overall mean nitrate-N concentration of topsoil (0-30 cm) and estimated marginal means (EMM) for the first season of the Potatoes site (C for control; CR for controlled release fertiliser; LF for split liquid fertiliser; and STD for standard practice). Standard deviation given in brackets.

Treatment	Observed mean (ppm nitrate-N)	EMM (log scale)^
C	5.0 (2.2)	1.5 Aa
CR	10.1 (9.0)	2.1 Bb
LF	16.6 (14.2)	2.5 Cc
STD	18.3 (11.6)	2.7 Cc

^ Different letters indicate significant differences between means of treatments (upper-case p<0.01 and lower-case p<0.05, Tukey method for p-value adjustment).

Subsoil (30-60 cm) nitrate-N concentration

The C treatment had the smallest mean nitrate-N concentrations during the sampling period (Figure 3.12). Nitrate-N concentrations in the fertilised treatment were larger than in the C treatment.

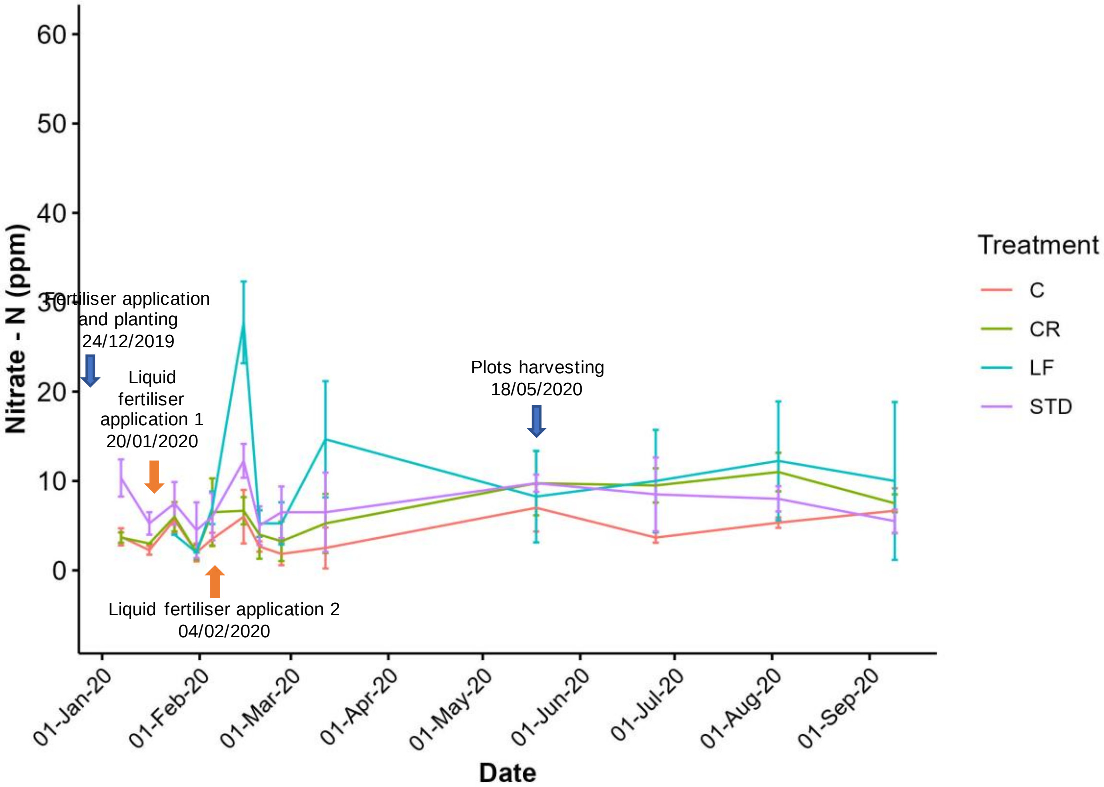

Figure 3.12. Mean nitrate-N concentrations in the 30-60 cm of soil from each treatment. Bars indicate standard deviations for each treatment (C for control; CR for controlled release fertiliser; LF for split liquid fertiliser; and STD for standard practice).

The subsoil (30-60 cm) mean nitrate-N concentrations were also fitted into a log-normal distribution GAMLSS analysis and pairwise comparison (Table 3.14). There was a statistically significant difference between the estimated marginal means from the model for the fertilised treatments and the C treatment. There was no statistically significant difference between the fertilised treatments.

Table 3.14. Overall mean nitrate-N concentration of subsoil (30 – 60 cm) and estimated marginal means (EMM) for the first season of the Potatoes site (C for control; CR for controlled release fertiliser; LF for split liquid fertiliser; and STD for standard practice). Standard deviation given in brackets.

Treatment	Observed mean (ppm nitrate-N)	EMM (log)^
C	4.0 (2.2)	1.2 Aa
CR	6.0 (3.4)	1.6 Bb
LF	9.6 (8.0)	1.9 Bb
STD	7.3 (3.2)	1.9 Bb

^ Different letters indicate significant differences between means of treatments (upper-case $p<0.01$ and lower-case $p<0.05$, Tukey method for p-value adjustment).

Crop yields

The C treatment had the smallest mean potato yield at 28 t ha^{-1}, followed by the LF treatment with 36 t ha^{-1}, the CR treatment with 42 t ha^{-1}, and the STD treatment with 52 t ha^{-1} (Figure 3.13). There were statistically significant differences in potato yields between all treatments.

The CR treatment achieved the second highest potato yield (Figure 3.13), whilst having the lowest soil nitrate-N concentrations of the fertilised treatments (Figures 3.11 and 3.12).

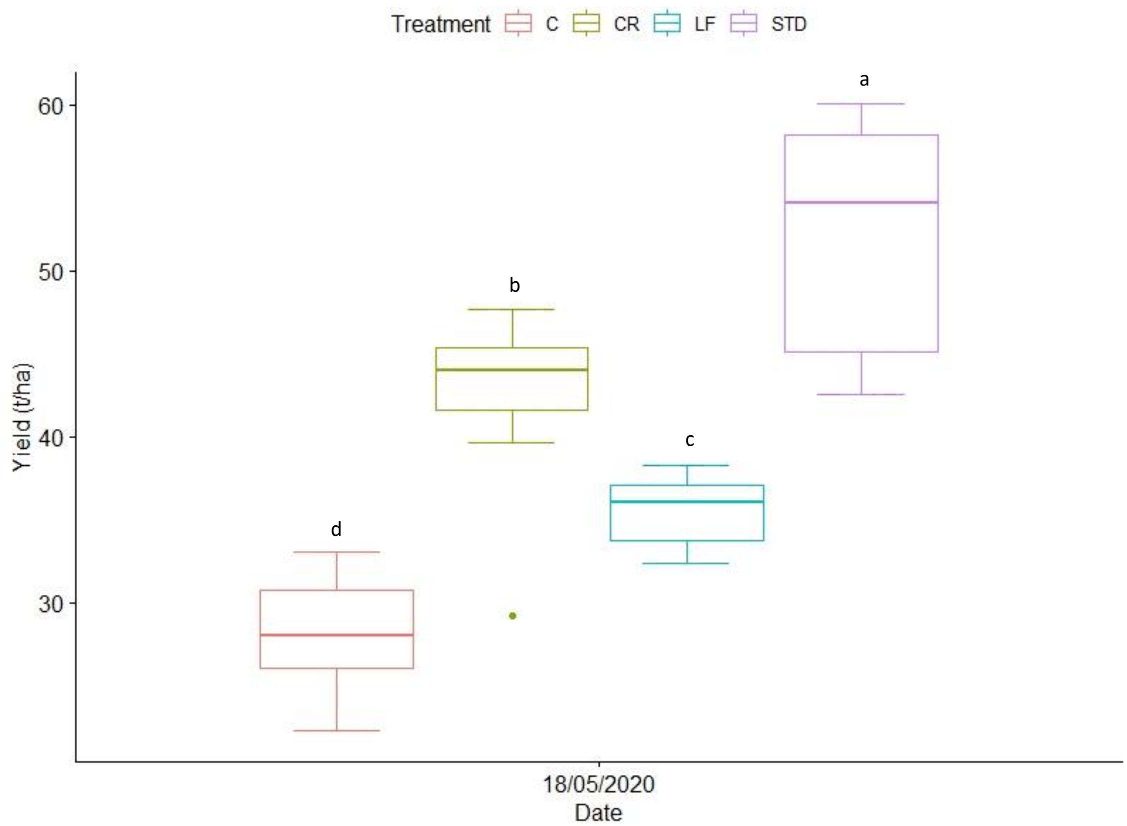

Figure 3.13. Potato yield medians, quartiles, maximums and minimums (t ha^{-1}) for each treatment (C for control; CR for controlled release fertiliser; LF for split liquid fertiliser; and STD for standard practice). Different letters indicate statistically significant differences between means of treatments ($p<0.05$, Tukey method for p-value adjustment).

3.4.1.3 Second potato growing season

Topsoil (0-30 cm) moisture content

The soil moisture sampling procedure for the second potato growing season was changed to investigate if more uniform soil moisture values could be obtained compared to the first crop season, particularly at depth (section 3.3.2.2).

A GAMLSS model with Weibull distribution indicated no statistically significant difference in mean topsoil (0 – 30 cm) moisture content between the treatments and, similar to the first season, the soil moisture content measurements increased over time (Table 3.15). The mean moisture content ranged between 25 and 35% across all the treatments.

Table 3.15. Mean volumetric moisture content in the 0-30 cm soil depth for each treatment on each sampling date (C for control; STD for standard practice; GP for good practice and EXC for excess practice). Standard deviation given in brackets.

Treatment/Date	**16/01/2021**	**17/02/2021**	**16/03/2021**	**16/04/2021**
	% cm^3 cm^{-3}			
C	26.8 (3.7)	34.4 (1.5)	32.0 (1.2)	35.2 (0.9)
STD	26.3 (2.0)	35.0 (2.2)	31.7 (1.7)	34.0 (1.2)
GP	25.1 (2.0)	35.1 (2.6)	32.5 (1.5)	34.6 (0.7)
EXC	25.6 (2.5)	34.4 (2.3)	31.8 (2.0)	34.3 (1.2)

Potatoes are sensitive to water stress, so it is recommended to keep the soil moisture content above 70% of plant available water in the first 60 cm of soil (Foroud et al., 1993). In general, the mean topsoil moisture content over both of the potato seasons was near this value (equivalent to a volumetric moisture content of 31%), except for the first measurements in March 2020 and in January 2021.

Subsoil (30-60 cm) moisture content

The moisture content results in the subsoil were highly variable in the second sampling period, ranging from 34% in the GP treatment to 45% in the C treatment (Table 3.16). According to a GAMLSS model with log-normal distribution, there was no significant difference between the means of subsoil (30-60 cm) moisture content of treatments.

Table 3.16. Mean volumetric moisture content in the 30-60 cm soil depth for each treatment on each sampling date (C for control; STD for standard practice; GP for good practice and EXC for excess practice). Standard deviation given in brackets.

Treatment/Date	**16/01/2021**	**17/02/2021**	**16/03/2021**	**16/04/2021**
	% cm^3 cm^{-3}			
C	40.6 (3.9)	45.0 (5.5)	40.6 (3.4)	45.0 (6.0)
STD	37.7 (5.5)	38.6 (5.0)	38.7 (4.5)	37.4 (3.2)
GP	34.2 (1.7)	36.4 (3.0)	38.6 (4.1)	39.3 (5.6)
EXC	34.1 (2.1)	40.9 (4.3)	37.6 (2.8)	38.6 (3.0)

The mean moisture content in the 30-60 cm depth was near to a value of 70% of plant available water (equivalent to a volumetric moisture content of 39%) during both sampling periods except for samples taken on 16/01/2021 in GP and EXC.

<u>Topsoil (0-30 cm) nitrate-N concentration</u>

The C treatment had the smallest nitrate-N concentrations in the topsoil (0-30 cm) (Figure 3.14, presented as a bar graph to reflect the infrequent sampling). As expected, the EXC treatment showed the largest concentrations, followed by the GP treatment and the STD treatment.

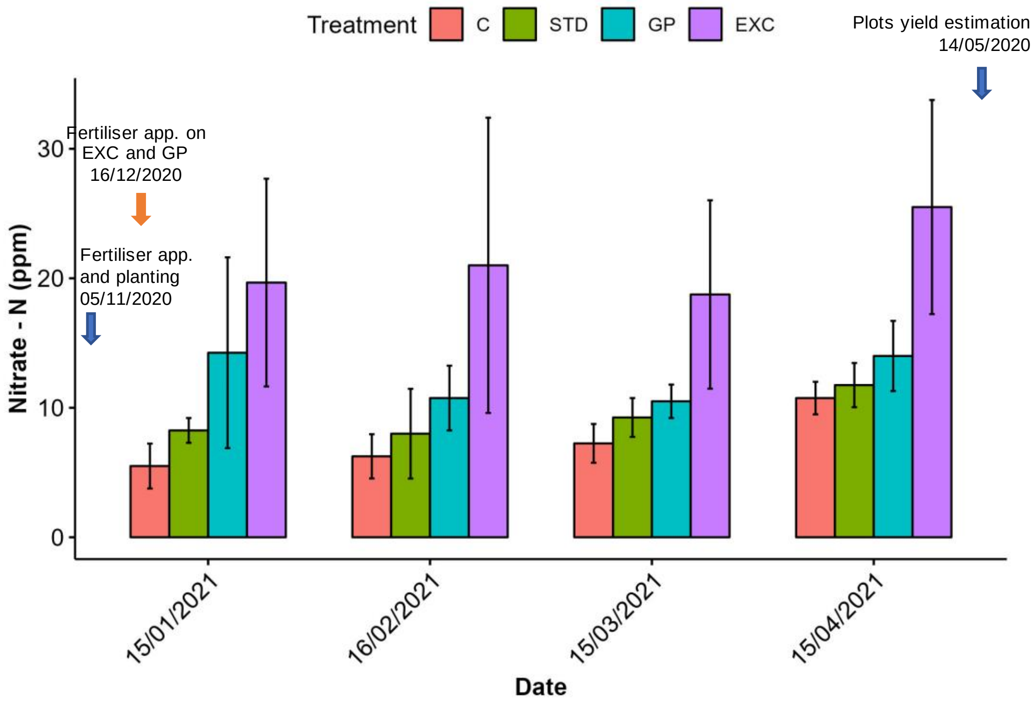

Figure 3.14. Mean nitrate-N concentrations in the 0-30 cm of soil from each treatment. Bars indicate standard deviations for each treatment (C for control; STD for standard practice; GP for good practice and EXC for excess).

The GAMLSS fitted with a lognormal distribution showed that there was a statistically significant difference between at least two treatments ($p<0.05$) (identified in the next paragraph), and there was a statistically significant effect of time on the GP treatment ($p=0.0259$) (Table 3.17).

Table 3.17. GAMLSS for topsoil nitrate-N concentrations (C for control; STD for standard practice; GP for good practice and EXC for excess).

Variable/Parameter	Estimate	Std. Error	t value	Pr(>|t|)	Significance^
(Intercept)	1.04	0.27	3.85	3.15E-04	***
TreatmentSTD	0.66	0.38	1.73	0.09	
TreatmentGP	1.37	0.38	3.57	0.00	***
TreatmentEXC	1.59	0.40	3.95	0.00	***
Days.from.sowing	0.00	0.00	3.47	0.00	**
TreatmentSTD:Days.from.sowing	-0.00	0.00	-1.12	0.27	
TreatmentGP:Days.from.sowing	-0.01	0.00	-2.30	0.03	*
TreatmentEXC:Days.from.sowing	-0.01	0.00	-1.49	0.14	

^Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05

A pairwise comparison indicated no significant difference between the STD and C treatments (p=0.0858) (Table 3.18), but there were statistically significant differences between the mean nitrate-N concentration of the GP and EXC treatments, and the EXC and STD treatments. The GP treatment's mean nitrate-N concentration was 1.66 times greater than the C treatment's mean nitrate-N concentration, and the EXC treatment's mean nitrate-N concentration was the largest with 2.87 times the C treatment's mean nitrate-N concentration. The mean nitrate-N concentration in the EXC treatment was 129% larger than in the STD treatment.

Table 3.18. Overall mean nitrate-N concentration of topsoil and estimated marginal means (EMM) for the second season of the Potatoes site (C for control; STD for standard practice; GP for good practice and EXC for excess). Standard deviation given in brackets.

Treatment	Observed mean (ppm nitrate-N)	EMM (log scale)^
C	7.4 (2.5)	2.0 Aa
STD	9.3 (2.4)	2.2 ABa
GP	12.4 (4.2)	2.5 Bb
EXC	21.3 (8.4)	3.0 Cc

^ Different letters indicate significant differences between means of treatments (upper-case $p<0.01$ and lower-case $p<0.05$, Tukey method for p-value adjustment).

Subsoil (30-60 cm) nitrate-N concentration

The C treatment had the smallest nitrate-N concentrations during the sampling period, while the EXC treatment had the largest (Figure 3.15).

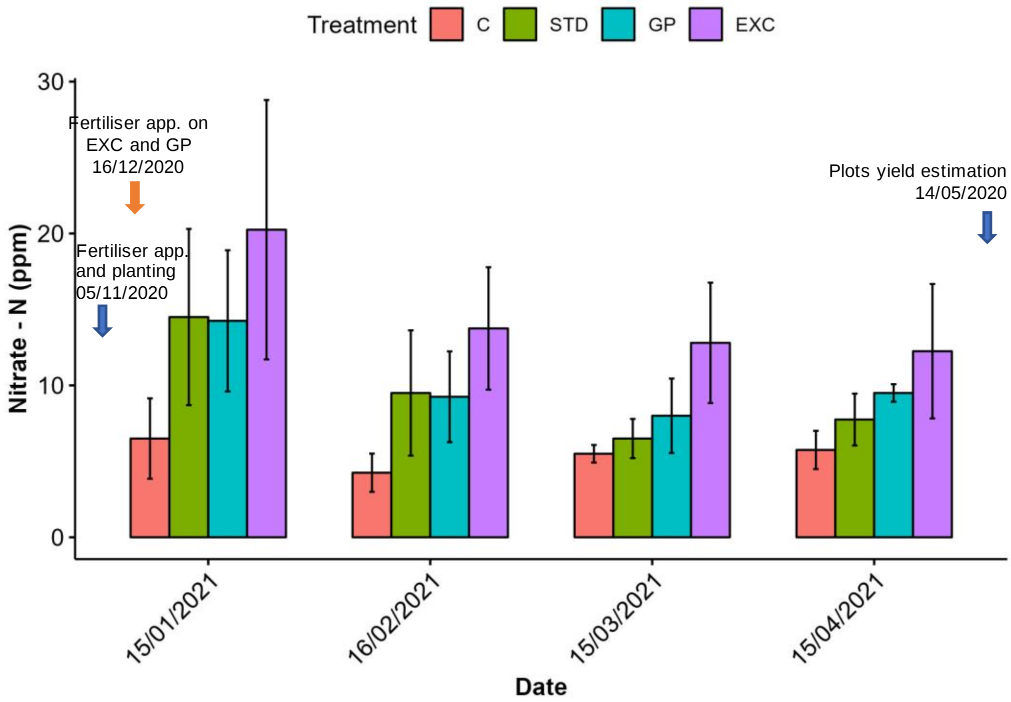

Figure 3.15. Mean nitrate-N concentrations in the 30-60 cm of soil from each treatment. Bars indicate standard deviations for each treatment (C for control; STD for standard practice; GP for good practice and EXC for excess).

The GAMLSS with lognormal distribution analysis showed that there was a statistically significant difference between at least two treatments ($p<0.05$) (identified in the next paragraph), and there was a statistically significant effect of time in the STD treatment ($p=0.0216$) (Table 3.19).

Table 3.19. GAMLSS for subsoil nitrate-N concentrations (STD for standard practice; GP for good practice and EXC for excess).

Variable/Parameter	Estimate	Std. Error	t value	Pr(>|t|)	Significance^
(Intercept)	1.66	0.26	6.46	3.10E-08	***
TreatmentSTD	1.33	0.36	3.67	0.00	**
TreatmentGP	1.11	0.36	4.23	0.00	**
TreatmentEXC	1.53	0.36	3.06	0.00	**
Days.from.sowing	0.00	0.00	0.02	0.98	
TreatmentSTD:Days.from.sowing	-0.01	0.00	-2.37	0.02	*
TreatmentGP:Days.from.sowing	-0.00	0.00	-1.69	0.10	
TreatmentEXC:Days.from.sowing	-0.01	0.00	-1.44	0.16	

^Significance codes: 0 '***' 0.001 '**' 0.01 '*' 0.05

A pairwise comparison between estimated marginal means from the model indicated statistically significant differences between the mean nitrate-N concentration of the fertilised treatments and the C treatment (Table 3.20), where the C treatment had the smallest nitrate-N concentrations (5.5 ppm). There was no statistically significant difference between the STD and the GP treatments. These treatments were 1.7 and 1.9 times greater than the C treatment, respectively. The soil nitrate-N concentration of the EXC treatment was greater ($p<0.05$) than the other fertilised treatments. The mean nitrate-N concentration in the EXC treatment was 53% and 46% greater than in the STD and GP treatments, respectively.

Table 3.20. Overall mean nitrate-N concentration of subsoil (30-60 cm) and estimated marginal means (EMM) for the second season of the Potatoes site (C for control; STD for standard practice; GP for good practice and EXC for excess). Standard deviation given in brackets.

Treatment	Observed mean (ppm nitrate-N)	EMM (log)^
C	5.5 (1.7)	1.7 Aa
STD	9.6 (4.6)	2.2 Bb
GP	10.3 (3.7)	2.3 Bb
EXC	14.7 (5.9)	2.6 Bc

^ Different letters indicate significant differences between means of treatments (upper-case $p<0.01$ and lower-case $p<0.05$, Tukey method for p-value adjustment).

Comparison of soil nitrate-N concentrations between treatments both in the surface and subsurface soil were as expected during the second season, where treatments with more N fertiliser applied (GP and EXC) had greater soil nitrate-N concentrations. The STD treatment had similar nitrate-N concentration values for topsoil and subsoil, whereas in other fertilised treatments, mean nitrate-N concentration values were higher in the topsoil.

As expected, the C treatment exhibited low and constant values of nitrate-N both in the topsoil and the subsoil, in both seasons. In the second season, mean nitrate-N concentrations in the topsoil (0-30 cm) and subsoil (30-60 cm) of the C treatment increased by 48% and 38% compared to the first season, respectively (Tables 3.13, 3.14, 3.18 and 3.20). This could reflect a residual effect of the previous potato crop including the N made available by cultivation and the decomposition of crop residue.

The STD treatment had considerably lower mean topsoil nitrate-N concentrations (decrease in 49%) and higher mean subsoil nitrate-N concentrations (increase in 32%) in the second season than in the first season.

Suction cups

The mean soil water nitrate-N concentration in suction cups in the C treatment fluctuated between 20 and 32 mg L^{-1} and were the smallest values among the treatments. In contrast, the EXC treatment had the largest concentrations, which fluctuated between 40 and 75 mg L^{-1} (Figure 3.16).

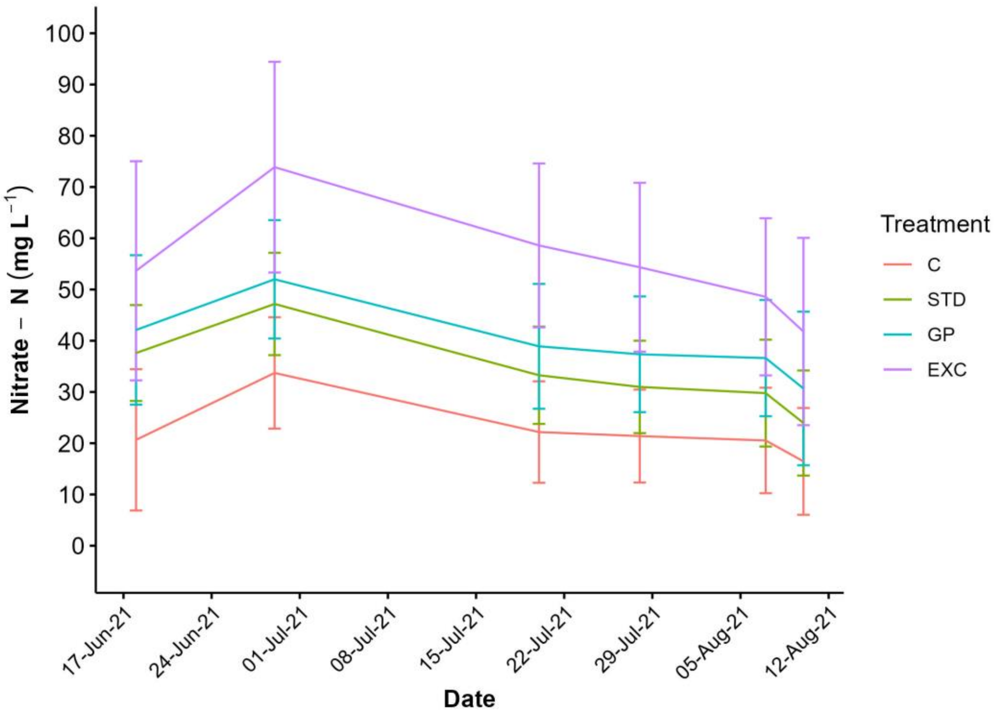

Figure 3.16. Mean soil water nitrate-N concentrations in the suction cups from each treatment. Bars indicate standard deviations for each treatment (C for control; STD for standard practice; GP for good practice and EXC for excess).

According to the linear mixed model analysis and later pairwise comparison (Table 3.21), all of the treatments had significantly different mean nitrate-N concentrations ($p<0.05$).

A more rigorous statistical analysis ($p<0.01$) indicated that there were statistically significant differences between mean nitrate-N concentrations of all treatments, except between treatments STD and GP (Table 3.21).

Table 3.21. Mean concentrations of nitrate-N in the suction cups and cumulative nitrate-N leaching rates for the period of sampling (C for control; STD for standard practice; GP for good practice and EXC for excess). Standard deviation given in brackets.

Treatment	n	Mean nitrate-N concentration ($mg\ L^{-1}$)	Cumulative nitrate-N leaching ($kg\ ha^{-1}$)
C	72	22.5 (11.7) Aa	43 (18)
STD	72	33.8 (11.9) Bb	65 (19)
GP	72	39.6 (13.9) Bc	76 (24)
EXC	71	55.2 (20.2) Cd	109 (34)

^ Different letters indicate significant differences between means of treatments (upper-case $p<0.01$ and lower-case $p<0.05$, Tukey method for p-value adjustment).

The soil water in suction cups under the C treatment had a mean concentration of approximately 22.5 mg nitrate-N L^{-1}, the lowest overall mean concentration. The soil water collected under the STD and GP treatments had mean nitrate-N concentrations which were 1.5 and 1.76 times greater than the C treatment, respectively. The soil water in suction cups under the EXC treatment had the highest mean nitrate-N concentration (55.2 mg nitrate-N L^{-1}), which was 145% and 63% greater than the C and STD treatments, respectively.

Cumulative nitrate-N leaching rates were calculated by using a water balance model to estimate drainage (Chapter 4) and nitrate-N concentrations in the drainage water (Figure 3.16). A simple independent water balance (Allen et al., 1998) generated very similar drainage rates to APSIM estimates. Nitrate-N leaching rates between May to August 2021 were calculated as 43, 65, 76 and 109 kg N ha^{-1} for treatments C, STD, GP and EXC, respectively (Table 3.21).

Crop yields

The C treatment had the lowest mean potato yield with 16.5 t ha^{-1}, followed by the STD treatment with 36.3 t ha^{-1}, GP treatment with 45.4 t ha^{-1}, and EXC treatment with 45.1 t ha^{-1} (Figure 3.17). All treatments showed significantly greater yields than the C treatment, but there were no significant differences between the yield of the treatments.

The harvested yield of the STD treatment was 15.8 t ha^{-1} less for the second season than for the first season.

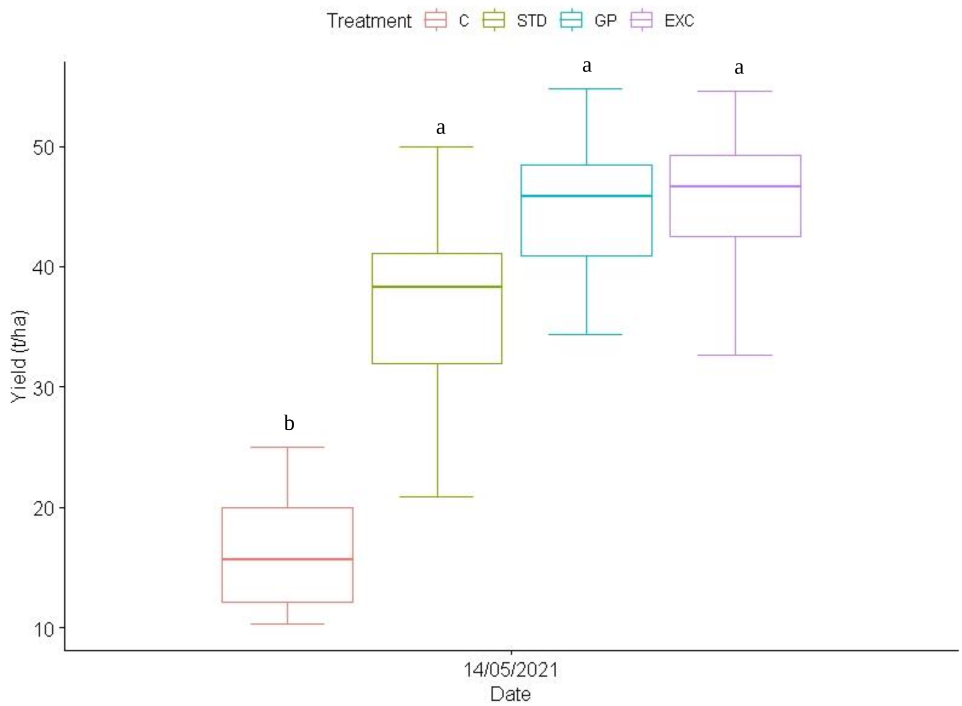

Figure 3.17. Potato yield medians, quartiles, maximums, and minimums (t ha^{-1}) for each treatment (C for control treatment; STD for standard practice treatment; GP for good practice treatment and EXC for excess treatment). Different letters indicate significant differences between means of treatments ($p<0.05$, Tukey method for p-value adjustment).

3.4.1.4 Onion growing season

For the onion season, a standard fertiliser programme (i.e., 126 kg N ha^{-1}) was applied to the field and nitrate-N concentrations were monitored with suction cups in beds where the previous C and STD treatments had been during the potato growing season (Figure 3.18).

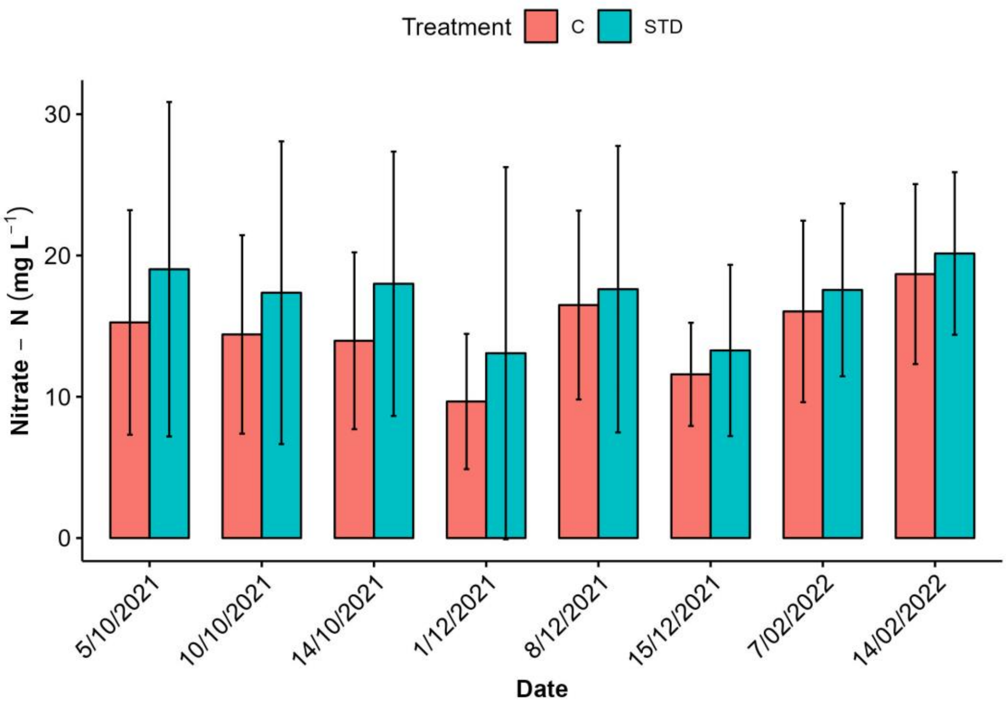

Figure 3.18. Mean soil water nitrate-N concentrations in the suction cups from two treatments under onions (C for control, and STD for standard practice). Bars indicate standard deviations for each treatment.

A GAMLSS was fitted into this data with a gamma distribution, which revealed no significant differences between the treatments ($p=0.0831$). Notice that the p-value is close to the significance threshold of 0.05. The means of soil nitrate-N in the suction cups from both treatments are summarised in Table 3.22.

Table 3.22. Mean concentrations of soil water nitrate-N in the suction cups under onions for the period of sampling (C for control, and STD for standard practice). Standard deviation given in brackets.

Treatment	n	Observed mean (mg nitrate-N L^{-1})
C	94	14.6 (6.8)
STD	93	16.9 (9.8)

Cumulative nitrate-N leaching rates for the onion season (October 2021 to February 2022) were calculated by estimated drainage volumes from a water balance model (Chapter 4) and nitrate-N concentrations in the drainage water (Figure 3.18). Nitrate-N leaching rates resulted in 27 kg N ha^{-1} for the C treatment and 30 kg N ha^{-1} for the STD treatment. Cumulative nitrate-N leaching between May 2021 and February 2022 in the treatment following the standard practice of the grower (STD) was calculated as 95 kg N ha^{-1}.

3.4.2 Green vegetables site

3.4.2.1 Soil characterisation

The soil in the lysimeters was not compacted and had homogeneous bulk density values throughout the profile (Table 3.23). Plant available water values are on the small side for a silt loam soil, with 88.4 mm in the soil profile of 55 cm (Foundation for Arable Research, 2010).

Table 3.23. Mean bulk density, porosity and volumetric moisture content of the soil at field capacity and permanent wilting point measured in lysimeters of Green vegetables site. Standard deviation given in brackets.

Depth (cm)	Mean bulk density (g cm^{-3})	Mean porosity	Mean volumetric moisture at Field Capacity [-10 kPa]	Mean volumetric moisture at Wilting Point [-1500 kPa]	Plant Available Water (mm)	Initial mineral N (kg ha^{-1})
0-10	1.08 (0.03)	0.59 (0.01)	0.39 (0.02)	0.22 (0.01)	17.1	50
10-20	1.15 (0.02)	0.57 (0.01)	0.42 (0.02)	0.26 (0.00)	16.2	54
20-30	1.13 (0.01)	0.57 (0.00)	0.43 (0.01)	0.25 (0.01)	17.4	8.4
30-40	1.06 (0.02)	0.60 (0.01)	0.39 (0.00)	0.24 (0.00)	14.6	20
40-55	1.05 (0.04)	0.60 (0.02)	0.40 (0.03)	0.25 (0.01)	22.5	59

Despite the careful repacking of soil into the lysimeters, the physical characteristics of the soil in the lysimeters were different from the surrounding field soil. The soil in the lysimeters had lower values for bulk density, moisture content at field capacity and permanent wilting point than the surrounding field soil. These differences should be considered when comparing results from both areas.

Topsoil (0-20 cm) moisture content

Mean topsoil (0-20 cm) moisture content values were fitted into a GAMLSS model with log-normal distribution, which indicated no significant differences between treatments. Measurements mean values ranged between 25 and 40% over time (Table 3.24).

Table 3.24. Mean volumetric moisture content in the 0-20 cm soil depth for each treatment on each date of sampling (C for control; STD for standard practice; CM for chicken manure, RF for reduced fertiliser programme and EXC for excess). Standard deviation given in brackets.

Treatment/Date	12/11/2020	22/05/2020	9/04/2021	19/11/2021
C	38.4 (2.0)	29.6 (0.5)	28.1 (2.8)	35.9 (1.5)
STD	37.8 (1.5)	28.7 (1.0)	28.0 (5.6)	35.1 (2.2)
CM	38.7 (3.0)	28.9 (1.0)	28.8 (2.1)	36.2 (2.0)
RF/EXC	37.5 (1.6)	28.6 (0.6)	25.4 (2.2)	35.5 (1.6)

Topsoil (0-20 cm) nitrate-N concentration

The treatments had similar values for mean topsoil nitrate-N concentration over time (Figure 3.19). The last two measurements (09-Apr-2021 and 19-Nov-2021) were the lowest for the sampling period, mainly because there were no further N fertiliser applications.

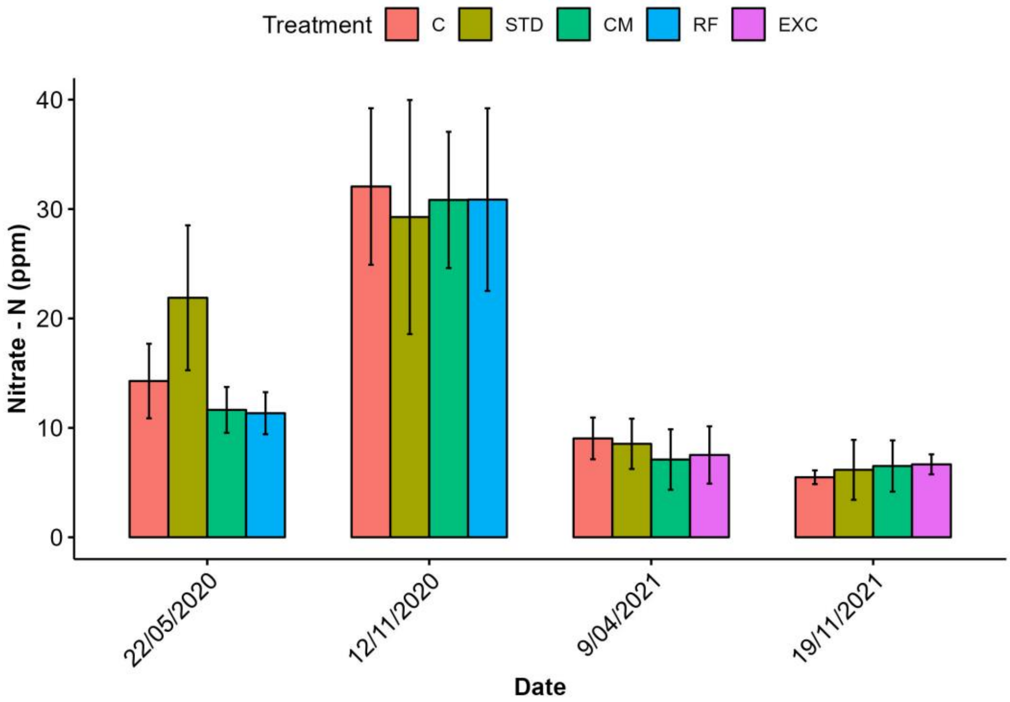

Figure 3.19. Mean nitrate-N concentrations in the 0-20 cm of soil from each treatment in the lysimeters. Bars indicate standard deviations for each treatment (C for control; STD for standard practice; CM for chicken manure, RF for reduced fertiliser programme and EXC for excess).

A GAMLSS fitted with log-normal distribution showed no significant differences between the treatments. The means of nitrate-N concentration observed in different treatments are summarised in Table 3.25.

Table 3.25. Overall mean nitrate-N concentration of topsoil for the sampling period (C for control; STD for standard practice; CM for chicken manure, RF for reduced fertiliser programme and EXC for excess). Standard deviation given in brackets.

Treatment	Observed mean (ppm nitrate-N)
C	16.3 (11.2)
STD	18.1 (11.6)
CM	14.9 (11.0)
RF/EXC	14.9 (11.2)

3.4.2.2 First year

Drainage – first year

Most drainage events occurred during the winter-spring period, however, some heavy early summer rainfalls also contributed to drainage volumes from the lysimeters (Figure 3.20). Cumulative drainage within treatments was highly variable.

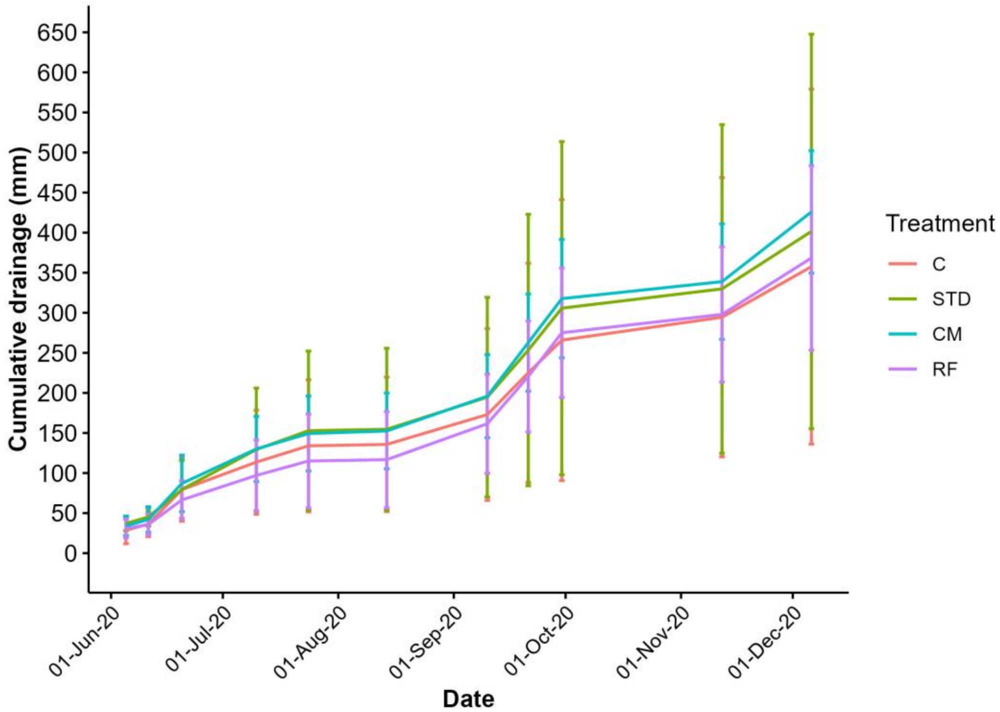

Figure 3.20. Mean cumulative drainage volumes from the lysimeters for the first year of sampling from each treatment. Bars indicate standard deviations for each treatment (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertiliser programme).

The cumulative drainage data was fitted into a GAMLSS with a Weibull distribution. This analysis indicated that there were no significant differences between the treatments. The means of cumulative drainage observed in different treatments are summarised in Table 3.26.

Table 3.26. Mean cumulative drainage and proportion of rainfall for the first year of monitoring (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertiliser programme). Standard deviation given in brackets.

Treatment	Cumulative drainage (mm)	Proportion of rainfall (%)
C	358 (222)	39
STD	402 (246)	44
CM	426 (76)	47
RF	368 (115)	40

Nitrate-N concentrations and leaching fluxes - first year

The STD treatment had the highest nitrate-N concentrations in the earlier drainage events, but in general, values observed during the first season were similar between treatments (Figure 3.21).

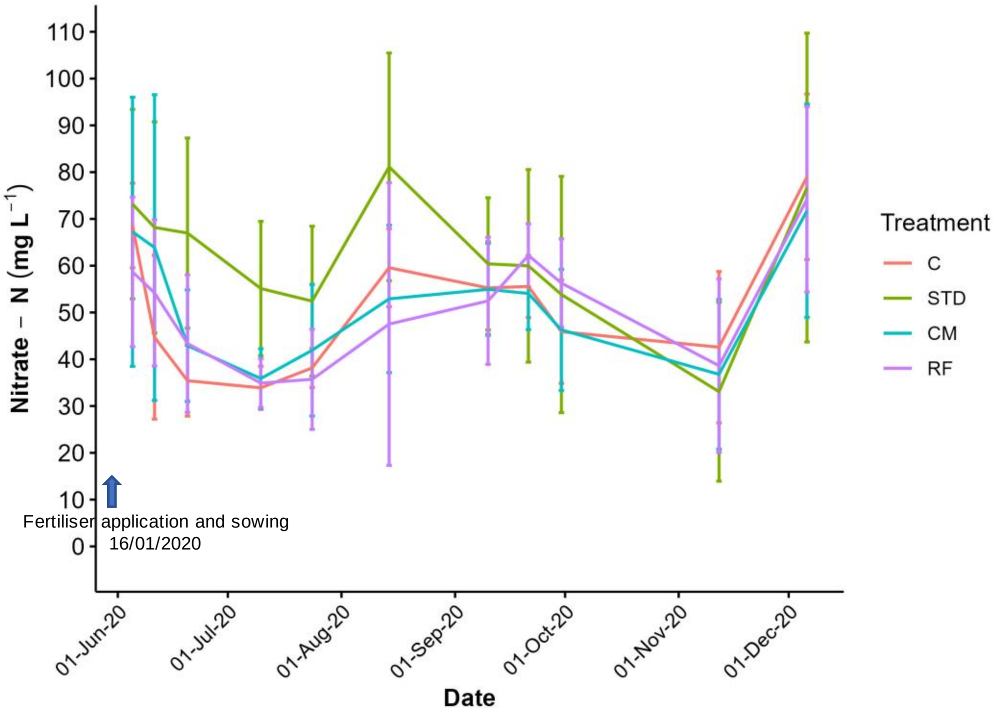

Figure 3.21. Mean nitrate-N concentrations in drainage for the first year of sampling from each treatment. Bars indicate standard deviations (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertiliser programme).

A gamma distributed GAMLSS and further pairwise comparison revealed that there were statistically significant differences between the mean nitrate-N concentrations in drainage from the STD treatment and the RF treatment (p=0.0497) (Table 3.27). The differences between the STD treatment, and the CM and C were not significant (p=0.08 and p=0.06, respectively). However, notice that the p-value is close to the significance threshold of 0.05.

Table 3.27. Mean concentrations of nitrate-N in drainage from the lysimeters (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertiliser programme). Standard deviation given in brackets.

Treatment	Nitrate-N concentration in drainage (mg L^{-1})
C	52.1 (17.4) ab
STD	61.9 (20.4) b
CM	51.7 (19.7) ab
RF	51.5 (18.3) a

^ Different letters indicate significant differences between means of treatments ($p<0.01$, Tukey method for p-value adjustment)

The drainage volumes and the concentrations were used to calculate the nitrate-N flux in drainage (Figure 3.22). The C treatment appeared to have the lowest cumulative nitrate-N leaching loads, although the values of different treatments were approximately similar. One of the most notable features of these measurements of leaching from the lysimeters is their very large magnitude.

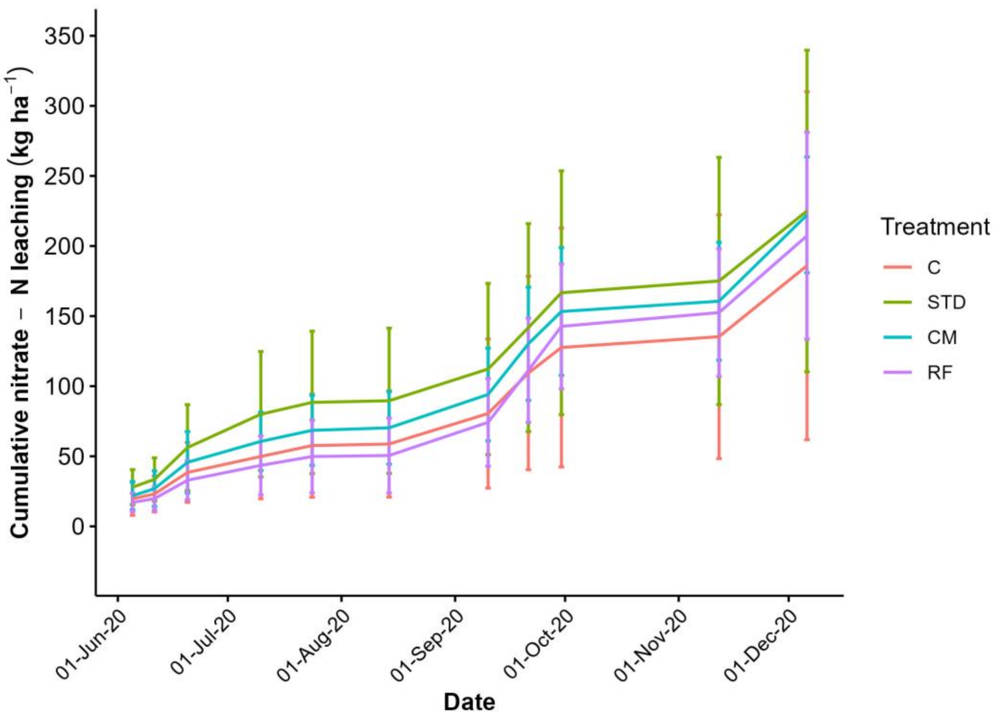

Figure 3.22. Mean cumulative nitrate-N leaching losses for the first year of sampling for each treatment in the lysimeters. Bars indicate standard deviations (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertiliser programme).

Results of cumulative nitrate-N losses were fitted into a GAMLSS following a normal distribution. This analysis showed no significant difference between the treatments. The means of cumulative nitrate-N leaching rates observed in different treatments are summarised in Table 3.28.

Table 3.28. Mean cumulative nitrate-N leaching losses for the first year of monitoring (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertiliser programme). Standard deviation given in brackets.

Treatment	**Cumulative nitrate-N leaching (kg ha^{-1})**
C	186 (124)
STD	225 (115)
CM	222 (41)
RF	207 (74)

3.4.2.3 Second year

Drainage – second year

As for the first year. drainage volume values tended to increase over the winter-spring season (Figure 3.23). As the experiment finished in October 2021, drainage volumes for the summer season of the second year were not measured.

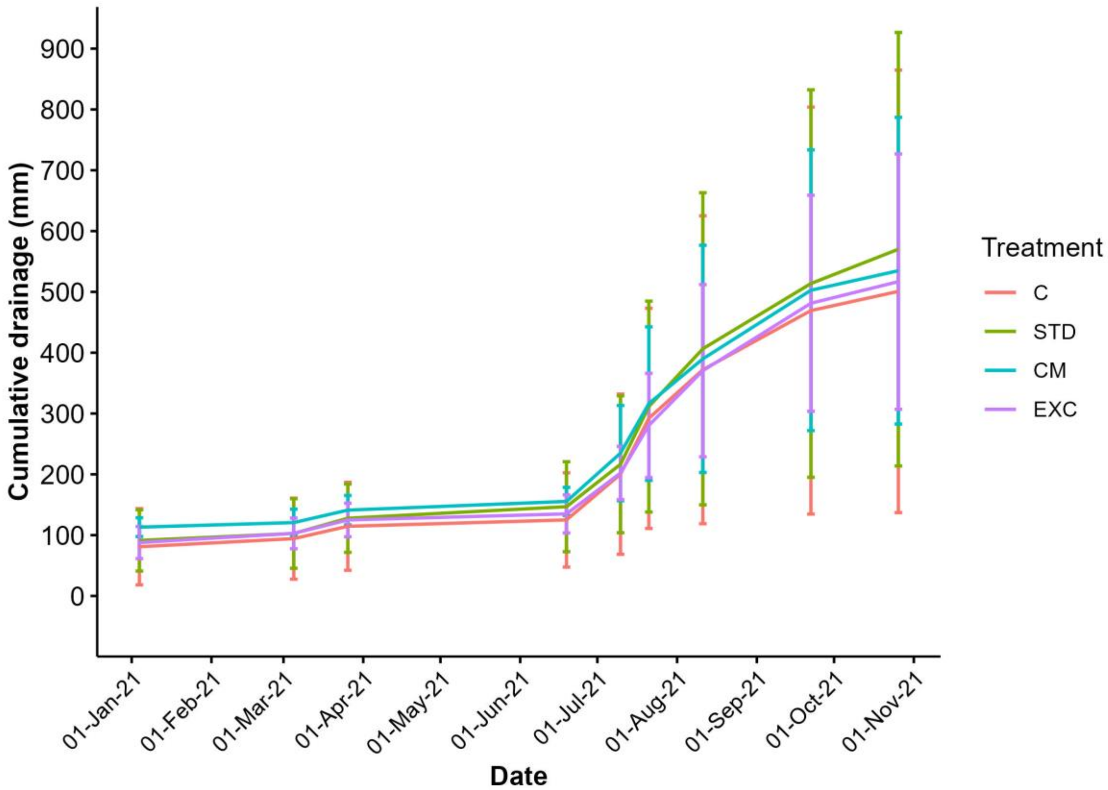

Figure 3.23. Mean cumulative drainage volumes for the second year of sampling for each treatment in the lysimeters. Bars indicate standard deviations for each treatment (C for control; STD for standard practice; CM for chicken manure and EXC for excess).

The cumulative drainage data was fitted into a GAMLSS with Weibull a distribution. This analysis indicated no significant differences between the treatments. The means of cumulative drainage observed in different treatments are summarised in Table 3.29.

Table 3.29. Mean cumulative drainage and proportion of rainfall for the second year of monitoring (C for control; STD for standard practice; CM for chicken manure and EXC for excess). Standard deviation given in brackets.

Treatment	Cumulative drainage (mm)	Proportion of rainfall (%)
C	501 (364)	47
STD	535 (252)	51
CM	517 (210)	49
EXC	570 (356)	54

Nitrate-N concentrations and leaching fluxes - second year

In the second year, nitrate-N concentration in drainage decreased over time, mostly because no fertiliser was added after January 2021 (Figure 3.24).

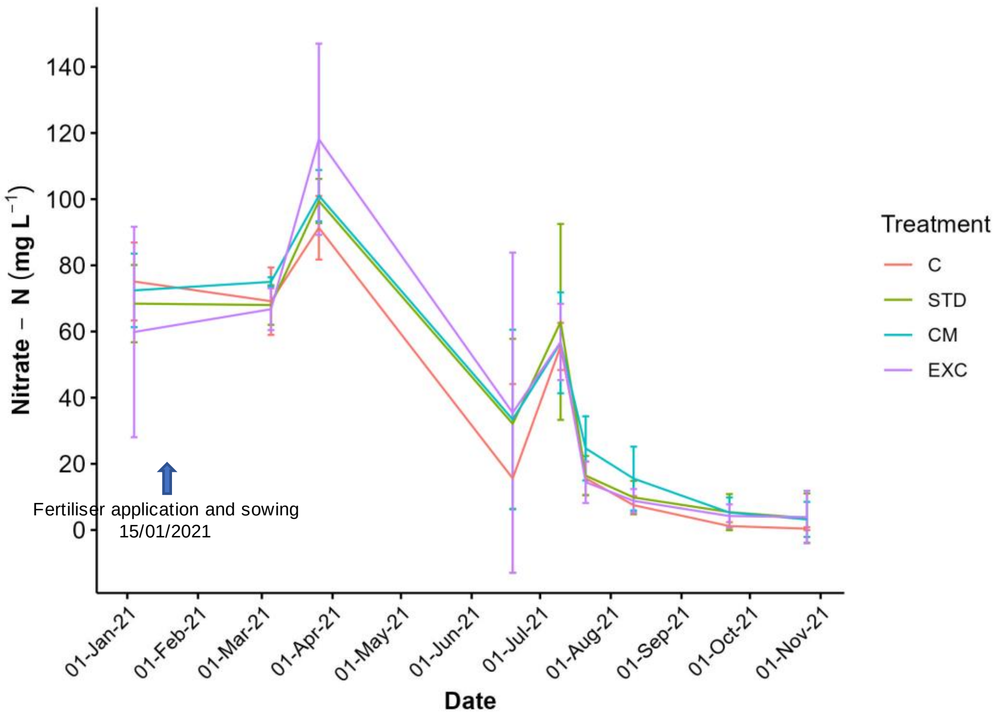

Figure 3.24. Mean nitrate-N concentrations in drainage for the second year of sampling from each treatment. Bars indicate standard deviations (C for control; STD for standard practice; CM for chicken manure and EXC for excess).

A GAMLSS fitted with gamma distribution showed that there were no statistically significant differences between the treatments. The means of nitrate-N concentrations observed in different treatments are summarised in Table 3.30.

Table 3.30. Mean concentrations of nitrate-N in drainage from the lysimeters in the second year (C for control; STD for standard practice; CM for chicken manure and EXC for excess). Standard deviation given in brackets.

Treatment	**Nitrate-N concentration in drainage (mg L^{-1})**
C	36.4 (35.5)
STD	40.1 (35.9)
CM	41.5 (35.0)
EXC	40.9 (41.7)

^ Different letters indicate significant differences between means of treatments ($p<0.01$, Tukey method for p-value adjustment)

Nitrate-N leaching losses from the treatments were calculated in the same way as the first year (Figure 3.25). The C treatment appeared to have the least nitrate-N leaching losses over time, although the values of different treatments were approximately similar.

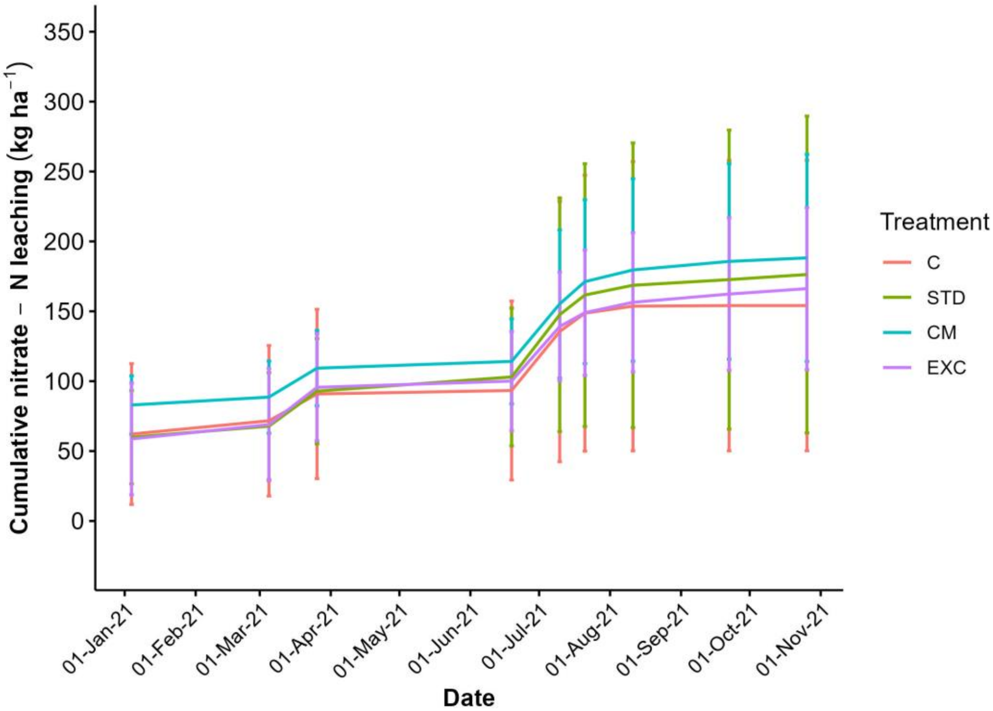

Figure 3.25. Cumulative nitrate-N leaching losses for the second year of sampling for each treatment in the lysimeters. Bars indicate standard deviations (C for control; STD for standard practice; CM for chicken manure and EXC for excess).

Results of cumulative nitrate-N were fitted into a GAMLSS following a normal distribution. This analysis showed no significant difference between treatments. The means of cumulative nitrate-N leaching rates observed in different treatments are summarised in Table 3.31. The treatment with the highest value was again the CM treatment with 188 kg nitrate-N ha^{-1}, followed by the STD, EXC and Cl treatments, with 176, 166 and 154 kg nitrate-N ha^{-1}, respectively.

Table 3.31. Mean cumulative nitrate-N leaching losses for the second year of monitoring (C for control; STD for standard practice; CM for chicken manure and EXC for excess). Standard deviation given in brackets.

Treatment	Cumulative nitrate-N leaching (kg ha^{-1})
C	154 (104)
STD	176 (113)
CM	188 (74)
EXC	166 (58)

3.4.2.4 Crop yields

The beetroot fresh tuber yield ranged from 63.3 t ha^{-1} to 73 t ha^{-1} (Figure 3.26). There were no statistically significant differences between the treatments (Figure 3.26).

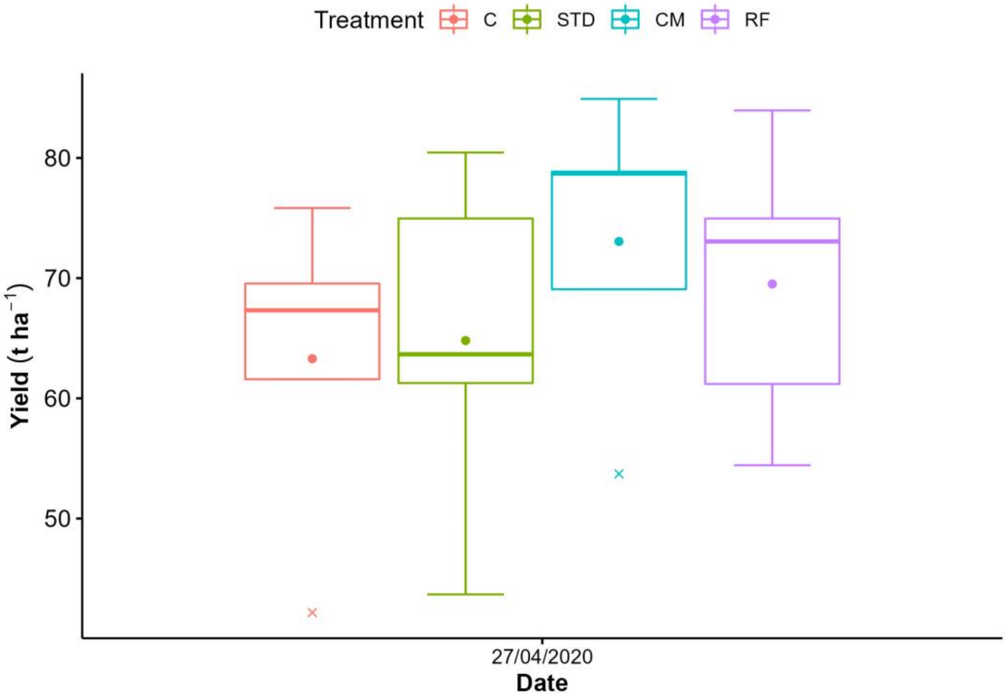

Figure 3.26. Beetroot fresh tuber yield means, medians, quartiles, maximums, and minimums (t ha^{-1}) for each treatment collected from the lysimeters (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertiliser programme).

For the Pak choi season, there were statistically significant differences between the C and EXC treatment (p=0.032). The lowest yield was measured on the C treatment with a mean of 95.7 t ha^{-1}, while the largest yield was measured on the EXC treatment with a mean of 130 t ha^{-1} (Figure 3.27).

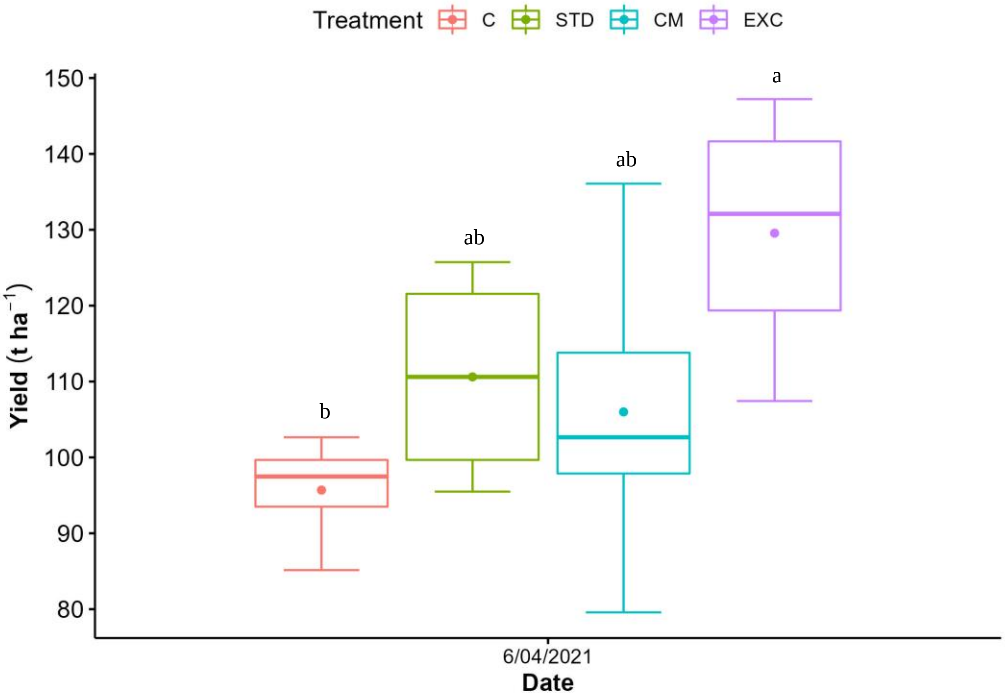

Figure 3.27. Pak choi yield means, medians, quartiles, maximums, and minimums (t ha^{-1}) for each treatment collected from the lysimeters (C for control; STD for standard practice; CM for chicken manure and EXC for excess). Different letters indicate significant differences between means of treatments ($p<0.05$, Tukey method for p-value adjustment).

3.4.2.5 Field trial results

Field soil nitrate-N

In the first year, there were no significant differences between mean soil nitrate-N concentrations of the treatments ($p=0.141$) (Figure 3.28). However, in 2021, there was a significant difference between the C and STD treatment ($p=0.01$) and there was significant difference between C and STD2 treatment at $p<0.1$ ($p=0.07$).

Differences in soil nitrate-N concentration were found between the treatments in the field during 2021, but not in the lysimeters. The most likely reason for this is that the grower applied 50 kg of extra N fertiliser to the STD on the 3rd of March, one month before Pak choi harvesting and soil sampling. This additional dose of N was not applied in the lysimeters.

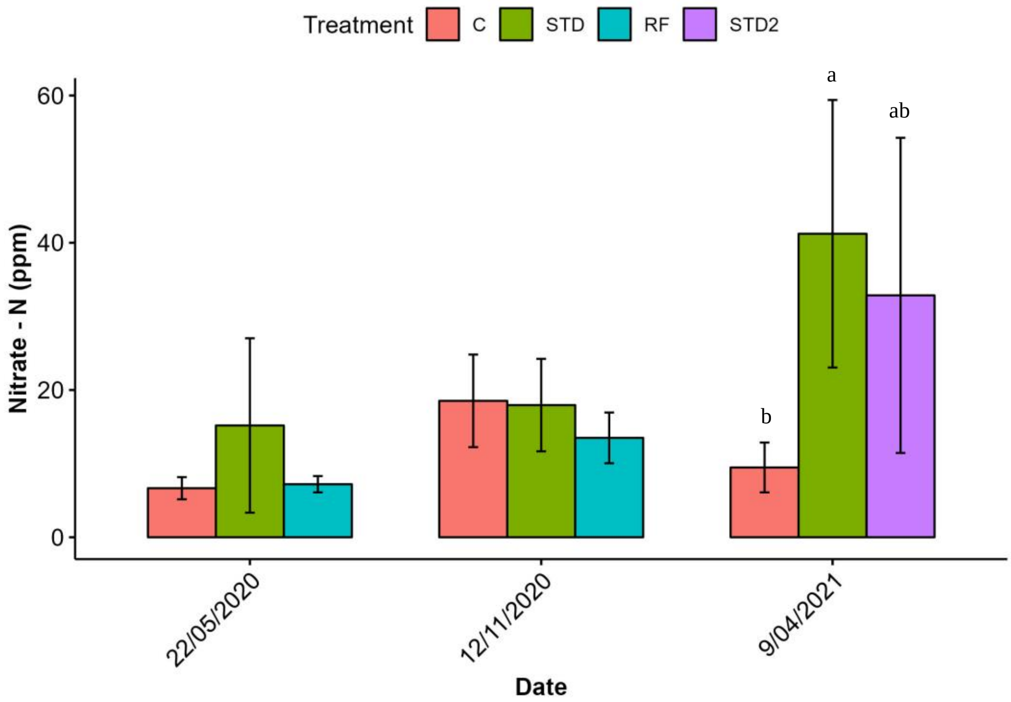

Figure 3.28. Mean nitrate-N concentrations in the 0-20 cm of soil for each treatment in the field trial (C for control; STD for standard practice; RF for reduced fertiliser programme and EXC for excess). Different letters indicate significant differences between means of treatments ($p<0.05$, Tukey method for p-value adjustment).

The ANOVA revealed no significant differences between the yields of treatments for the year 2020 ($p=0.794$) (Figure 3.29). Mean fresh tuber yields were 86.9 t ha^{-1} for the RF treatment, 99.3 t ha^{-1} for the STD treatment and 101.9 t ha^{-1} for the C treatment.

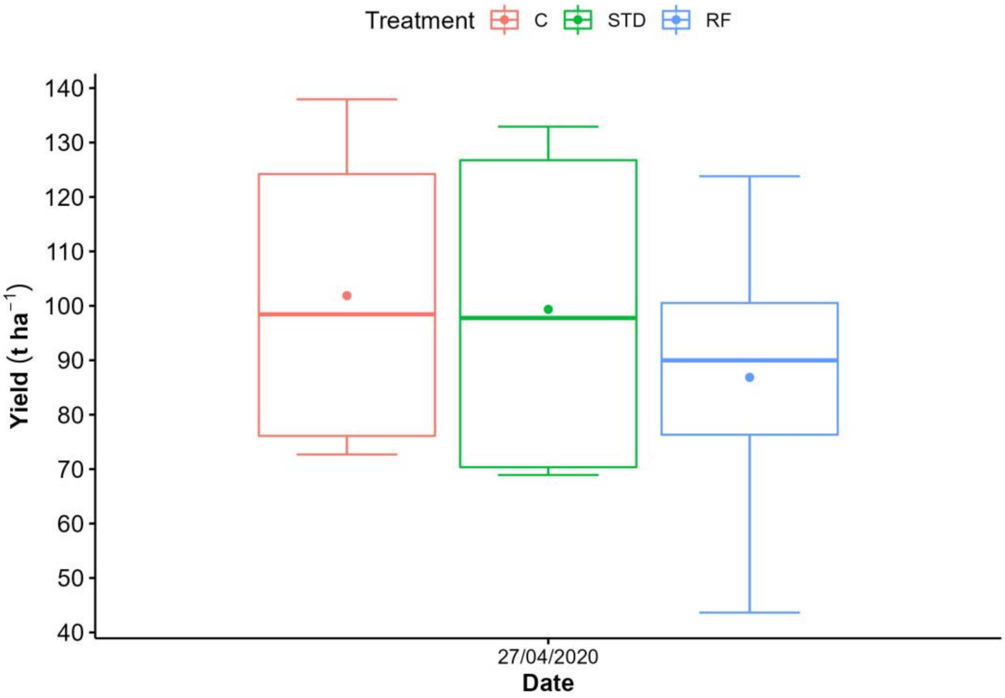

Figure 3.29. Beetroot tuber yield means, medians, quartiles, maximums, and minimums (t ha^{-1}) for each treatment collected from the field in the first growing season (C for control treatment; STD for standard practice treatment and RF for reduced fertiliser programme).

Piezometers

The highest level of shallow groundwater in the piezometers was measured during the summer season (Figure 3.30) following large rainfall events. The depths of groundwater level observed in the two different shallow piezometers (1 m depth) were similar during the sampling period.

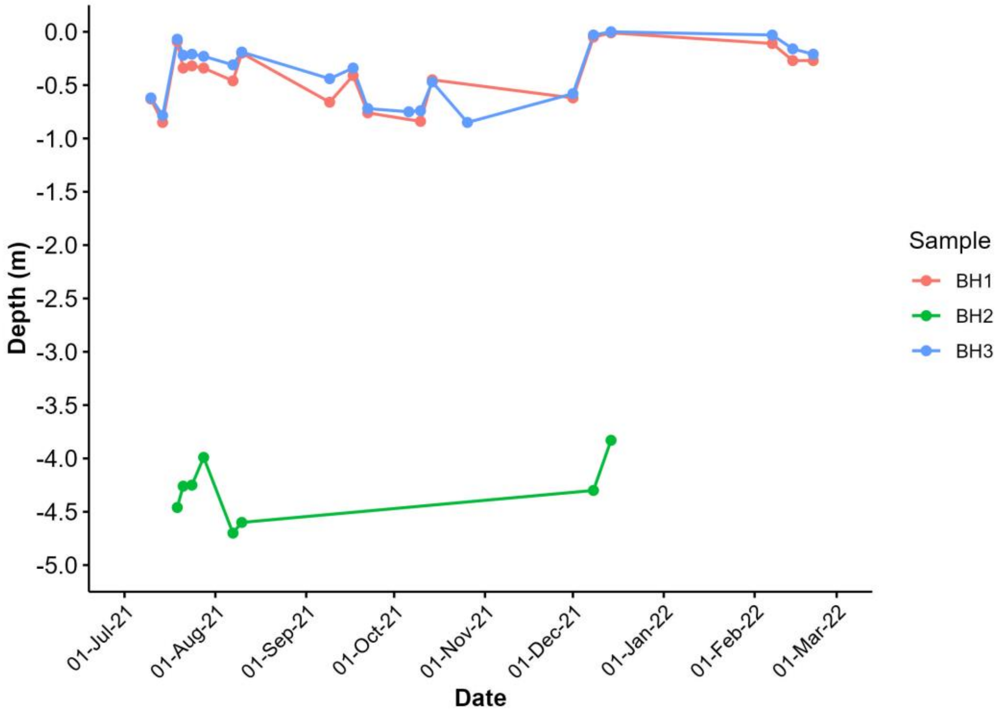

Figure 3.30. Depth of water table (m) of piezometers from ground level (BH1 for piezometers 1 (1 m depth), BH2 for piezometer 2 (5 m depth) and BH3 for piezometer 3 (1 m depth)).

BH1 had considerably higher nitrate-N concentrations than the other piezometers (Figure 3.31). Overall, nitrate-N concentrations of piezometers tended to be higher in July 2021, although nitrate-N concentrations in BH1 were greatest in December 2021. Mean nitrate-N concentrations in the piezometers were 31.6, 22.0 and 10 m L^{-1} in BH1, BH2 and BH3, respectively. Shallow piezometers (1 m depth) had an overall mean nitrate-N concentration of 20.8 mg L^{-1} between July 2021 and March 2022.

Nitrate-N concentrations in both shallow piezometers (1 m depth) had a mean of 21.4 mg L^{-1} between July and October 2021, which was similar to the mean nitrate-N concentration of drainage water in the STD treatment in the lysimeters (19.6 mg L^{-1}) for the same period.

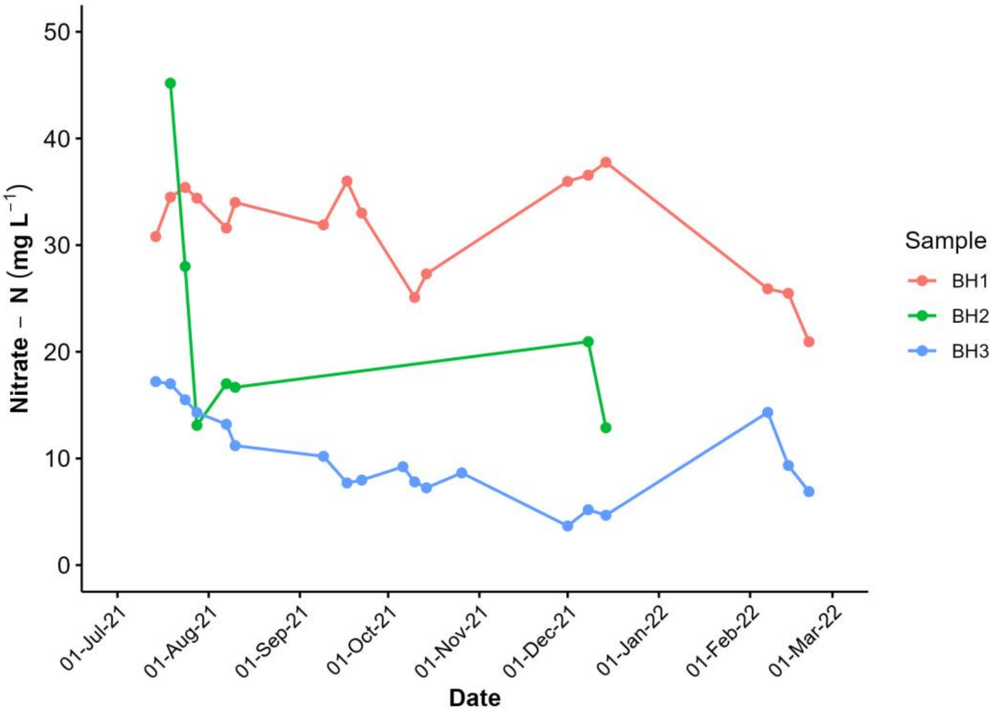

Figure 3.31. Nitrate-N concentrations in the piezometers (BH1 for piezometers 1 (1 m depth), BH2 for piezometer 2 (5 m depth) and BH3 for piezometer 3 (1 m depth)).

3.5 Discussion

3.5.1 Soil moisture

There were no differences in soil moisture content across different treatments at both sites and in either year. Soil moisture contents had small within-treatment standard deviations in the first soil layers (0-30 cm), while variability was larger in deeper layers (30-60 cm). These results agree with field observations, where some sampling points were noticeably wetter than others at the time of sampling. Similar findings were reported by Foroud et al. (1993) who, in their study of potatoes in Canada during years 1988 and 1989, found high variability in soil moisture content at 45 cm depth.

A number of factors could have been involved in the variability of subsoil (30-60 cm) moisture content measurements. The soil depth to the compacted layer and also to the parent material (gravels) varied across the sampling points. The use of traffic interrows could also have contributed to the variability as the greater soil density of these rows could lead to surface runoff into lower areas of the rows and/or water movement back into cultivation beds. All these factors can affect the subsoil environment and hydraulic properties.

There are other reports in the literature of large variability in soil moisture content both in space and time (Brocca et al., 2007; Li et al., 2019; Mälicke et al., 2020; Peterson et al., 2019; Rolston et al., 1991; Starr & Timlin, 2004; Zehe et al., 2010). According to several studies (Brocca et al., 2007; Del Toro-Guerrero et al., 2018; Entin et al., 2000; Guo et al., 2020; Mälicke et al., 2020; Starr & Timlin, 2004), changes in soil moisture content at the small-scale are primarily driven by variability in vegetation cover, topography, soil properties (such as organic carbon content, clay content, field capacity and soil depth) and previous moisture content, all of which can interact. Authors including Starr & Timlin (2004), suggest that, besides influencing evapotranspiration, the canopy of plants can intercept water from the rainfall and disperse it around, potentially to the interrow.

The variability of soil moisture in the subsoil was addressed by revising the sampling strategy (Figure 3.4). As a result, the mean standard deviation within each treatment's sampling date was reduced by 11% and helped obtaining more accurate values.

3.5.2 Soil nitrate-N and crop yields

Generally, soil nitrate-N concentrations were greater under the green vegetables crop rotation than potatoes (Figures 3.11, 3.14 and 3.19). Greater pre-planting soil nitrate-N concentrations found under green vegetables can be largely explained by the mineralisation of N from residues of the previous crop. Additionally, larger mean soil nitrate-N concentrations found in the lysimeters than in the field were associated to greater mean yields in the field, and therefore, greater N uptake by these plants (assuming similar N concentrations in plants).

Management by growers, and subsequently measurements in this study, were considerably impacted by the COVID-19 pandemic. For instance, the beetroot crop in the Green vegetables field was not harvested early enough by the grower, mainly due to low customer demand and logistic problems. Therefore, the beetroot yields at both the field and the lysimeters were larger than yields usually obtained by the grower. These above-normal yields were not harvested but were incorporated in the soil, which possibly resulted in large soil nitrate-N concentrations later in the season due to mineralisation. This could explain why greater nitrate-N concentrations in the soil and water were found in the first year (Figures 3.19 and 3.21) than in the second year (Figures 3.19 and 3.24) at the Green vegetables site.

Rainfall intensity was found to greatly influence soil nitrate-N concentrations, soil water nitrate-N concentration and crop yields. For instance, potato yields in the second growing season (Figure 3.17) were smaller than in the first growing season (Figure 3.13) despite very similar agronomic management. The grower attributed this difference to the intensive rains that occurred shortly after planting in November 2020, with seven days of continuous rainfall including one day with 33 mm. This could also explain why subsoil nitrate-N concentrations were larger in the second growing season (Figure 3.15, Table 3.20) than in the first season (Figure 3.12, Table 3.14), as greater rainfall amounts could transport nitrate-N to deeper soil layers. Similarly, in the Green vegetables site, larger concentrations of nitrate-N in the first year of monitoring (January 2020 to January 2021) were found in the STD treatment. Probably the second application of N in the STD treatment (February 2020), which was also the largest, was not completely absorbed by the beetroot crop and it was transported to lower layers of the soil by the drainage associated with the large rainfalls on days five and seven after application (effective root depth = 0.3-0.6 m (Haberle et al., 2018; Irrigation New Zealand Inc, 2007; Starke Ayres, 2014)). The effects of heavy rainfall on vegetable crops are not well understood, but studies indicate that waterlogging is likely to have negative impacts on root function, growth, and development, shoot growth and development, and crop yields (Evans & Fausey, 1999; Satchithanantham, 2013).

Differences in, amongst other factors, crops, fertiliser rates and management practices make it difficult to directly compare the current study and similar research (i.e., under vegetables) in New Zealand. However, Table 3.32 shows the soil nitrate-N results of this research along with values from other New Zealand studies under potatoes. The variable nature of the results reported by other workers is noteworthy. For example, the topsoil nitrate-N amounts found by Martin et al. (2001) are almost three times greater than those reported by Francis et al. (2003) despite similar fertiliser rates. This variability notwithstanding, the values from the current study, with its smaller fertiliser rates, agree reasonably well with the values recorded in other studies.

Table 3.32. Comparison of means of soil nitrate-N at/near potato harvesting with other studies (STD for standard practice; GP for good practice and EXC for excess practice).

Reference	Fertiliser (kg N ha^{-1} growing $season^{-1}$)	Topsoil (0-30 cm) nitrate-N means (kg ha^{-1})	Subsoil (30-60) nitrate-N means (kg ha^{-1})
STD first season	180	52	35
STD second season	180	26	33
GP second season	220	35	35
EXC second season	300	74	45
Martin et al. (2001)	242	67	41
Martin et al. (2001)	350	123	93
Martin et al. (2001)	472	123	129
Francis et al. (2003)	481	47	70

Soil nitrate-N content was variable at both sites and years; other researchers have also reported large variability in their measurements. Fraser et al. (2013), in their study of arable crops (wheat, barley and peas) under intensive tillage in the Canterbury region, reported an overall mean of 125 kg N ha^{-1} soil nitrate-N with a standard deviation of 103 kg N ha^{-1} in the 0-60 cm soil depth (back-calculated from confidence intervals and assuming t distribution, Higgins et al. (2022)). In Canada, Cambouris et al. (2008) reported an overall mean of 99 kg of residual nitrate-N ha^{-1} and a standard deviation of 28 kg soil nitrate-N ha $^{-1}$ in the 0-70 cm soil depth under potatoes receiving 150-180 kg N ha^{-1} fertiliser. These values are comparable to the variability found in the present study. For example, the STD treatment had a mean of 78 kg soil nitrate-N ha^{-1} and a standard deviation of 44 kg nitrate-N ha^{-1} in the first season, and a mean of 61 kg soil nitrate-N ha^{-1} and a standard deviation of 22 kg nitrate-N ha^{-1} in the second season.

The use of controlled release fertiliser as an alternative practice was successful in decreasing soil nitrate-N concentrations in the soil profile, particularly in the topsoil (45% of reduction) (Figure 3.11, Table 3.13). However, potato yields from this treatment were smaller than those under current practice (Figure 3.13), and therefore, it is unlikely to be adopted as a mitigation measure by growers in this area. As expected, topsoil nitrate-N concentrations in the CR treatment were constant over time when compared to other treatments, which could confirm that nitrate-N is made available steadily. Zotarelli et al. (2021) have characterised the N demand curve for potato plants, which indicates that the peak demand occurs approximately 40 days after planting, just as tuber initiation commences (Figure 3.32). Given this, the CR practice might not have supplied sufficient N at this time to meet this sharp increase in demand. Reid & Morton (2019) and Wilson et al. (2010) suggest that a complementary application of soluble N fertiliser could address this shortfall in N supply at this critical time. Thus, there could be some scope to utilise this practice as an in-field mitigation measure to decrease N losses without compromising crop yields, but more studies of any such practice are needed in New Zealand.

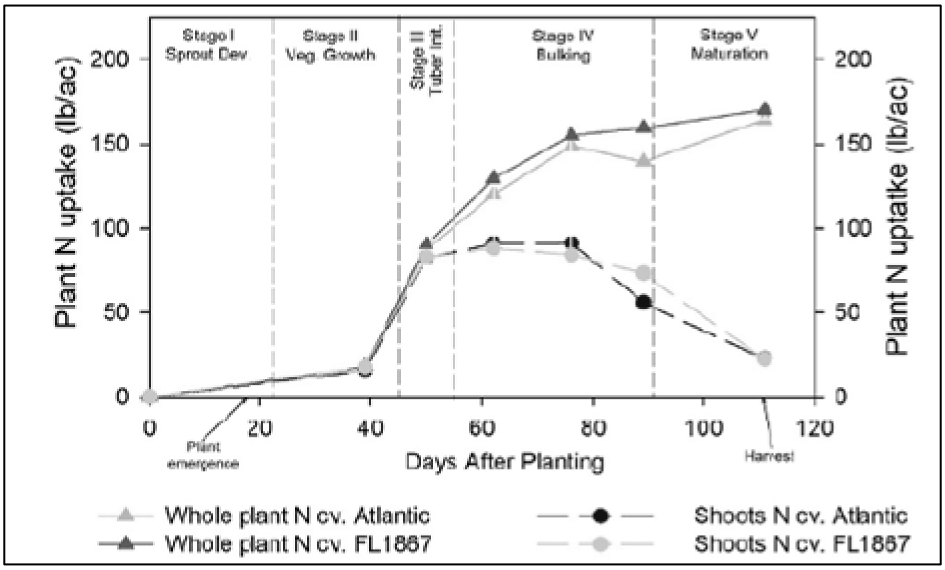

Figure 3.32. N demand from potato crops (Zotarelli et al., 2021)

In general, other studies suggest that vegetable crops receiving CR fertilisers have similar yields to those with conventional fertilisers, but results for soil nitrate-N are less consistent. In their study of CR fertilisers applied to potatoes in Pukekohe, Martin et al. (2001) indicated that when compared with soluble ammonium sulphate nitrate (ASN), ASN coated with DMPP (a N release inhibitor) decreased N leaching – implying less soil nitrate-N – without reducing yields significantly. However, they also indicate that more N is left in the soil after harvesting (Martin et al., 2001). Studies in North America by Wilson et al. (2010) and Xing et al. (2016) reported similar yields and significantly lower leaching losses from potatoes under polymer coated urea (PCU) – a similar fertiliser to the one used in the present study – than under conventional fertiliser, particularly when high amounts of N were applied. Wilson et al. (2010) did not find any significant differences in postharvest soil N between both treatments. Zebarth et al. (2012) and Clément et al. (2021) obtained similar yields for potatoes receiving the same PCU fertiliser and a conventional fertiliser, but soil nitrate-N concentrations were higher under the PCU treatment. Therefore, CR fertilisers should be used with caution as there is a chance of increasing soil nitrate-N at certain times, particularly after harvesting.

The use of split liquid fertiliser did not reduce soil nitrate-N under vegetable crops and did not increase crop yields (Figure 3.33). As for the CR treatment, the response in soil nitrate-N concentrations to liquid fertiliser was delayed, as it was applied later in the season (after emergence). This could broadly explain why this treatment was one of the least successful. Studies from Rens et al. (2018), Sparrow & Chapman (2003) and Zotarelli et al. (2021) highlight the importance of fertilising potatoes in the initial stages (before and at emergence) to maximise tuber yields. In addition, the larger nitrate-N concentrations found in deeper layers under this practice might suggest that more N is prone to leaching and less available to plants.

Figure 3.33. Photographic register of the difference between split liquid fertiliser treatment (centre, inside the poles) and standard practice treatment (sides, outside the poles) on the day 37 after planting.

The effect of increased N fertiliser on soil nitrate-N was variable depending on the site. While at the Potatoes site, soil nitrate-N concentrations increased accordingly to increasing N fertiliser, it was not the case for the Green vegetables site. This could be related to an overall relatively small increase in N fertiliser compared to the normal practice, that could have been removed by a somewhat greater crop N uptake (assuming similar crop N concentrations between treatments).

The beetroot crop at the Green vegetables site was found to be unresponsive to N fertiliser (Figures 3.26 and 3.29). This might be related to elevated residual soil nitrate-N concentrations in the profile. In their study, Bilbao et al. (2004) suggested that, in soils with little leaching and no water stress, pre-plant soil nitrate-N concentrations above 40 mg kg^{-1} in the first 30 cm (approximately 132 kg ha^{-1} for a bulk density of 1.1 g cm^{-3}) would lead to no response to N fertiliser in sugar beet yields.

Generally, excess N application did not produce significantly larger yields than those associated with the regular N rates used by growers (Figures 3.17 and 3.27). Similar results were obtained by Craighead & Martin (2003) for process potatoes in Canterbury. They suggested that there is some increase in process potato yields with increasing N fertiliser above 150 kg N ha^{-1}, but this increase is marginal, with an extra 2 t ha^{-1} yield for every 30 to 50 kg N ha^{-1} applied, and in some process varieties like Kennebec, there is no such increase in yield (Craighead & Martin, 2003). Furthermore, these authors found that N fertiliser applications increased potato yields to an optimum but also reduced tuber dry matter (Craighead & Martin, 2003). Similarly, Reid et al. (2020) observed almost no yield improvement for beetroot receiving between 320 and 480 kg N ha^{-1} of applied N. Other authors agree with the marginal yield response to larger N applications (Martin et al., 2001; McPharlin & Lancaster, 2010; Sparrow & Chapman, 2003). However, they also highlight that this response changes with soil texture (i.e., potato yields decrease if grown in loamy soils and/or reach a plateau in sandy soils when applying large amounts of N fertiliser) (McPharlin & Lancaster, 2010) and that yields can often depend more on the field history rather than the fertiliser programme (Sparrow & Chapman, 2003). Therefore, there is no generic recommended optimum N fertiliser rate.

3.5.3 Drainage

Most of the drainage monitored in this study occurred between winter and early spring, when rainfall was more frequent, and evapotranspiration was smaller. However, some intense rainfall and drainage events also occurred in summer. Cumulative drainage values recorded in the present study were slightly larger than values reported by others' research for the area. Norris et al. (2017) reported approximately 350 mm of mean cumulative annual drainage for a rainfall of approximately 1050 mm. The pattern of rainfall and drainage identified by these authors (Norris et al., 2017) reflects the rainfall and drainage pattern described here.

The variability in cumulative drainage from lysimeters is noteworthy, however, studies in New Zealand have reported somewhat similar variability (Table 3.33). Other authors, such as Gunaratnam (2021), Knappe et al. (2002) and Rodríguez Gelós (2020) have also reported high variability in measured drainage values.

Table 3.33. Comparison of cumulative rainfall and drainage and variability measured using lysimeters in New Zealand studies.

Reference	Period	Location	Soil	Crop	Cumulative rainfall (mm)	Overall cumulative drainage (mm)	Overall standard error (mm)
Avendano et al. (2022)	2020	Manawatu	Shannon silt loam	Beetroot-fallow	1056	388.5	73.7
Avendano et al. (2022)	2021	Manawatu	Shannon silt loam	Pak choi	1270	530.7	132.2
Herath et al. (2014)	2011-2012	Manawatu	Manawatu fine sandy loam	Potato	1014	263.8	84
Duncan et al. (2016)	2010-2013	Canterbury	Hororata stony	Pasture	2562	1015	103.2
Duncan et al. (2016)	2011-2013	Canterbury	Eyre-Paparua stony	Pasture	1170	725	24.6
Duncan et al. (2016)	2010-2013	Canterbury	Templeton silt loam	Pasture	1711	456	29.8
Giltrap et al. (2014)*	2009	Waikato	Horotiu silt loam	Pasture	1100	520	40

* Assuming that reported values are standard error of the mean

There are a number of factors that could have influenced within-treatment drainage variability in this study, but most likely, the small differences in soil properties, including structure, and vegetation cover led to variations in drainage that became larger as they accumulated over time. Furthermore, some of the variability in drainage could be associated with the repacking of the lysimeters (Corwin, 2000; Saporito et al., 2016). Cameron et al. (1996) and Cassel et al. (1974) also explain that the repacking of soil in lysimeters can affect pore size distribution and hydraulic properties. Finally, there is also some intrinsic variability associated with the lysimeter method. For instance, studies have suggested that edge flow effect can considerably influence measurements in lysimeters, and this could occur either with or without the use of a petroleum seal (Williams et al., 2020).

3.5.4 Nitrate-N concentrations and fluxes in drainage water

Similar to nitrate-N concentrations in the soil, increasing N fertiliser rates generally produced larger nitrate-N concentrations in drainage water (Figure 3.16 and Table 3.21). As discussed previously, exceptions to this rule were found in the Green vegetables site where increases in nitrate-N concentrations in drainage with increasing N fertiliser rates were small, possibly due to the removal of N by crop uptake.

As noticed in previous studies (Di & Cameron, 2002; Fraser et al., 2013), high nitrate-N concentrations and N leaching losses occurred under green vegetable crop cultivation particularly during the fallow period, and so care should be taken to minimise the potential leaching associated with these periods.

Large nitrate-N concentrations were found in piezometers installed in the green vegetable crop field (1 m and 5 m depth). These results support the high nitrate-N concentrations found in the drainage water in lysimeters, as mean nitrate-N concentrations between both areas were similar (mean difference of 1.76 mg L^{-1} between July and October 2021). The difference in nitrate-N concentrations between the two piezometers installed at 1 m depth could have been associated with a dilution effect, as runoff and ponding tended to occur near piezometer BH3 (Figure 3.9). This runoff, ponding and dilution occurred because the paddock slopes towards BH3, and runoff water normally has low nitrate-N concentrations (Burkitt, 2014). In addition, constant saturation conditions could have promoted denitrification process in this piezometer.

N leaching losses from the control treatments (no N fertiliser programme) were, relative to corresponding treatments, large. This is likely due to an accumulation of N in the soil profile as vegetable systems tend to have shallow roots and absorb N only in the surface horizons of soil. Martin et al. (2001) and Francis et al. (2003), who also used ceramic cups at 60 cm under potatoes, obtained similar mean nitrate-N concentrations in drainage to those reported here for their control treatment.

The use of chicken manure produced similar nitrate-N leaching results to the current practice under green vegetable crops (Figures 3.22 and 3.25, Tables 3.28 and 3.31). Therefore, organic amendments might not reduce N leaching losses from vegetable systems. Further research with a more detailed monitoring of drainage volumes should be conducted.

This study confirms that high nitrate-N concentrations in drainage can be expected under vegetable crop systems in New Zealand (Clothier et al., 2007; Horizons Regional Council, 2012; Norris et al., 2017);), with a mean concentration of 33.8 mg nitrate-N L^{-1} under the potato crop, 16.3 mg nitrate-N L^{-1} under onion crop, and an overall mean concentration of 51 mg nitrate-N L^{-1} under vegetable crops (resulting in an overall mean loss of 201 kg nitrate-N ha^{-1} yr^{-1}). By using a water balance model to estimate drainage (Chapter 4) and the nitrate-N concentrations in drainage water from suction cups, nitrate-N leaching losses under the potato crop were calculated as 65 kg ha^{-1} for the monitoring period (May to August 2021) and 30 kg ha^{-1} under the onion crop (October 2021 to February 2022). Nitrate-N concentrations and nitrate-N leaching losses in drainage water were generally much larger in soils cultivated with green vegetables, where residual nitrate-N concentrations in the soil profile were also larger than those cultivated with a potato-onion rotation. This suggests that residual nitrate-N contents should be used to inform fertiliser and crop management programs if large nitrate-N leaching losses in vegetable systems are to be avoided. Since 2019, growers of the area started to adopt new fertiliser recommendations (Reid and Morton, 2019) and have begun using Nitrate Quick Test sampling before the application of fertiliser (Bloomer et al., 2020).

Calculated nitrate-N leaching losses for the potato crop of this study were smaller than those reported by Martin et al. (2001) (approximately 160 kg N ha^{-1}), and by Francis et al. (2003) (164 kg N ha^{-1}), possibly due to the large difference in N fertiliser applications. However, values reported here are similar to those reported by Williams and Tregurtha (2003) for an Oamaru site (65-94 kg N ha^{-1}), and to those reported by Herath et al. (2014) (67.7 kg N ha^{-1}) for the Manawatu area. In the onion crop growing season, calculated nitrate-N leaching losses reported here were greater than those reported by Thompson et al. (2022) (8.4 kg N ha^{-1}). Generally, nitrate-N concentrations and N leaching amounts for vegetable crops reported here were similar to those reported by Norris et al. (2017) (20-225 kg N ha^{-1}) for similar crops at a nearby location, but considerably larger than those reported by Francis et al. (2003) (134 kg N ha^{-1}). Variability in nitrate-N leaching rates, though seemingly high, was also within the normal range, with similar values for the standard error to those reported by Fraser et al. (1994), Cameron et al. (2002) and Di and Cameron (2002).

Outside of New Zealand, studies have often measured leaching losses over a range of depths. In Canada and the US, studies have measured nitrate-N concentrations under potatoes using suction cups at 90 cm (Clément et al., 2020) and 120 cm (Wilson et al., 2010), reporting lower concentrations (18.7 to 24.8 g nitrate-N m^{-3}, and 1.7 to 14.1, respectively) than the values measured in the present study. Clément et al. (2020)

explained that variability in nitrate-N concentrations in drainage over years is highly associated with the distribution of seasonal rainfall and irrigation timing.

3.6. Conclusion

Large nitrate-N concentrations in soil and leaching losses under vegetable crops were found in the Horowhenua. Green vegetable crop rotations had larger nitrate-N concentrations in drainage water than potato-onion crop rotations due to larger amounts of residual N and N mineralisation. However, there were also exceptional circumstances raised by the COVID-19 pandemic that led to increasing nutrient losses from the green vegetable rotation in part of the study period.

The use of controlled release fertiliser or liquid fertiliser did not maintain or increase potato yields compared to regular practices, probably because of a deficit in N at crop planting and emergence. However, controlled release fertiliser application produced lower levels of nitrate-N in the soil, and a small complementary dose of conventional N fertiliser at the start could produce comparable yields, with lower risk of N leaching losses.

N fertiliser applications larger than the current practice of the growers did improve crop yields, but produced significantly larger nitrate-N concentrations in drainage water. Therefore, optimum N fertiliser programmes that aim towards reaching yields before this plateau in yield are encouraged in order to reduce the risk of N leaching.

Experimental observations from this chapter are key to the calibration and validation of field-scale models such as APSIM Next Generation to further quantify nitrate-N losses under different cropping and fertiliser scenarios in the study area.

Chapter 4: APSIM Next Generation: model calibration, validation and use to identify in-field practices to reduce N leaching under intensive vegetable production systems

4.1 Abstract

Research has shown that vegetable crop systems contribute substantially to the deterioration of water quality. A quantification of nitrate-N losses from intensive vegetable production systems is key to develop and help implement targeted and effective measures to reduce its potential impacts on quality of receiving waters. However, due to high costs, and spatial and temporal variability, field experiments are practically limited to effectively measure nitrate-N losses from intensive vegetable production systems. In this regard, simulation models can be helpful to provide an idea of how much and when are nutrients being lost to the environment. In this study, APSIM Next Generation model was parameterised and validated using detailed field observations of soil water and nutrient dynamics in two vegetable crop fields of the Manawatu-Whanganui Region. The modelling efficiency (*NSE*) was used to assess the goodness-of-fit between both measured and predicted values of soil moisture content, drainage depth, soil nitrate-N concentration, soil water nitrate-N concentration and nitrate-N leaching. *NSE* values for soil moisture content ranged from -1.19 to 0.8, with values often >0.35 even at daily time-step, which indicated an overall 'satisfactory' performance of the model. Similarly, APSIM had a 'satisfactory' performance in predicting cumulative drainage depths, with *NSE* values ranging from 0.31 to 0.7. Observations of nitrate-N concentration in soil and water were highly variably and very challenging to predict in APSIM. *NSE* values for nitrate-N concentration in soil and water ranged between -6.85 and 0.84, and between -7.33 and -0.10, respectively. However, APSIM estimations were reasonable in

most cases, following a realistic trend in time, and responding appropriately to N inputs and the weather or management actions. Cumulative nitrate-N leaching estimations from APSIM had *NSE* values between 0.15 and 0.85, indicating a very 'satisfactory' performance. The calibrated and validated APSIM was then applied to assess potential effects of a suite of in-field mitigation measurements on nitrate-N leaching losses in the study area. These scenarios included: a) change in soil type, b) potato harvest at maturity followed by a ryegrass crop during the drainage season, c) potato harvest at maturity followed by a winter catch crop, d) potential of drainage management, e) other alternative cover crops and catch crops instead of ryegrass, f) "in-season" versus "out-of-season" production, and g) alternative crop rotations. Nitrate-N leaching reductions from these scenarios ranged from 6 to 52% compared to the normal practice of growers of the area.

4.2 Introduction

Agriculture is a crucial sector of the economy including the creation of employment in New Zealand and the Horowhenua Region. This industry has undergone sustained intensification over the past few decades. Outdoor vegetable farming, in particular, has increased the use of fertiliser and irrigation to maximise crop yields (Gibbs, 2011; Horizons Regional Council, 2017). The Horowhenua Region has been especially impacted by this intensification, and Lake Horowhenua is now one of the most trophic lakes in New Zealand (Land Air Water Aotearoa, 2022). This situation has led to increasing concern in the community and a desire amongst many to regulate the maximum nitrogen leaching allowed from agricultural lands (Horizons Regional Council, 2021). Consequently, implementing in-field mitigation strategies that can effectively reduce nutrient losses has become increasingly important to policy-makers, researchers, and growers (Brown et al., 2019). However, monitoring implementation and effectiveness of such strategies has been difficult; due to their high costs, complexity and large variability, direct measurements of nutrient losses from individual farms are not a practical option. Fortunately, a combination of both, field experiments (Chapter 3) and appropriately calibrated simulation models, can help to identify and assess the effectiveness of the, sometimes unique, mitigation strategies required on farms (Cichota et al., 2010a; Delgado et al., 2008; Langeveld et al., 2007; Monaghan et al., 2007; Veihe et al., 2006; Vogeler et al., 2013).

The APSIM (and APSIM Next Generation) model is a worldwide recognised framework to simulate agricultural systems. It is constituted by a suite of modules that interact with each other to represent complex processes and interaction in the soil, plant, the environment, and management (Cichota et al., 2010b; Holzworth et al., 2010). The use of the APSIM model has been extensive internationally, including a range of climates, soils and cropping systems (Asseng et al., 1998; Brilli et al., 2017; Holzworth et al., 2014; Huth et al., 2009; Borus et al., 2016; Cheeroo-Nayamuth et al., 2000; Gaydon et al., 2017). Several studies have demonstrated the ability of APSIM to successfully represent N leaching losses from pastoral systems in New Zealand (Cichota et al., 2010a, Cichota et al., 2010b; Giltrap et al., 2014; Cichota et al., 2012; Trolove et al., 2019b). However, APSIM's suitability to describe soil water and nutrient cycling under intensive vegetable farming requires further calibration and validation, especially in the Horowhenua Region.

The aims of this chapter are: 1) parameterise the APSIM Next Generation model to reflect the crop, soil and environmental conditions of the Horowhenua region of New Zealand, 2) evaluate the ability of APSIM to predict crop growth, soil moisture content, and drainage, 3) assess the ability of APSIM to predict nitrate-N leaching losses under intensive vegetable farms in the Horowhenua Region, 4) identify the likely values of nitrate-N leaching in a range of climate and drainage regimes over the long-term period, and 5) to evaluate a suite of in-field management practices to potentially reduce N-leaching losses using APSIM.

4.3 Materials and Methods

4.3.1. Field trials

The measured data to calibrate and validate the APSIM Next Generation model were taken from the field trials described in Chapter 3. These comprise results from the two different vegetable crop sites in Levin, Manawatu-Whanganui over two years (2020 to 2022). Inputs for the model's calibration include meteorological data, crop management practices (dates for actions such as sowing or fertiliser applications, crop types, fertilisation, ploughing, irrigation, etc.) and soil parameters. A brief description of the trial and measurements done are presented below. More specific details about these measurements and practices can be found in Chapter 3.

4.3.1.1 Potatoes site

The first trial, in brief, was conducted between December 2019 and February 2022 on a farm with a soil classified as a moderately well drained Waitohu silt loam. Potatoes (*Solanum tuberosum* L. cv 'Nadine') were grown in two seasons, from December 2019 to May 2020 and from November 2020 to May 2021, and onions (*Allium cepa* L.) were grown from September 2021 to February 2022. For the potato and onion seasons, there were four nitrogen fertiliser treatments with four replicate plots of each treatment. The plots were 16 m long and 3.5 m wide (4 rows). The intervening onion crop was sown across all plots and all the plots received the same type and amount of fertiliser, but only the (former) 'Standard' and 'Control' plots were monitored. Full details of the trial and fertiliser treatments are given in Tables 3.3, 3.4 and 3.5 (Chapter 3).

Soil moisture was monitored *in situ* every 30 minutes by two probes, that were 90 cm in length, using frequency domain reflectometry (FDR) (Sentek "drill and drop" soil probes, series: BT001141 and BT001144) (Sentek, 2022). The probes were calibrated with volumetric soil moisture measurements calculated from oven-dried soil samples taken between 2020 and 2021 (Chapter 3). These probes were placed in two locations under the 'Standard' treatment in every season.

Ammonium-N, nitrate-N, organic matter, organic carbon and C/N ratio were measured for the topsoil (0-30 cm) and subsoil (30-60 cm) every month. During the end of the second (June 2021 to August 2021) and over the third crop season (October 2021 to February 2022), nitrate-N concentrations from drainage water were measured using porous suction cups at 60 cm depth (Briggs & McCall, 1904; Raij-Hoffman et al., 2020).

To compare with nitrate-N leaching simulated by APSIM, observed cumulative nitrate-N leaching rates in the Potatoes site were calculated as the product of the mean nitrate-N concentration in each treatment and the drainage volume predicted by APSIM (Fraser et al., 2013).

Fresh and dry matter potato yields were measured when the crops were deemed ready for harvesting by the farmer. The potatoes were not actually harvested at this time but were ground-stored and harvested to meet market demand. Onion yield was not measured.

4.3.1.2 Green vegetables site

The second trial site is also described in detail in Chapter 3. In brief, the trial took place between January 2020 and November 2021 in 20 lysimeters filled with imperfectly drained Shannon silt loam. During the first year, beetroot (*Beta vulgaris* L. ssp. *vulgaris* var. *vulgaris*) was sown in January 2020 and incorporated in April 2020, followed by a fallow period for the rest of that year. Pak choi (*Brassica rapa* subsp. *chinensis*) was then sown in January 2021, reincorporated, and resown in February (as uniformity was not achieved to the farmer's standard), and harvested in April 2021. Finally, perennial ryegrass (*Lolium perenne* L.) was grown from June 2021 until the end of the experiment (November 2021).

Four treatments with five replicates were established in lysimeters (20 cm diameter and 60 cm length) in the farmer's field, with different N fertiliser treatments for the beetroot and Pak choi season. Details of the fertiliser treatments are given in Tables 3.7 and 3.8 in Chapter 3.

Additionally, because the lysimeters were repacked, the soil physical parameters were considerably different compared to the surrounding field (due to soil disturbance caused by refilling), two replicates of 40 m length and 1 m wide beds were used in the field to measure soil moisture and nitrate-N concentrations in the topsoil (0-20 cm). These measurements, both in the field and the lysimeters, were used to compare simulated N losses in APSIM Next Generation.

Soil samples were taken at 20 cm depth to measure soil moisture, nitrate-N and ammonium-N, both in the lysimeters and in the field. Samples were taken before sowing and after harvesting of the crop.

Drainage water was collected after major rain events from each lysimeter, and water volume, nitrate-N, and ammonium-N were analysed.

Crop fresh and dry matter yields were measured for the lysimeters when the crops were deemed ready for harvesting by the farmer. Ryegrass above-ground biomass was estimated by cutting and weighing fresh matter.

4.3.2. Modelling

4.3.2.1 Basic setup

APSIM Next Generation model (version 2022.1.7029) was used to simulate crop performance and N leaching losses at the two field experiments (Holzworth et al., 2018). This model can be downloaded freely from www.apsim.info, where the documentation for the various modules as well as an extensive list of publications is available. APSIM is a modular framework, meaning that a variety of modules can be plugged in to describe different conditions and crops. For example, the soil water balance and solute transport are described using the SoilWater module (Probert et al., 1998b). For the characterisation of the C and N cycle, the Nutrient module was used for the Potatoes site (Probert et al., 1998b), and the SoilNitrogen module was used for the Greens site (Probert et al., 1998b), and both were coupled to the SurfaceOrganicMatter module (Probert et al., 1998b) to simulate residue decomposition.

Soil data from both sites were used to parameterise the soil inputs in the model (Cichota et al., 2021; Dalgliesh et al., 2016). The main physical soil parameters were measured in the laboratory and included the bulk density (*BD*), the soil water retention at field capacity (*DUL*), wilting point (*LL15*), and saturation (*SAT*) (Tables 4.1, 4.3 and 4.5). These values, although given in Chapter 3, are presented here so the reader can make a comparison between the original values and adjusted values. Other soil physical parameters used include [near]saturated water flow factor (*SWCON*) and saturated hydraulic conductivity (*KS*), which were estimated either by the use of Pedo-Transfer Functions (Cichota, Vogeler, et al., 2013) or by the use of the APSoil database (Dalgliesh et al., 2012). APSoil is a regularly updated online database of soil characteristics specifically built for use in modelling and can be accessed freely at https://www.apsim.info/apsim-model/apsoil/. *SAT* was calculated as a fraction of porosity for each layer, following the recommendations of Cichota et al. (2021) and Dalgliesh et al. (2016). *DUL* values were adjusted slightly to better match soil moisture measurements from the field (by increasing or reducing the value to fit the moisture measured in wet soil, after equilibration). Soil chemical parameters included: the initial organic carbon content (*OC*), soil CN ratio and initial concentrations for nitrate-N and ammonium-N (Table 4.2, 4.4 and 4.6). Measured data was complemented with data from the New Zealand National Soil Database (Landcare Research, 2022; Palmer & Wilde, 2007).

Table 4.1. Soil physical parameters inputted to APSIM for simulations of the Potatoes site.

Soil layer/ Parameter	***BD* (g cm^{-3})**	***SAT* (mm mm^{-1})**	***DUL* (mm mm^{-1})**	***LL15* (mm mm^{-1})**	***SWCON* (d^{-1})**	***KS* (mm day^{-1})**
Source	Measured	Measured-estimated	Measured-estimated	Measured	Pedo-transfer functions	Pedo-transfer functions
0-10	0.87	0.60	0.33	0.19	0.598	1,500
10-20	0.93	0.59	0.33	0.23	0.590	750
20-30	1.03	0.56	0.33	0.27	0.603	450
30-40	1.11	0.54	0.36	0.28	0.552	200
40-50	1.17	0.53	0.36	0.31	0.549	100
50-60	1.17	0.53	0.36	0.31	0.495	50
60-70	1.49	0.44	0.4	0.30	0.524	18
70-80	1.49	0.44	0.4	0.30	0.513	15
80-100	1.47	0.45	0.41	0.30	0.397	15
100-110	1.51	0.43	0.39	0.26	0.494	15

Table 4.2. Soil chemical parameters inputted to APSIM for simulations of the Potatoes site.

Soil layer/ Parameter	***OC* (%)**	***Soil CN ratio* (g g^{-1})**	***Initial soil nitrate-N* (mg kg^{-1})**	***Initial soil ammonium-N* (mg kg^{-1})**
Source	Measured	Measured	Measured	Measured
0-10	3.3	10.9	5.0	2.8
10-20	3.3	10.9	5.0	2.8
20-30	3.3	10.9	5.0	2.8
30-40	2.8	10.3	4.0	2.6
40-50	2.8	10.3	4.0	2.6
50-60	2.8	10.3	4.0	2.6
60-70	1.8	13.5	1.0	2.6
70-80	1.8	13.5	1.0	2.6
80-100	1.8	13.5	1.0	2.6
100-110	1.8	13.5	1.0	2.6

Table 4.3. Soil physical parameters inputted to APSIM for simulations of the lysimeters at the Greens site.

Soil layer/ Parameter	***BD*** **(g cm^{-3})**	***SAT*** **(mm mm^{-1})**	***DUL*** **(mm mm^{-1})**	***LL15*** **(mm mm^{-1})**	***SWCON*** **(d^{-1})**	***KS*** **(mm day^{-1})**
Source	Measured	Measured-estimated	Measured-estimated	Measured	APSoil	APSoil
0-10	1.08	0.59	0.39	0.22	0.604	1,414
10-20	1.15	0.57	0.42	0.26	0.550	365
20-30	1.13	0.57	0.43	0.25	0.550	242
30-40	1.06	0.60	0.39	0.24	0.503	118
40-50	1.05	0.60	0.40	0.25	0.500	118
50-55	1.05	0.60	0.40	0.25	0.500	118

Table 4.4. Soil chemical parameters inputted to APSIM for simulations of the lysimeters at the Greens site.

Soil layer/ Parameter	***OC*** **(%)**	***Soil CN ratio*** **(g g^{-1})**	***Initial soil nitrate- N*** **(mg kg^{-1})**	***Initial soil ammonium-N*** **(mg kg^{-1})**
Source	Measured-estimated	Measured-estimated	Measured-estimated	Measured-estimated
0-10	3.6	10.9	30-35	4.2
10-20	3.6	10.9	30-35	4.2
20-30	3.6	10.9	30-35	4.2
30-40	3.2	10.3	30-35	4.2
40-50	3.2	10.3	30-35	4.2
50-55	3.2	10.3	30-35	4.2

Table 4.5. Soil physical parameters inputted to APSIM for simulations of the field at the Greens site.

Soil layer/ Parameter	*BD* (g cm^{-3})	*SAT* (mm mm^{-1})	*DUL* (mm mm^{-1})	*LL15* (mm mm^{-1})	*SWCON* (d^{-1})	*KS* (mm day^{-1})
Source	Measured	Measured-estimated	Measured-estimated	Measured	Pedo-transfer functions	Pedo-transfer functions
0-10	0.92	0.60	0.54	0.28	0.604	1,413
10-20	0.98	0.59	0.54	0.27	0.520	365
20-30	1.11	0.54	0.54	0.27	0.511	241
30-40	1.11	0.54	0.44	0.27	0.503	118
40-50	1.48	0.44	0.43	0.29	0.474	15
50-60	1.48	0.44	0.43	0.29	0.474	8
60-70	1.48	0.44	0.43	0.29	0.426	8
70-80	1.48	0.44	0.43	0.29	0.426	8
80-100	1.48	0.44	0.43	0.29	0.426	8
100-120	1.64	0.38	0.33	0.29	0.426	8

Table 4.6. Soil chemical parameters inputted to APSIM for simulations of the field at the Greens site.

Soil layer/ Parameter	*OC* (%)	*Soil CN ratio* (g g^{-1})	*Initial soil nitrate-N* (mg kg^{-1})	*Initial soil ammonium-N* (mg kg^{-1})
Source	Measured-estimated	Measured-estimated	Measured-estimated	Measured-estimated
0-10	3.6	10.9	40	4.2
10-20	3.6	10.9	40	4.2
20-30	3.6	10.9	40	4.2
30-40	3.2	10.3	30	4.2
40-50	3.2	10.3	30	4.2
50-60	3.2	10.3	30	4.2
60-70	1.8	13.5	26	3.0
70-80	1.8	13.5	26	3.0
80-100	1.8	13.5	26	3.0
100-120	1.8	13.5	25	3.0

The remaining parameters, such as; the daily fraction of water available to be extracted by plants from each soil layer (*KL*), the root exploration factor (*XF*), the fraction of microbial biomass pool (*FBiom*), the proportion of inert organic C (*FInert*), the amount and C/N ratio of fresh root material (*Root Wt and Root CN*) were based on recommendations in the APSIM documentation (www.apsim.info) and the literature (for instance, Romera et al., 2012; Teixeira et al., 2015, 2018; Cichota et al., 2021). The values of the parameters controlling evaporation (*U* and *ConA*) were adjusted to fit observed data, while the remaining were left unchanged (default).

APSIM contains functions that handle the transformation of various N forms, but these do not include slow-release N fertilisers. Equation 4.1, adapted from Trolove et al. (2019a), was coded into a manager script to describe the N release rate (i.e., daily addition of N) in the controlled release fertiliser treatment:

$$Y = 0.04x - 1 \qquad (4.1)$$

where, Y is the cumulative percentage of fertiliser released as mineral N, x is the cumulative growing degree days, and 0.04 is the N release rate (/°C). Once Y reached a value of 70% of the fertiliser released, the coefficient rate of release was adjusted to 0.004 (10% of the original rate) to reflect a slower release rate from the remaining fertiliser. This decrease in release rate has been described by Trolove et al. (2019a) and Shaviv et al. (2003), and it is due to a decrease in nutrient concentration inside the granule. After communications with experts (R. Cichota, personal communication), a reference release rate of 10% of the original value was adopted as a suitable release rate after the main release rate phase was completed (after 70% of the fertiliser had been released).

The APSIM crop models used for the Potatoes site were; the ‘Potato’ model with the cultivar set to Nadine (Brown et al., 2011), the ‘Simple Crop Resource Uptake Model’ (SCRUM) for onions and the AgPasture model for ryegrass (Li et al., 2011). For the Green vegetables site, SCRUM was used for beetroot, Pak choi and ryegrass (more details about the crop models can be found in their documentation and publications (www.apsim.info; Khaembah & Horrocks (2018)). Potential yields in the simulation of the Green vegetables site were set to match the yield estimations in the lysimeters and field as recommended (Brown & Zyskowski, n.d.).

Crop management practices (inputs, quantities, and timing) were obtained directly from the farmer and described in APSIM using default manager script modules. Additional management scripts were developed whenever a default option was not available; these were also used to modify certain parameters in the model (Table 4.7).

Table 4.7. Crop and residue parameters in APSIM simulations of the Green vegetables site.

Parameter	Module	Value	Units
Beets expected yield	SCRUM	14,000-15000	g m^2
Pak choi expected yield	SCRUM	600-750	g m^2
FOM decomposition temperature optimum in soil layer 1	SoilNitrogen	28	°C
FOM decomposition temperature optimum in soil layer 2	SoilNitrogen	28	°C
FOM decomposition temperature factor at zero degrees in soil layer 1	SoilNitrogen	0.05	-
FOM decomposition temperature factor at zero degrees in soil layer 2	SoilNitrogen	0.05	-
SOM mineralisation temperature optimum in soil layer 1	SoilNitrogen	28	°C
SOM mineralisation temperature optimum in soil layer 2	SoilNitrogen	28	°C
SOM mineralisation temperature factor at zero degrees in soil layer 1	SoilNitrogen	0.05	-
SOM mineralisation temperature factor at zero degrees in soil layer 2	SoilNitrogen	0.05	-

Meteorological data for the Met model parameterisation and simulations were retrieved from the National Climate Database of NIWA (https://cliflo.niwa.co.nz/). For the simulation period, daily values for rainfall (mm), solar radiation (MJ m^{-2}), and minimum and maximum temperatures (°C) were obtained from the Levin EWS station (Lat. 40°37'37.2"S, Long. 175°15'43.0"E), as this was the closest station to both sites (1-2 km away).

4.3.2.2 Goodness of fit

To gauge the goodness of fit between APSIM simulations and the observed data, the Nash-Sutcliffe efficiency coefficient (*NSE*) was estimated (Equation 4.2).

$$NSE = 1 - \frac{\sum_{i=1}^{m}(P_i - O_i)^2}{\sum_{i=1}^{m}(O_i - \bar{O})^2} \qquad (4.2)$$

where, m is the number of observed values; P_i and O_i are predicted and observed values; and $\bar{O}$ is the mean of the observed values (Liang et al., 2022). This coefficient was calculated with multiple observed values (O_i) for a specific day, and the predicted value (P_i) was compared against each observed value. A negative *NSE* value indicates that means of the observed data are a better predictor than simulated values, *NSE* equal to 0 indicates that the model prediction is as accurate as the mean of observed data, and positive *NSE* values indicate that simulated values are a better predictor than observed mean data (Nash & Sutcliffe, 1970; Vogeler & Cichota, 2018). However, due to the complexity of soil N transformations and high spatial variability of soil variables, several studies suggest considering *NSE* values < -1.0 as indicating 'unsatisfactory' performance of the model, and *NSE* values > -1.0 as indicating a 'satisfactory' performance of the model (Hu et al., 2006; Kersebaum et al., 2007; Liang et al., 2016, 2020a, 2020b, 2022; Roelens, 2018; Vogeler & Cichota, 2018; Yang et al., 2014).

Observed data, such drainage and nitrate-N concentrations in the soil and soil water, were obtained to a depth of 60 cm, and therefore, comparisons between observed and predicted data were also made at 60 cm soil depth. However, to ensure that they have left the rootzone, N leaching losses and drainage values were also simulated at 100 cm soil depth in the 'Potatoes site' section of this chapter. N losses and drainage simulations at 60 cm soil depth can be found in Figures C1, C2, C3, C4, C5 and C6, and Tables C3, C4, C5 and C6 in the Appendix C.

4.3.2.1 Scenario setup

The same modules and parameters as those previously described in the basic setup were used for long-term modelling and in-field mitigation scenarios. As only 6 years of data were available for the EWS weather station, data from Metservice's Levin AWS station (Lat. 40°37'19.2"S, Long. 175°15'25.2"E) was used for the long-term simulations (31-year time span) from 1991 to 2021. Gaps in these data were be filled with values from the closest available weather station in the area, generally Levin MAF station (Lat. 40°39'00.0"S, Long. 175°16'08.4"E).

Following parameterisation of APSIM and a comparison of the simulated and measured data for the years 2019 to 2022, the likely range of N losses in drainage under a potato-onion rotation for a wider range of climate conditions was investigated by simulating this crop rotation APSIM for a 31-year period (1991 to 2021).

Variations to the potato-onion rotation

Long-term simulations of potato-fallow-potato-fallow-onion-ryegrass-onion-ryegrass (hereafter referred to as 'potato-onion') rotation were conducted to compare a number of variations to the potato-onion rotation. These scenarios included: a) Scenario 1: change in the soil type, b) Scenario 2: potato harvest at maturity followed by a ryegrass crop during the drainage season, and c) Scenario 3: potato harvest at maturity followed by a winter catch crop. The catch crops evaluated were – ryegrass, beetroot, and fodder beet. These crops were recommended by an agronomist with longstanding experience of vegetable cropping in the Horowhenua (D. Bloomer, personal communication, September 2022)

The following common setup/assumptions were used to describe the management of the field in the long-term simulation and in the alternative scenarios:

Assumptions:

- Initial soil nitrate-N values were set to the mean residual nitrate-N of the previously calibrated simulations (4 mg kg^{-1} on average) (section 4.4.1).
- Soil nutrient status was reset every 2 years before crop sowing to avoid the excessive loss of some of the humic fraction in the model.
- For two continuous years, potato was sown on the first date between 01-November and 15-December when there was a minimum of 60 mm of available soil water.
- Crops were harvested on September for normal long-term simulation (ground stored potatoes) and between February and April for early harvest scenarios.
- In Scenario 2, ryegrass was sown after potato harvest (between 15-February and 30-April) with a minimum of 60 mm of soil extractable water, and was incorporated on 15-September. Ruminant dung (200 kg DM ha^{-1}, with a C/N ratio of 25) was added on the same date.
- In Scenario 3, beetroot or fodder beet were sown after potato harvest (between 01 and 30-May) and harvested on 15-October. Beetroot tubers were harvested, and leaves were left as residue, but fodder beet was harvested completely.
- Onions were sown between 01-September and 30-September every year and with a minimum of 60 mm of soil extractable water, and harvested when phenological maturity was reached in the model.
- Ryegrass was sown after onions that were harvested before March.
- Irrigation is infrequent on Horowhenua vegetable farms, and only occurs when soils are very dry. Therefore, irrigation in the model occurred between 01-January and 01-March whenever plant available water was equal to the Permanent Wilting Point.

<u>Green vegetable crops in the long-term</u>

As for the potatoes site, simulations were created for the Green vegetables site using typical management rules to simulate the range of nitrate-N leaching values likely to occur over a 31-year period. Even though a rotation like this would not be used by the grower every year, this exercise helps to identify the range in nitrate-N leaching losses under different climatic conditions. This long-term simulation also served as a reference to compare with alternative scenarios. These scenarios included: Scenario 1: change in the soil type, Scenario 2: potential of drainage management, Scenario 3: other alternative cover crops and catch crops instead of ryegrass, Scenario 4: "in-season" versus "out-of-season" production, and Scenario 5: alternative crop rotations.

The following common assumptions were used to describe the management of the field in the long-term simulation and in the alternative scenarios:

Assumptions:

- Soil nutrient status was reset every 2 years before crop sowing to avoid the excessive loss of some of the humic fraction in the model.
- The initial soil nitrate-N content has a very marked effect on N leaching as predicted by APSIM. Therefore, two scenarios were explored here. In the first case, the initial soil nitrate-N content was set, and then regularly re-set (every two years) at a value of 34 mg kg^{-1} as measured at the start of the lysimeter study. This analysis allows for comparison with the lysimeter results. However, as discussed in Chapter 3, an initial nitrate-N concentration of 34 mg kg^{-1} is unlikely in the long-term, therefore, in an alternative scenario, the initial soil nitrate-N values were set to a value of 8.1 mg kg^{-1} which is more typical (reference) and also the residual nitrate-N measured at the lysimeter study (section 4.4.2).
- In Scenario 2, tile drains were placed at 60 cm soil depth. Results from lysimeters and piezometers from Chapter 3 indicated large nitrate-N concentrations in the soil water, suggesting a potential case for simulating the placement of tile drains. This was mimicked in APSIM by coding a script that instructed the model to intercept a given percentage of leaching losses (70 and 90%) whenever there was drainage above 0.01 mm in the soil layer immediately above the tile drain's depth. The percentages assumed here were informed by the Hooghoudt equation

(Hooghoudt, 1940), using the subsoil hydraulic parameters previously shown, and assuming 40-m drain spacing. These values were only indicative and reflect the likely effectiveness of an artificial drainage system in a Shannon silt loam soil. It was assumed that all of the nitrate-N in the simulated tile drainage was removed by using an edge-of-field technology such as a denitrifying bioreactor.

- Crop rotations ended on every 05-November when ryegrass was sprayed with herbicide.
- After the beetroot and Pak Choi crops, ryegrass was sown whenever the available soil water was ≤79 mm in the 0-30 cm soil depth.
- Beetroots tubers were harvested, and leaves were returned to the soil as residue.

Scenario 3. Other alternative cover crops and catch crops instead of ryegrass

Although the farmer already uses ryegrass as cover crop to decrease nitrate leaching during winter, other crops were also evaluated for their ability to reduce nitrate leaching.

The normal beetroot-**ryegrass**-Pak choi-ryegrass rotation from the field was modified to a beetroot-**cover/catch crop**-Pak choi-ryegrass rotation, where the cover/catch crop was one of the following crops:

Grass:

- Ryegrass sown in May

Cereals:

- Barley
- Oats
- Rye
- Triticale
- Wheat

Brassicas and other broad-leaf plants:

- Fodder beet
- Winter broccoli
- Winter cauliflower
- Winter cabbage
- Radish for seed
- Rapeseed
- Mustard for seed
- Mustard for forage
- Plantain
- Turnip
- Lettuce
- Chicory

The crop selection was based on the available alternatives in the SCRUM model (with in-built parameters) and the season. Legumes were not selected due to their ability to fix nitrogen and potentially increase the risk of N leaching (Bernhard, 2010).

In addition to the previously described assumptions, this scenario had the following:

- The cover/catch crop sowing window started earlier than the normal rotation, from May until late July, whenever sowing conditions were favourable (extractable soil water ≤79 mm in the 0-30 cm soil depth between 01 and 30-July each year).
- The cover/catch crop was incorporated on 01-November, with no removal of plant material from the field.
- Default crop parameters of APSIM-SCRUM model were used, such as expected yield, harvest index, rooting depth, among other factors.

Scenario 4. "In-season" versus "out-of-season" production

It is often claimed that growing crops 'out of season' to meet year-round consumer demand for all types of vegetables, results in increased N leaching losses. Using Overseer, Stout (2021) compared the effect of growing crops "in-season" versus "out-of-season" on nitrate-N leaching from a Waitohu silt loam soil in the Horowhenua region. In the "in-season" scenario, crops in these rotations were grown in their favoured or natural season (i.e., when yield was likely to be greatest for smaller inputs of N fertiliser). In the "out-of-season" scenario, crops were grown in unfavoured seasons when conditions were not as conducive to their growth (Stout, 2021). After interviewing expert crop agronomists who were familiar with cropping patterns in the Horowhenua Region, Stout (2021) developed a series of representative crop rotations for farms in the area (Table 4.8). However, as Stout (2021) used Overseer, she was only able to predict leaching losses under these scenarios for an average year. Furthermore, she modelled leaching from the Waitohu silt loam which is not commonly used for green vegetables. In this study, APSIM was used to evaluate, over the long-term (1991 to 2021), the notion that "in-season" crop production reduces leaching from the soil where green vegetables are commonly grown (Shannon silt loam). To the extent that it was possible, expected crop yields and assumptions regarding management decisions and practices in APSIM were matched to those used by Stout (2021) in Overseer.

Table 4.8. Representative crop rotations compiled by Stout (2021) (*s* for crops produced in the summer season, and *w* for crops produced in the winter season).

Crop rotation	Scenario
Broccoli (*s*)-cabbage (*w*)	Out-of-season
Broccoli (*s*)-lettuce (*w*)	
Cabbage (*s*)-broccoli (*w*)	In-season
Lettuce (*s*)-broccoli (*w*)	
Cauliflower (*s*)-cabbage (*w*)	Out-of-season
Cauliflower (*s*)-lettuce (*w*)	
Cabbage (*s*)-cauliflower (*w*)	In-season
Lettuce (*s*)-cauliflower (*w*)	

Scenario 5. Alternative crop rotations

To investigate the effect of crop rotation on nitrate-N leaching, a range of intensive vegetable crop rotations and fertiliser rates typical of the Horowhenua region (Bloomer et al., 2020; Jolly, 2020) were simulated (Table 4.9). Fertiliser rates were based on the fertiliser guidelines for vegetable crops in New Zealand and other works in the area (Reid and Morton, 2019; Bloomer et al., 2020 and Stout, 2021), and can be found in Table C7 in the Appendix C.

Table 4.9. Some representative rotations from the growers of the area.

Rotation	Type	Year 1												Year 2											
		J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D
1	Brassica rotation		Broccoli				Broccoli				Broccoli				Broccoli				Broccoli				Broccoli		
2			Cauliflower				Cauliflower				Cauliflower				Broccoli				Broccoli				Broccoli		
3	Brassica -vegetable rotation		Spinach				Broccoli				Broccoli				Spinach				Broccoli				Broccoli		
4		Lettuce					Broccoli				Broccoli			Lettuce					Broccoli				Broccoli		
5		Squash					Cauliflower				Broccoli				Squash				Cauliflower				Broccoli		
6	Intense vegetable rotation		Spring onion				Lettuce				Spinach				Maize				Ryegrass				Cabbage		
7			Spinach				Beetroot				Ryegrass				Cabbage				Lettuce				Spinach		
8			Maize				Oats				Ryegrass				Lettuce				Cabbage				Spinach		
9			Pak choi				Spinach				Ryegrass				Beetroot				Ryegrass				Spring onion		

Assumptions:

- Onion-group plants were only sown 2 years after having planted onion.
- Ryegrass sowed if it is possible to enter the paddock during winter (extractable soil water ≤79 mm in the 0-30 cm soil depth).
- Ryegrass or oats were not fertilised and were incorporated in the soil at harvest.
- 25% of plant material were left in the paddock as residue for Pak choi.
- Automatic irrigation was triggered whenever there was ≤60% available water between December and March each year.

N fertiliser rates and expected yields for cabbage, broccoli, cauliflower, lettuce, maize and oats were mostly based on value the values given by Stout (2021), however, N fertiliser rates were restricted at sowing as described by Reid and Morton (2019) in order to decrease leaching risks (Table C7 in the Appendix C). For beetroot and Pak choi, fertiliser rates were based on the grower's management at the Green vegetables site (Tables 3.7 and 3.8 in Chapter 3). For other crops, the yields and fertiliser rates were based on Reid and Morton (2019) and Bloomer et al. (2020). The recommended N fertiliser rate to apply assumed that the soil had a maximum of 50 kg N ha^{-1}.

4.4 Results

Annual rainfall in the catchment was 1056 mm and 1270 mm for 2020 and 2021, respectively (Figure 4.1). Most intense rainfall occurred in December in both 2020 and 2021. In the Potatoes site, irrigation in 2020 totalled 30 mm and was applied on 18th of February, in 2021, 30 mm were applied on the 29th of January and 15th of February, totalling 60 mm, and in 2022, 30 mm were applied on the 14th of January and the 27th of January. Total cumulative irrigation in the Potatoes site was 150 mm during the period of study.

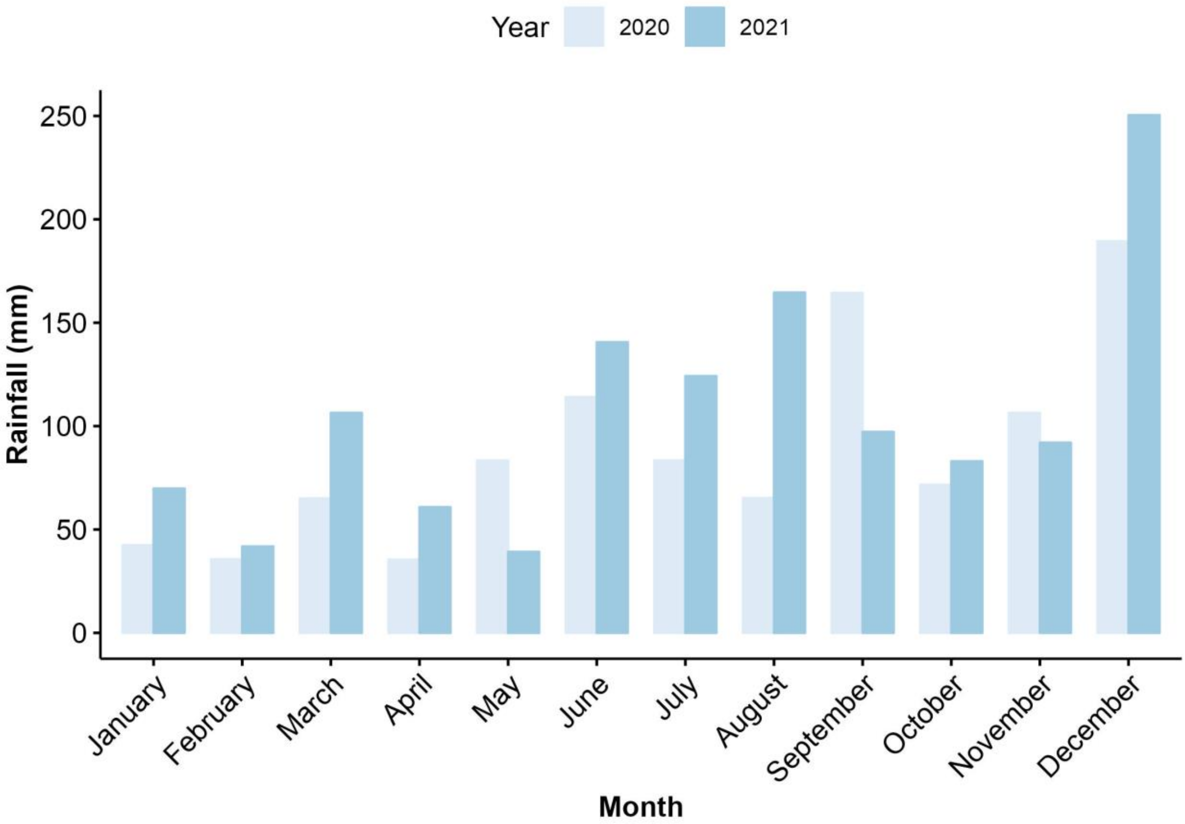

Figure 4.1. Monthly rainfall during the sampling period (covering two years) in the Arawhata catchment.

4.4.1 Potatoes site

4.3.1.1 Soil moisture – first crop season

As for the measured values, predicted soil moisture content showed similar values between treatments in the topsoil (0-30 cm). In all treatments soil moisture was very low (close to Permanent Wilting Point) in early autumn and hovered around field capacity over winter and spring. A comparison between the measured topsoil (0-30 cm) and the APSIM predicted values for topsoil (0-30 cm) volumetric moisture content showed 'satisfactory' agreement, as indicated by each *NSE* value (Figure 4.2).

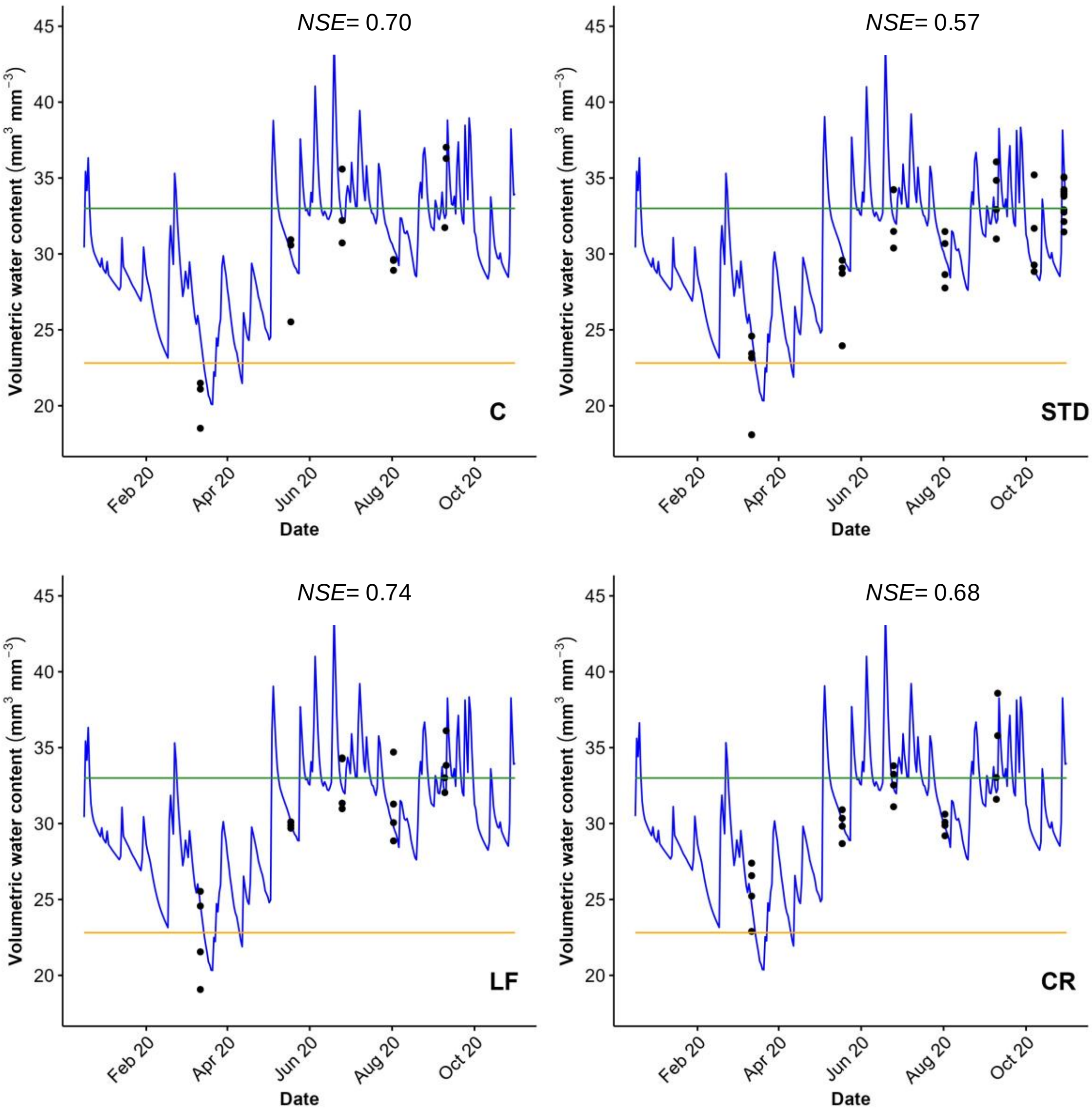

Figure 4.2. Volumetric moisture content in the 0-30 cm soil depth for each treatment (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser). Black dots indicate the observed data, the blue line indicates the predicted values, the green line indicates the field capacity value, and the orange line indicates the permanent wilting point value.

4.3.1.2 Soil moisture – second crop season

Predicted and measured values agreed well in the second season (Figure 4.3) and again there were similar values between the treatments. The smallest mean topsoil moisture between November 2020 to August 2021 was simulated in the C treatment with 31.7 mm^3

mm^{-3}, and the fertilised treatments were 0.4 to 0.9% greater. When considering both potato seasons, the model had an overall *NSE* of 0.63 for the STD treatment and 0.66 for the C treatment.

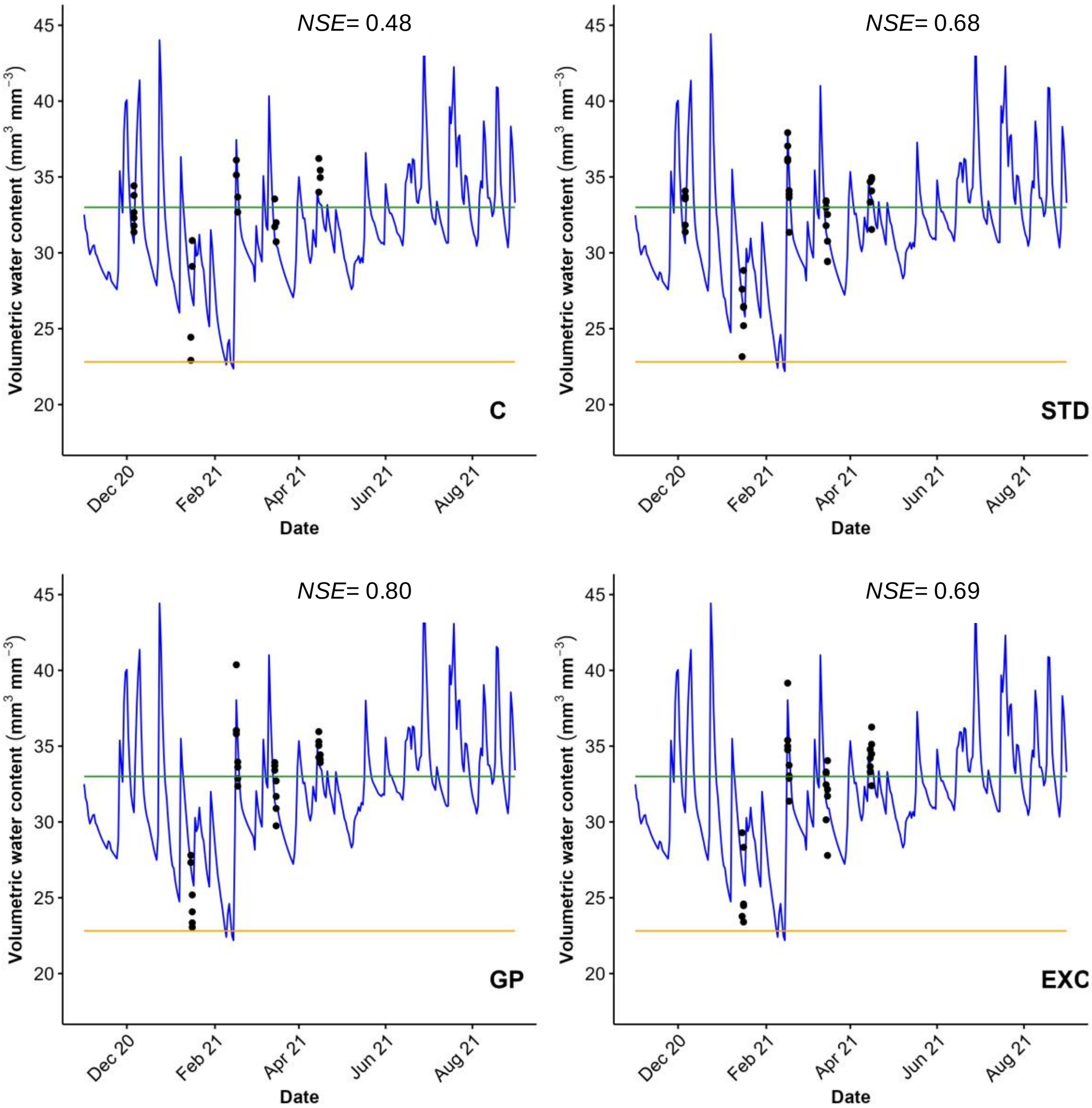

Figure 4.3. Volumetric moisture content in the 0-30 cm soil depth for each treatment (C for control; STD for standard practice; GP for good practice and EXC for excess). Black dots indicate the observed data, the blue line indicates the predicted values, the green line indicates the field capacity value, and the orange line indicates the permanent wilting point value.

4.3.1.3 Soil moisture probes

A comparison between simulated volumetric topsoil moisture contents and values monitored by both probes in the field (installed in the STD treatment) confirmed the 'satisfactory' agreement between the predicted and measured soil moisture contents (Figure 4.4). The overall *NSE* was 0.62 when comparing the probe and the predicted values across both seasons.

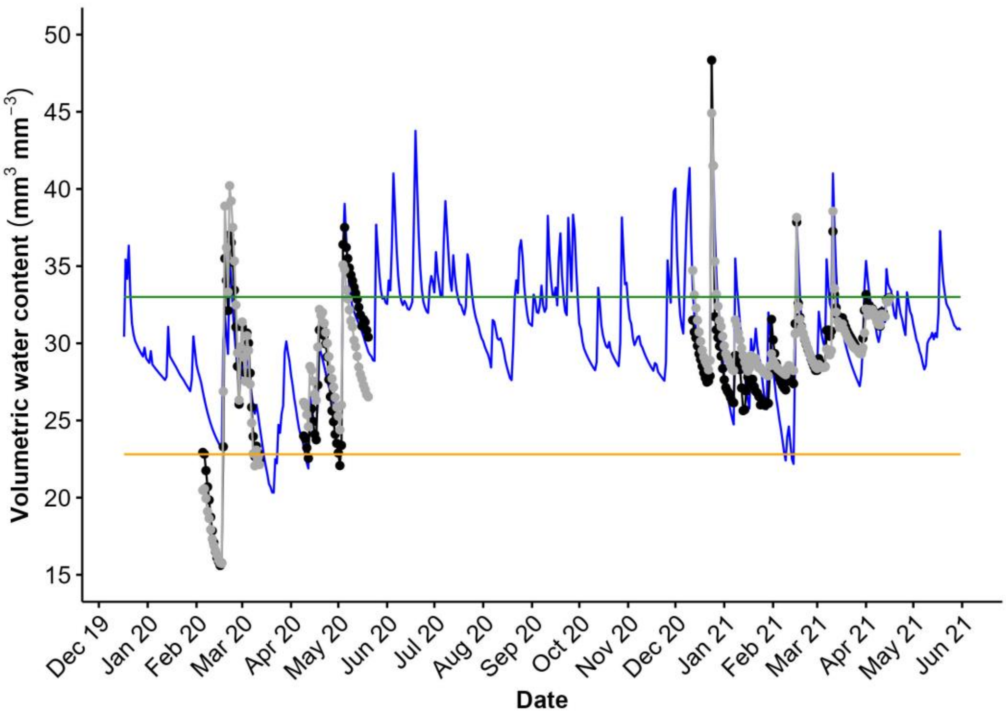

Figure 4.4. Volumetric moisture content in the 0-30 cm soil depth on the STD treatment. Black dots with black line indicate the observed data from probe 1, grey dots with grey line indicate the observed data from probe 2, the blue line indicates the predicted values, the green line indicates the field capacity value, and the orange line indicates the permanent wilting point value.

In the subsoil (30-60 cm), APSIM was also able to predict the moisture content reasonably well, with an *NSE* of 0.44 (Figure 4.5).

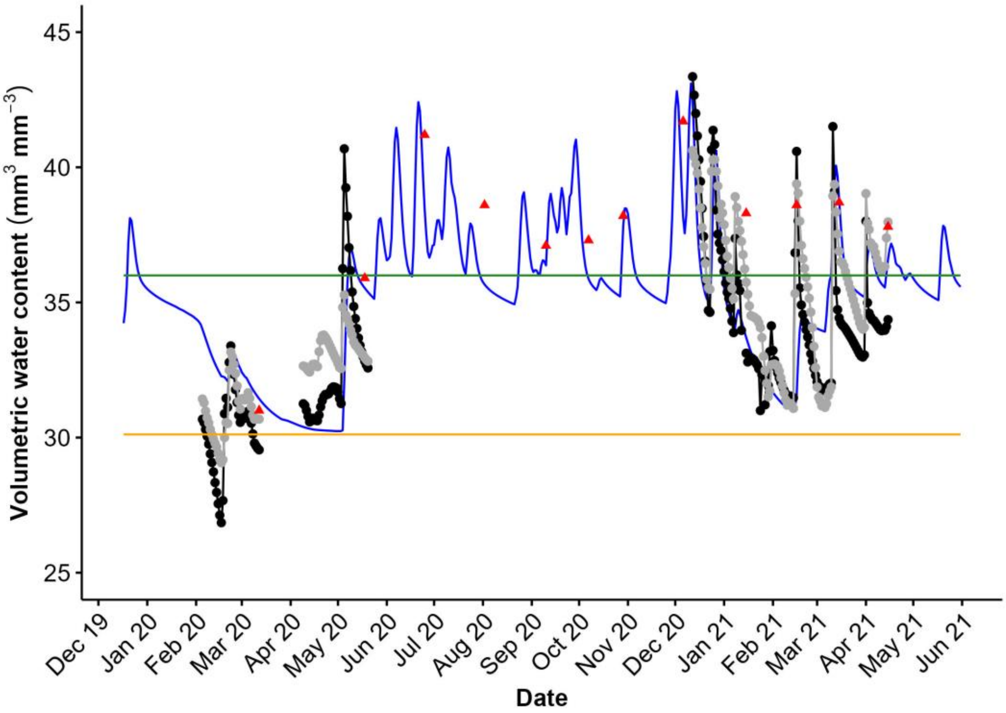

Figure 4.5. Volumetric moisture content in the 30-60 cm soil depth. Black dots with black line indicate the observed data from probe 1, grey dots with grey line indicate the observed data from probe 2, red triangles indicate the mean observed data from oven-dried samples blue line indicates the predicted values, the green line indicates the field capacity value, and the orange line indicates the permanent wilting point value.

4.3.1.4 Soil nitrate-N concentrations – first crop season

APSIM predicted that in soil nitrate-N concentrations would peak a few days after N fertiliser was applied to the fertilised treatments (Figure 4.6). APSIM's predictions of topsoil nitrate-N concentration had an 'unsatisfactory' *NSE* value when compared to the measurements, except for the LF treatment (Figure 4.6). The predicted and measured values deviated the most in the early stages (first 8 weeks); comparisons after the 27th of February 2020, showed much better agreement across the fertilised treatments, with an *NSE* of 0.5 in the STD treatment, -0.09 in the LF treatment and -0.2 in the CR treatment. Although the measured data and the model predictions were not aligned during peak nitrate-N concentrations in the soil, importantly, APSIM correctly estimated how much nitrate-N was available for leaching at the commencement of the drainage season (see agreement in early winter values in Figure 4.16). Generally, the modelled concentrations followed a similar trend to the observed data.

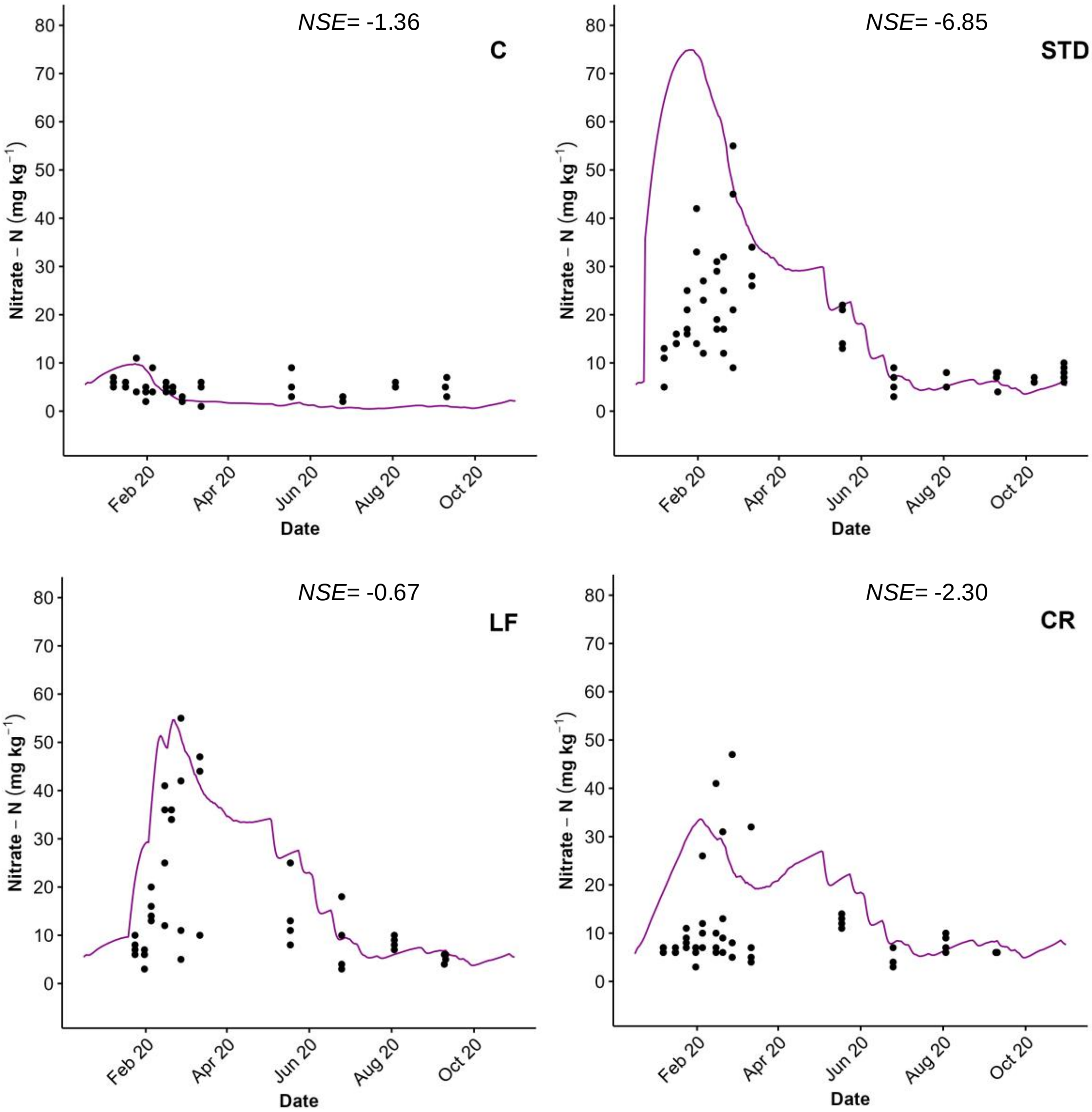

Figure 4.6. Nitrate-N concentrations (mg kg^{-1}) in the 0-30 cm of soil from each treatment (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser). Black dots indicate the observed data, and the magenta line indicates the predicted values.

Subsoil nitrate-N

The measured subsoil nitrate-N concentrations showed only small variations across the season, but proportionally high variability within each sampling date (Figure 4.7). Despite what visually seems to be, overall, a reasonable agreement between observed and predicted data, the value of *NSE* was considered 'satisfactory' only in the LF and CR treatments (Figure 4.7).

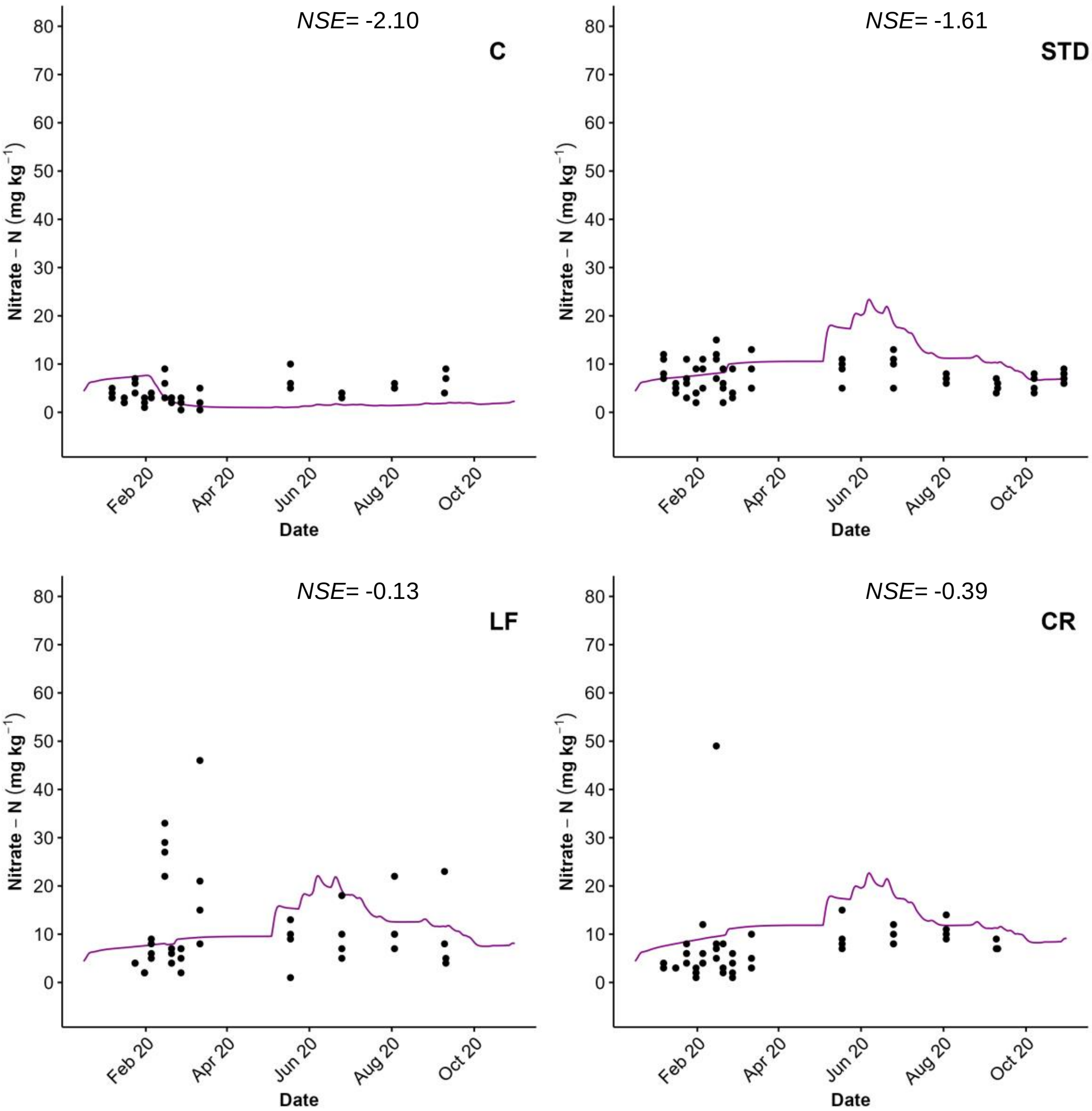

Figure 4.7. Nitrate-N concentrations (mg kg^{-1}) in the 30-60 cm of soil from each treatment (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser). Black dots indicate observed data, and the magenta line indicates predicted values.

4.3.1.5 Soil nitrate-N concentrations – second crop season

Topsoil nitrate

Predicted and measured topsoil nitrate-N concentrations in the second season varied in a somewhat similar manner to the first year (Figure 4.8). The predictive power of the model was considered ‘satisfactory’ in the STD, GP and EXC treatments, and ‘unsatisfactory’ in the C treatment, where within-date variability of the observed data was comparable to that between dates (Figure 4.8).

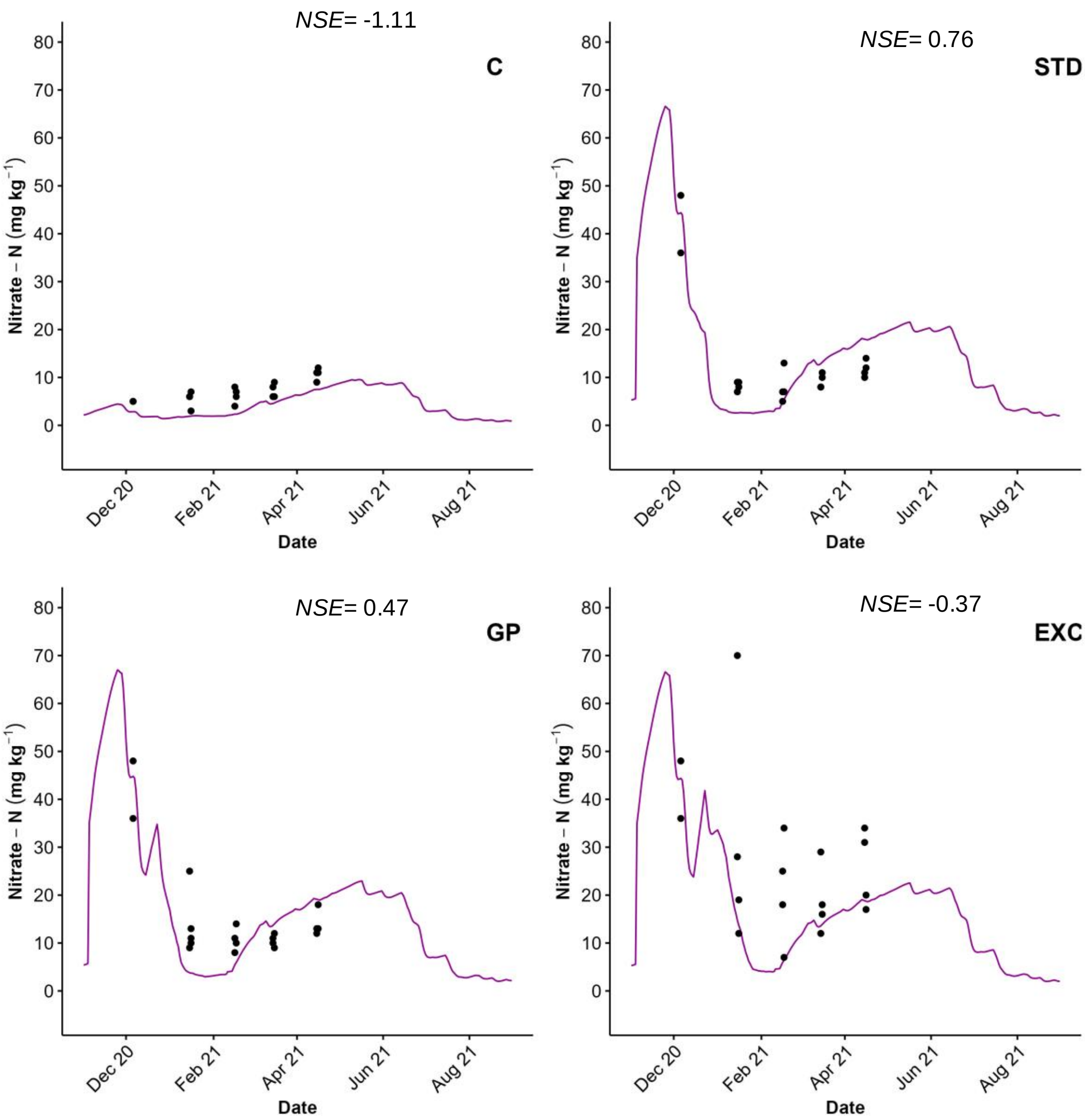

Figure 4.8. Nitrate-N concentrations (mg kg^{-1}) in the 0-30 cm of soil from each treatment (C for control; STD for standard practice; GP for good practice and EXC for excess). Black dots indicate observed data, and the magenta line indicates predicted values.

Subsoil nitrate

Similar to the first season, visual inspection would suggest that there is good agreement between the model and observed data (Figure 4.9). However, NSE values were 'satisfactory' only for the STD and GP treatments.

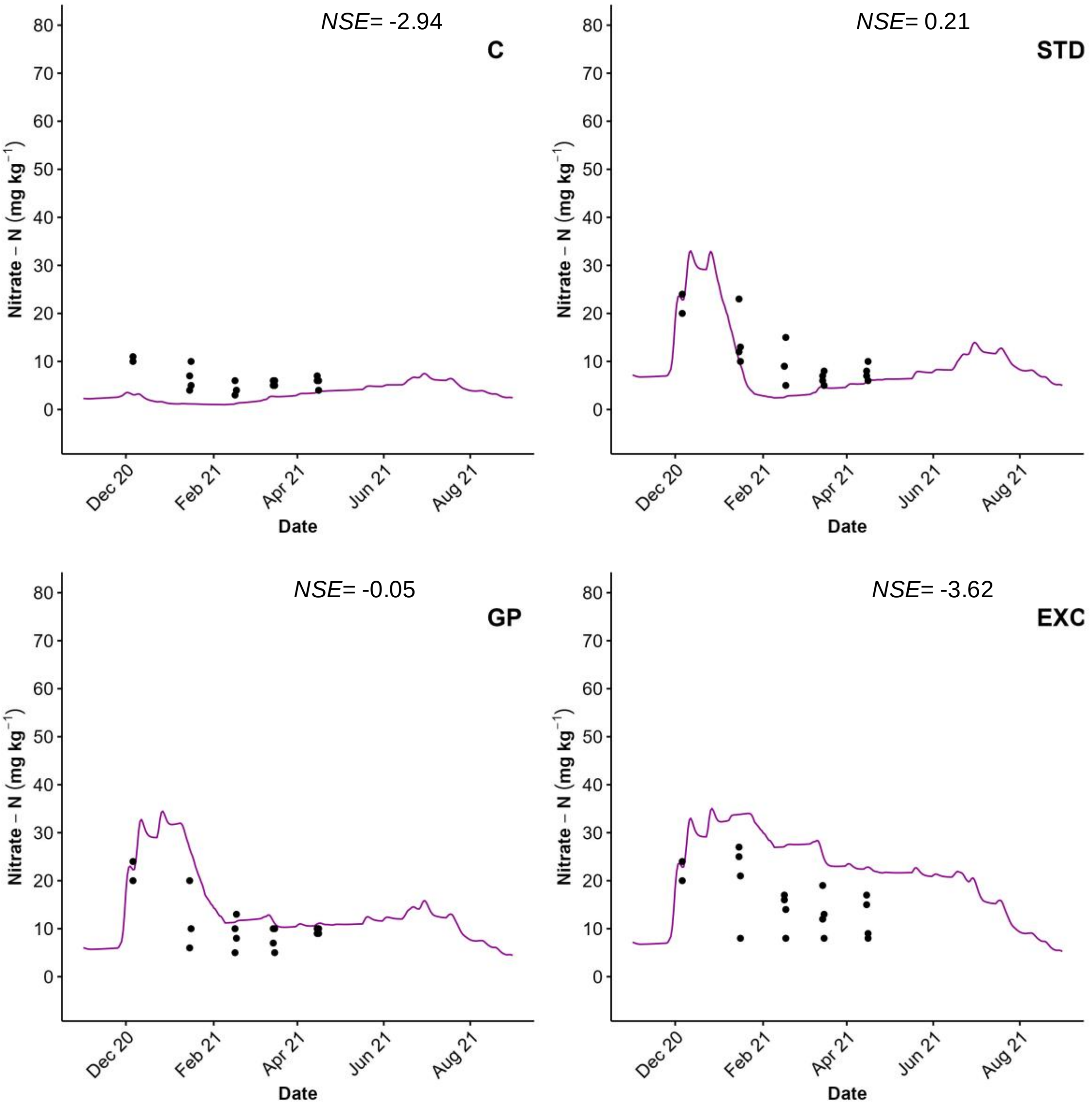

Figure 4.9. Nitrate-N concentrations (mg kg^{-1}) in the 30-60 cm of soil from each treatment (C for control; STD for standard practice; GP for good practice and EXC for excess). Black dots indicate observed data, and the magenta line indicates predicted values.

4.3.1.6 Drainage and nitrate-N leaching fluxes – first crop season (2019-2020)

Monthly drainage at 100 cm simulated by APSIM was similar between the treatments, with the C treatment having slightly more drainage than the rest of the treatments. Most of the drainage occurred in winter and spring months (Figure 4.10). The drainage season coincides with the fallow period, when potato tubers are being ground-stored. Cumulative drainage during this period accounted for 28% of the rainfall for the same period (Table 4.10).

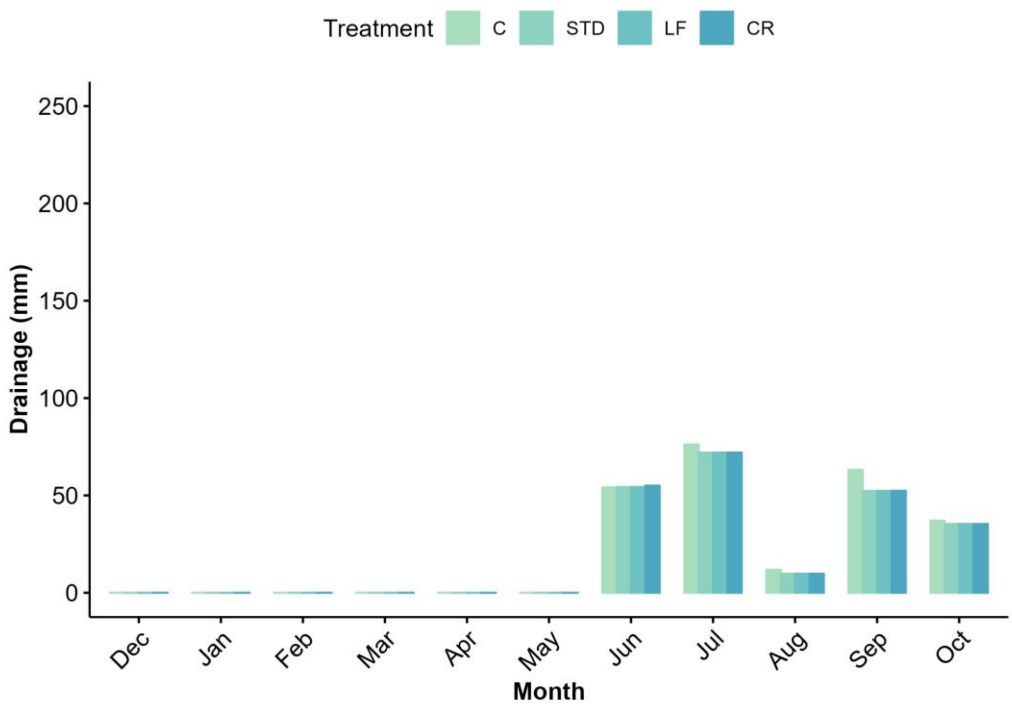

Figure 4.10. Simulated monthly drainage volumes (mm) at 100 cm for the first crop season from each treatment (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser).

Table 4.10. Cumulative simulated drainage volumes (mm) for the first potato growing season at 100 cm soil depth (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser).

Treatment	Cumulative drainage at 100 cm soil depth (mm)	Proportion of rainfall (+ irrigation) (%)
C	242	29
STD	224	27
LF	224	27
CR	225	27

Following a similar distribution pattern to the drainage, simulated nitrate-N leaching fluxes in the crop growing season occurred mostly during winter and spring (Figure 4.11). The C treatment had noticeably smaller nitrate-N leaching losses (10 kg N ha^{-1}) than the rest of the treatments (50 – 51 kg N ha^{-1}) (Table 4.11). There were no appreciable differences between the fertilised treatments, with the largest difference being 1 kg N ha^{-1} between both the STD and CR treatment, and the LF treatment (Table 4.11).

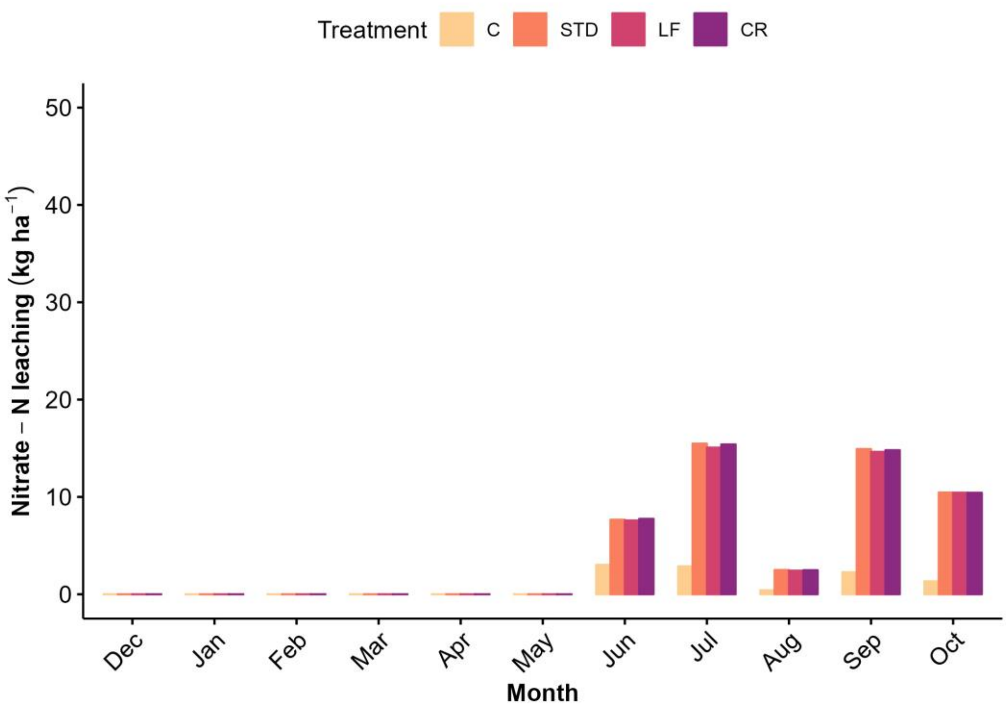

Figure 4.11. Simulated monthly nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 100 cm for the first crop season from each treatment (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser).

Table 4.11. Cumulative simulated nitrate-N leaching losses for the first potato growing season at 100 cm soil depth (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser).

Treatment	**Cumulative nitrate-N leaching at 100 cm soil depth (kg ha^{-1})**
C	10
STD	51
LF	50
CR	51

4.3.1.7 Drainage and nitrate-N leaching fluxes – second crop season (2020-2021)

A comparison between measured and simulated nitrate-N leaching (April 2021 – August 2021)

As previously mentioned in Chapter 3, measured nitrate-N concentrations in the drainage water during the winter of the second growing season showed very large variability within each sampling date, and this variability was considerably greater than any variation between dates (Figure 4.12). According to the values of *NSE*, APSIM predictions of nitrate-N concentrations in the drainage water did not agree well with measured values (Figure 4.12). Only the C treatment had an ‘satisfactory’ *NSE* coefficient, whereas the STD treatment simulation had the largest disagreement. Given the variability in the measured data, perhaps this lack of agreement is unsurprising. APSIM estimates did show some visual agreement with general trends, but the large variations meant that any comparison was not really possible. Nonetheless, comparing the cumulative nitrate-N leaching losses, computed using APSIM simulated drainage volumes, with APSIM’s predictions resulted in *NSE* coefficients that can be considered as ‘satisfactory’, despite the large variability (Figure 4.13). This agreement suggests that indeed the trend in N concentration were captured well enough by APSIM, but was not good enough for individual dates. The cumulative values are therefore used for further analyses, but these results should be taken with some caution.

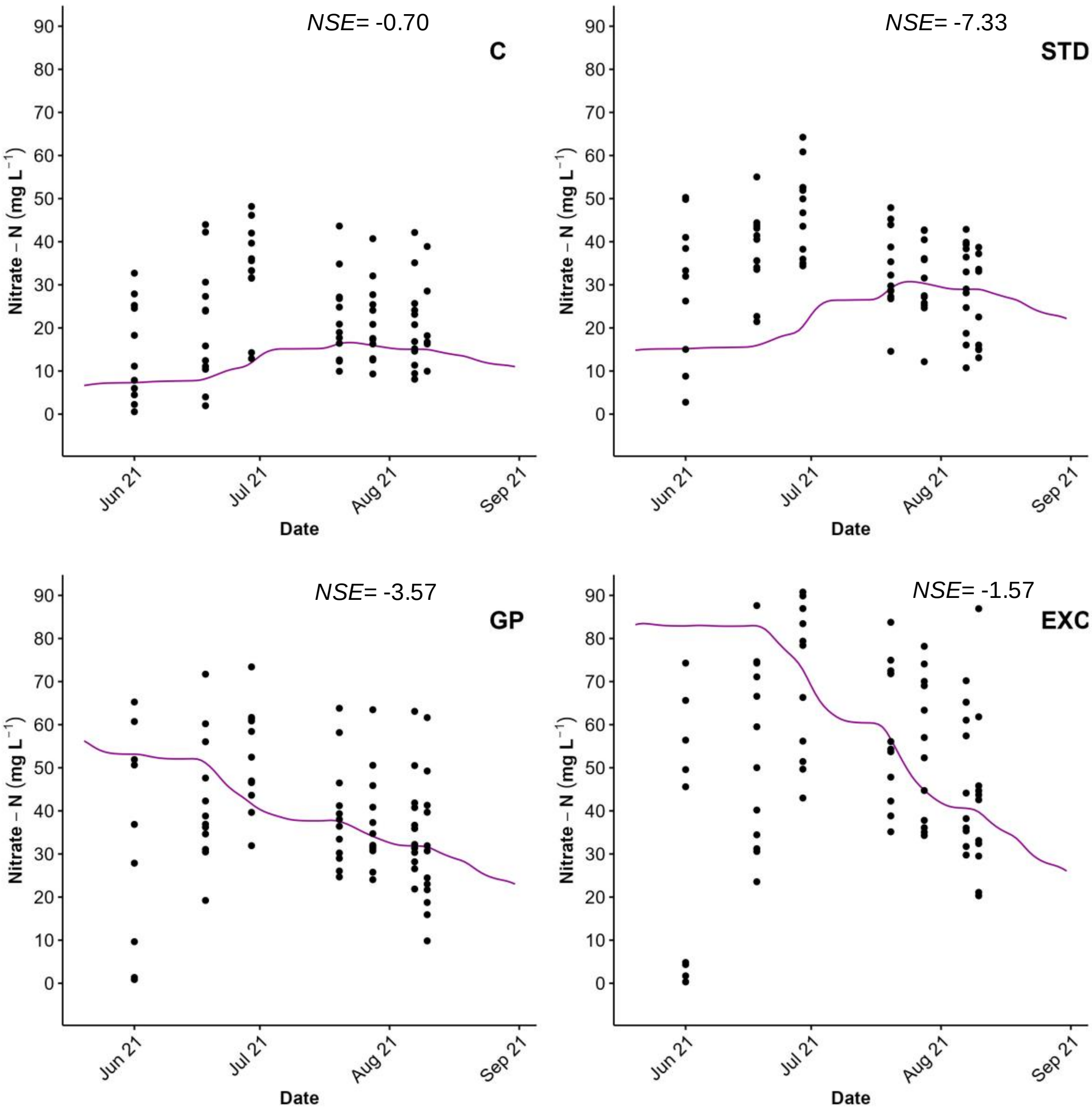

Figure 4.12. Nitrate-N concentrations (mg L^{-1}) in the drainage water at 60 cm soil depth from each treatment for the period April 2021 to August 2021 (C for control; STD for standard practice; GP for good practice, and EXC for excess). Black dots indicate observed data, and the magenta line indicates predicted values.

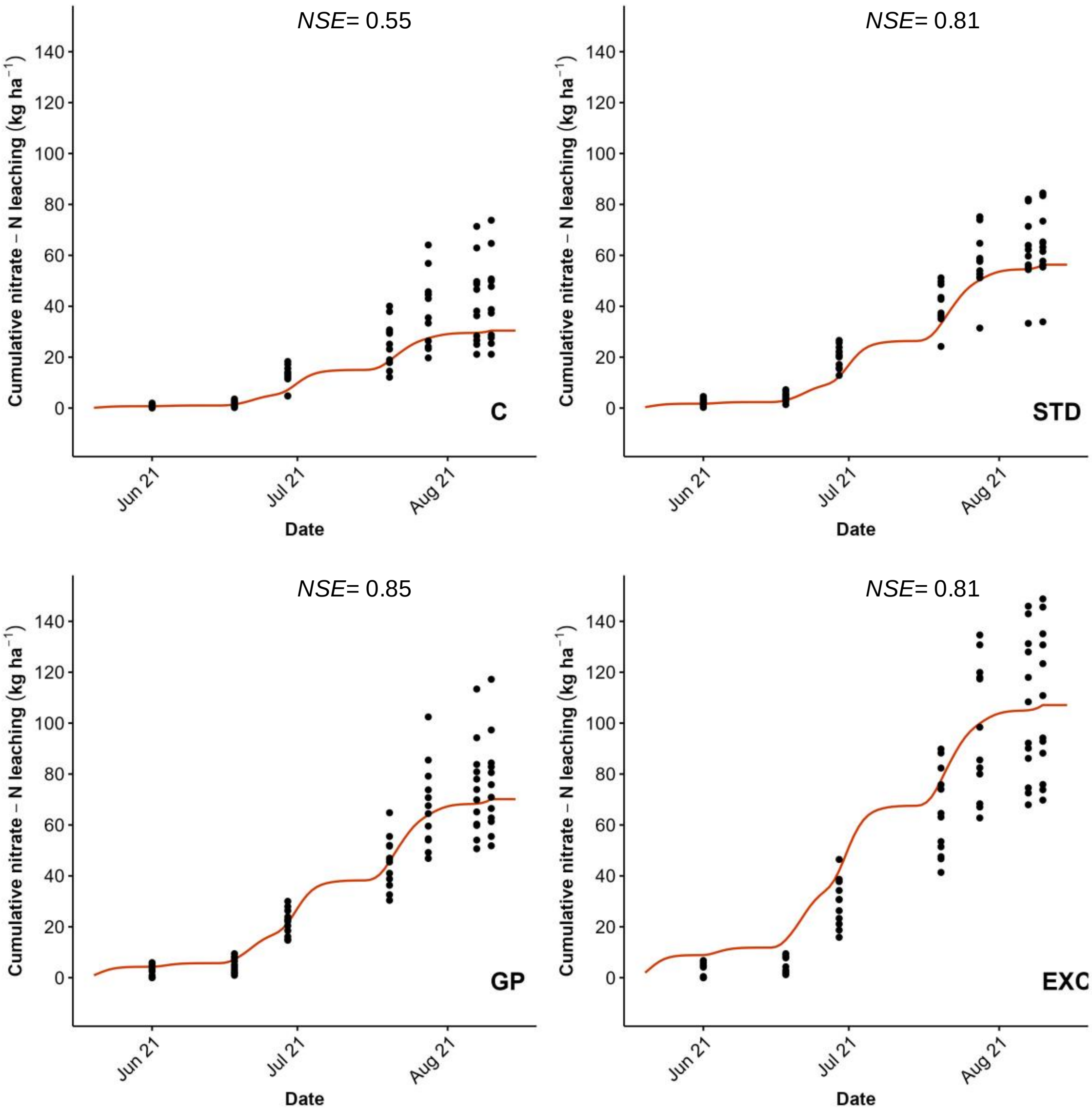

Figure 4.13. Cumulative nitrate-N leaching losses (kg nitrate-N ha^{-1}) in the drainage water at 60 cm soil depth using simulated drainage from each treatment for the period April 2021 to August 2021 (C for control; STD for standard practice; GP for good practice, and EXC for excess). Black dots indicate data based on observed values of nitrate-N concentrations and orange line indicates predicted values.

The predicted and observed cumulative nitrate-N leaching losses for each of the treatments were similar, with the control treatment having the greatest difference between the measured and simulated values (Table 4.12). Both, the APSIM predictions and the observed data agreed that the EXC treatment had the highest cumulative nitrate-N leaching, followed by the GP, STD, and C treatments.

Table 4.12. A comparison of observed mean cumulative nitrate-N leaching losses (kg nitrate-N ha^{-1}) measured using suction cups at 60 cm with the leaching values predicted by APSIM for the period April 2021 to August 2021 (C for control; STD for standard practice; GP for good practice, and EXC for excess). Standard deviation given in brackets.

Treatment	Observed cumulative nitrate-N leaching (kg ha^{-1})	Predicted cumulative nitrate-N leaching (kg ha^{-1})
C	43 (18)	30
STD	65 (19)	56
GP	76 (24)	70
EXC	109 (34)	107

Simulated leaching losses for the second season at 100 cm soil depth

For the second growing season, predicted monthly drainage was, again, very similar across the different treatments (Figure 4.14). Significant drainage events occurred in December 2020, although most of the drainage was again concentrated in winter. The drainage that occurred in summer, as a result of considerable rainfall (190 mm in the summer of 2020, compared to an annual average of 104 mm), coincided with the initial stages of crop growth, and the drainage that occurred in winter was from bare soil. Cumulative drainage during this period accounted for 36% of the rainfall for the same period (Table 4.13).

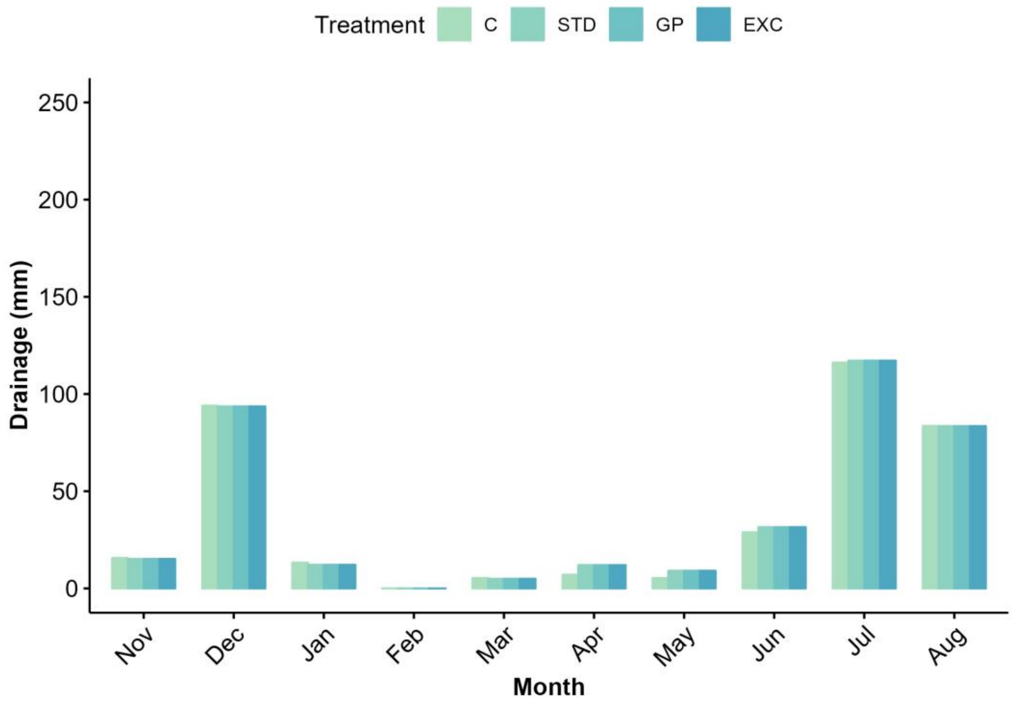

Figure 4.14. Simulated monthly drainage volumes (mm) at 100 cm for the second crop season (year) from each treatment (C for control; STD for standard practice; GP for good practice and EXC for excess).

Table 4.13. Cumulative simulated drainage volumes (mm) for the second potato growing season at a 100 cm soil depth (C for control; STD for standard practice; GP for good practice, and EXC for excess).

Treatment	Cumulative drainage at 100 cm soil depth (mm)	Proportion of rainfall (+ irrigation) (%)
C	369	34
STD	379	35
GP	379	35
EXC	379	35

Most of the leaching occurred in December 2020 and the winter of 2021 (Figure 4.15). There were important differences in simulated monthly nitrate-N leaching across treatments, particularly after June 2021. The EXC treatment had the greatest cumulative value of nitrate-N leaching, totalling 141 kg ha^{-1}, followed by treatments GP, STD, and C, respectively (Table 4.14).

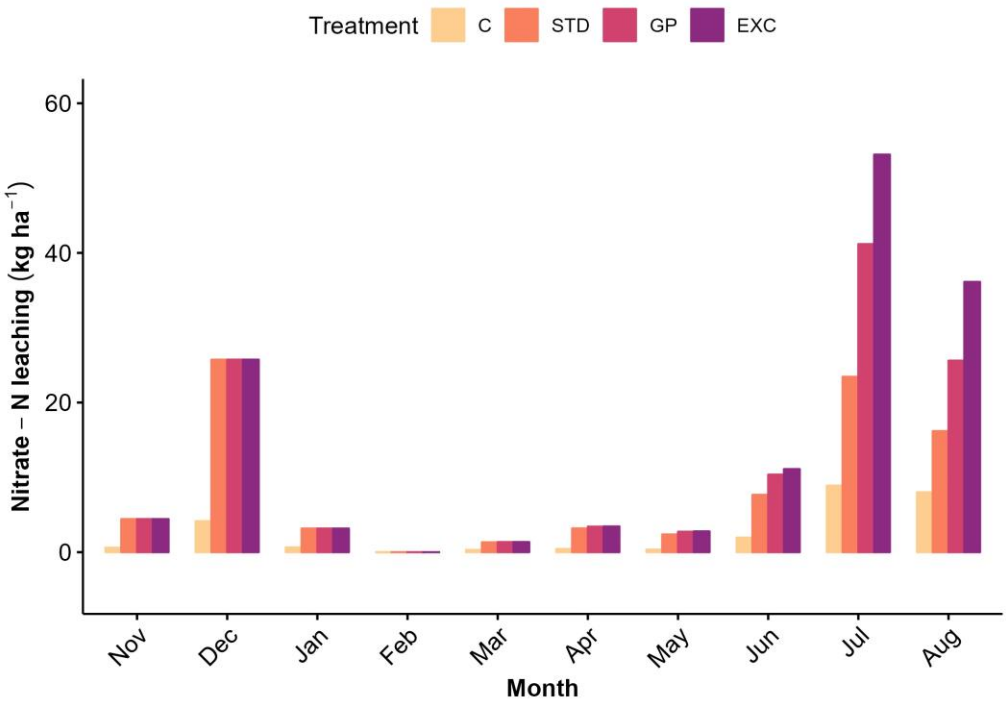

Figure 4.15. Simulated monthly nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 100 cm for the second crop season from each treatment (C for control; STD for standard practice; GP for good practice and EXC for excess).

Table 4.14. Cumulative simulated nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 100 cm for the second potato growing season (C for control; STD for standard practice; GP for good practice, and EXC for excess).

Treatment	**Cumulative nitrate-N leaching at 100 cm soil depth (kg ha^{-1})**
C	25
STD	88
GP	118
EXC	141

4.3.1.8 Nitrate-N leaching fluxes – third crop season (2021-2022)

In the onion growing season, nitrate-N concentrations in drainage water were also very variable at each sampling date. Despite this, the values were better predicted by APSIM than was the case for potatoes over its second growing season (Figure 4.16). The model had an 'satisfactory' performance for both treatments as measured by the *NSE* values.

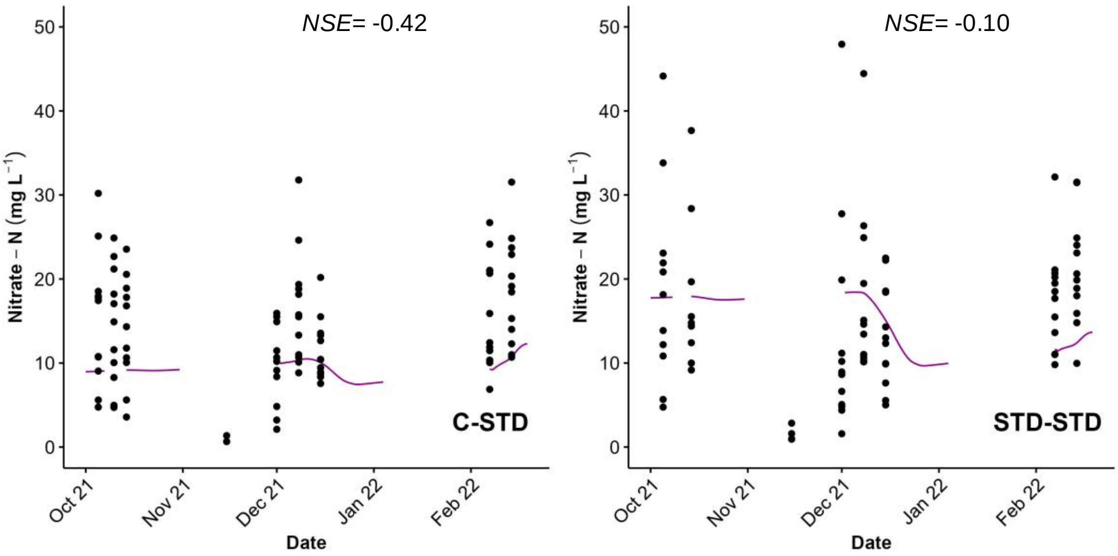

Figure 4.16. Nitrate-N concentrations (mg L^{-1}) in the drainage water from each treatment for the sampling period (C-STD for control in the previous growing season and standard in the current season; STD-STD for standard practice in the previous growing season and standard practice in the current growing season). Black dots indicate observed data, and the magenta line indicates predicted values.

Similar to the results for the potato growing season, the predicted cumulative nitrate-N leaching under onions agreed reasonably well with predictions from APSIM (Figure 4.17, Table 4.15). This agreement provides further support for the ability of APSIM to predict cumulative nitrate-N leaching which is the most important output here.

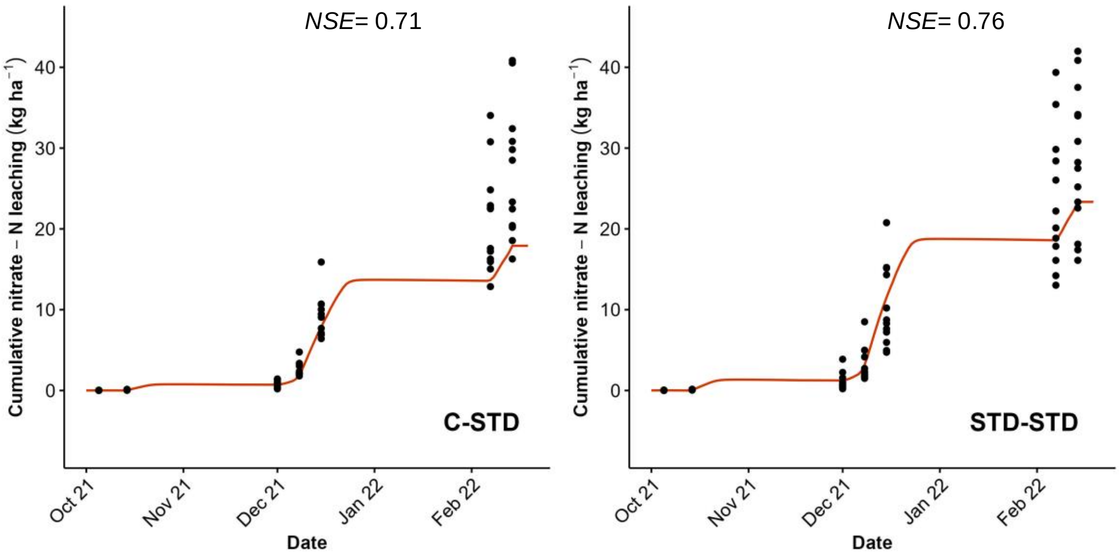

Figure 4.17. Cumulative nitrate-N leaching losses (kg nitrate-N ha^{-1}) in the drainage water using simulated drainage from each treatment for the sampling period (C-STD for control in the previous growing season and standard in the current season; STD-STD for standard practice in the previous growing season and standard practice in the current growing season). Black dots indicate observed data, and the orange line indicates predicted values.

Table 4.15. A comparison of mean cumulative nitrate-N leaching losses (kg nitrate-N ha^{-1}) estimated using suction cups with the leaching values predicted by APSIM (C-STD for control in the previous growing season and standard in the current season; STD-STD for standard practice in the previous growing season and standard practice in the current growing season). Standard deviation given in brackets.

Treatment	Observed cumulative nitrate-N leaching (kg ha^{-1})	Predicted cumulative nitrate-N leaching (kg ha^{-1})
C-STD	27 (10)	18
STD-STD	30 (12)	23

4.3.1.9 Crop yields

For the first potato growing season, APSIM's predicted yields for treatments C and STD were different to the observed yields; the predicted yield for C was 1.3 times higher, and the STD predicted yield was 0.75 times lower than the observed data (Table 4.16). For the rest of the treatments, predicted and observed yields were quite similar.

Table 4.16. Mean observed and predicted yields (t ha^{-1}) for the first potato growing season (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser). Standard deviation given in brackets.

Treatment	Mean observed yields (t ha^{-1})	Mean predicted yields (t ha^{-1})
C	28 (4)	36
STD	52 (7)	39
LF	36 (2)	39
CR	42 (6)	39

For the second potato growing season, the yield predictions for both C and STD were higher than the observed ones, with APSIM's yields being 1.5 and 1.25 times the observed yields, respectively (Table 4.17). For the rest of the treatments, yields were very similar.

Table 4.17. Mean observed and predicted yields (t ha^{-1}) of the second potato growing season (C for control; STD for standard practice; GP for good practice and EXC for excess). Standard deviation given in brackets.

Treatment	Mean observed yields (t ha^{-1})	Mean predicted yields (t ha^{-1})
C	17 (5)	26
STD	36 (10)	45
GP	45 (7)	45
EXC	45 (8)	45

4.4.2 Green vegetables site - lysimeters

4.4.2.1 Soil moisture

For the green vegetables site, topsoil (0-20 cm) volumetric moisture contents predicted by APSIM showed no differences between the treatments, which agreed with the results based on the measured data (Figure 4.18). Calculated *NSE* coefficients were 'satisfactory', with the STD and C treatments having the greatest *NSE* coefficient with 0.5, followed by the RF/EXC and CM treatment (Figure 4.18).

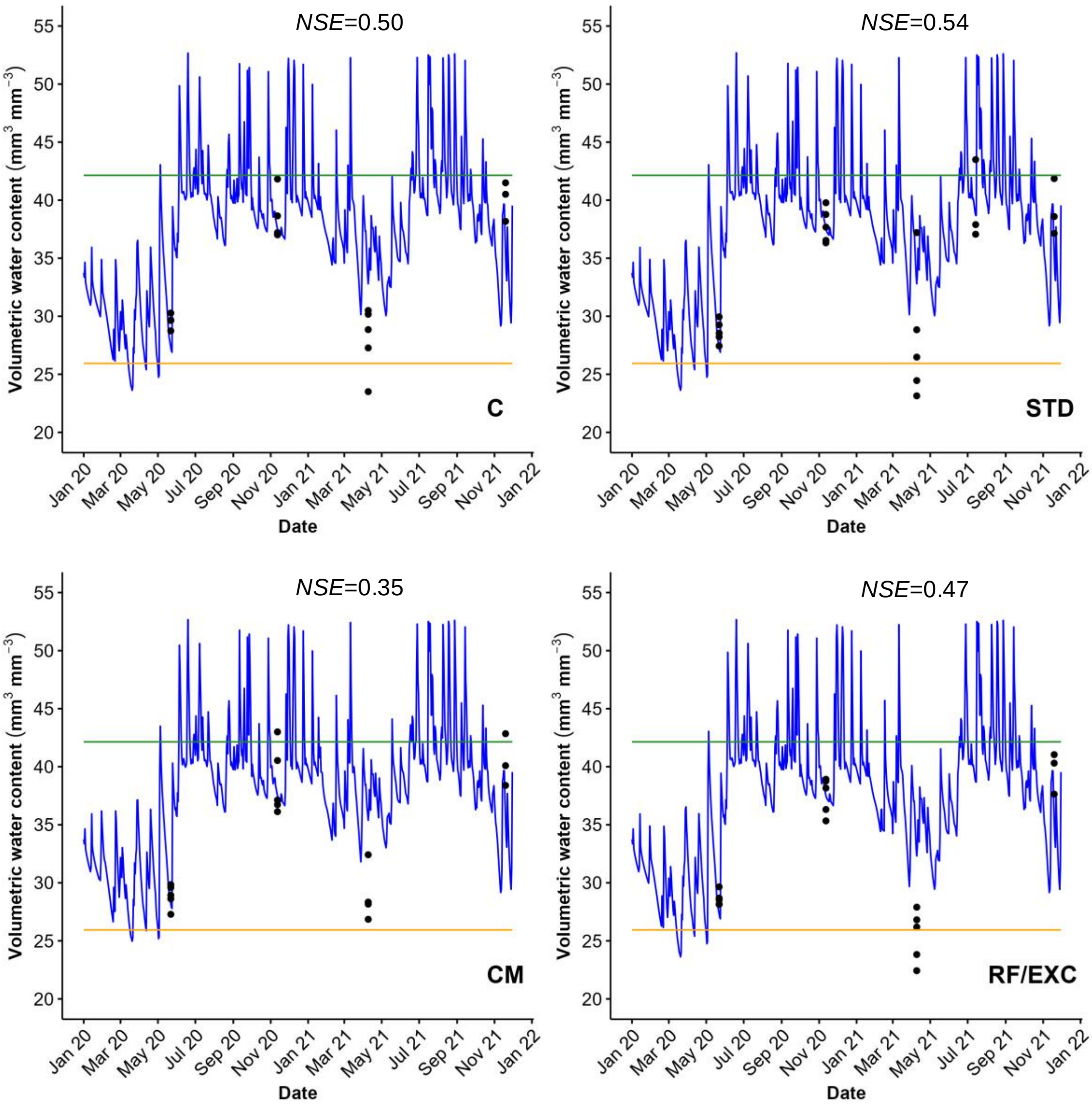

Figure 4.18. Volumetric moisture content in the 0-20 cm soil depth for each treatment in the lysimeters (C for control; STD for standard practice; CM for chicken manure; RF for reduced fertilisation and EXC for excess). Black dots indicate observed data, the blue line indicates predicted values, the green line indicates the field capacity value, and the orange line indicates permanent wilting point value.

4.4.2.2 Soil nitrate-N concentrations

APSIM's prediction for topsoil nitrate-N concentrations observed in the lysimeters scored very 'satisfactory' against measurements according to the *NSE* coefficient (Figure 4.19). The largest *NSE* was achieved in the CM treatment with 0.84, followed by the RF/EXC, C, and STD treatments.

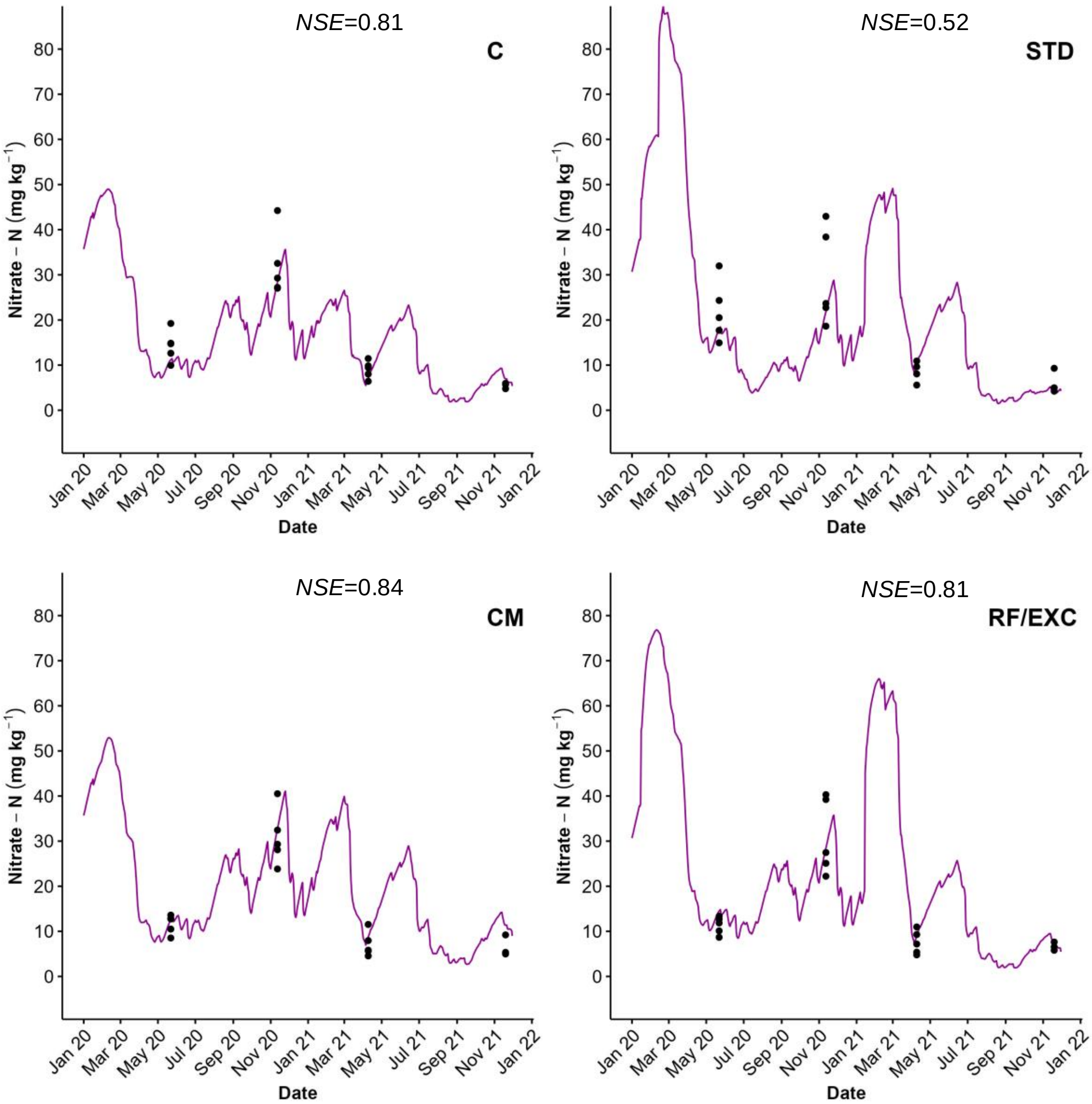

Figure 4.19. Nitrate-N concentrations (mg kg^{-1}) in the 0-20 cm of soil from each treatment in the lysimeters (C for control; STD for standard practice; CM for chicken manure; RF for reduced fertilisation and EXC for excess). Black dots indicate observed data, and the magenta line indicates predicted values.

4.4.2.3 Drainage and nitrate-N leaching fluxes – first year (2020)

Drainage collected at the bottom of the lysimeter showed large variability within treatments (Figure 4.20). In the first year of measurements, APSIM was able to predict well the drainage for all treatments (Figure 4.20). The greatest *NSE* score was achieved in the RF treatment, followed by CM, C, and STD. There were no significant differences in cumulative drainage between the treatments, with simulated cumulative drainage of 292 mm, 296 mm, 296 mm, and 292 mm of drainage for the C, STD, CM, and RF treatments, respectively. Similar to the Potatoes site, most of the drainage occurred during winter and spring. Cumulative drainage accounted for 33% of the rainfall amount in the same period.

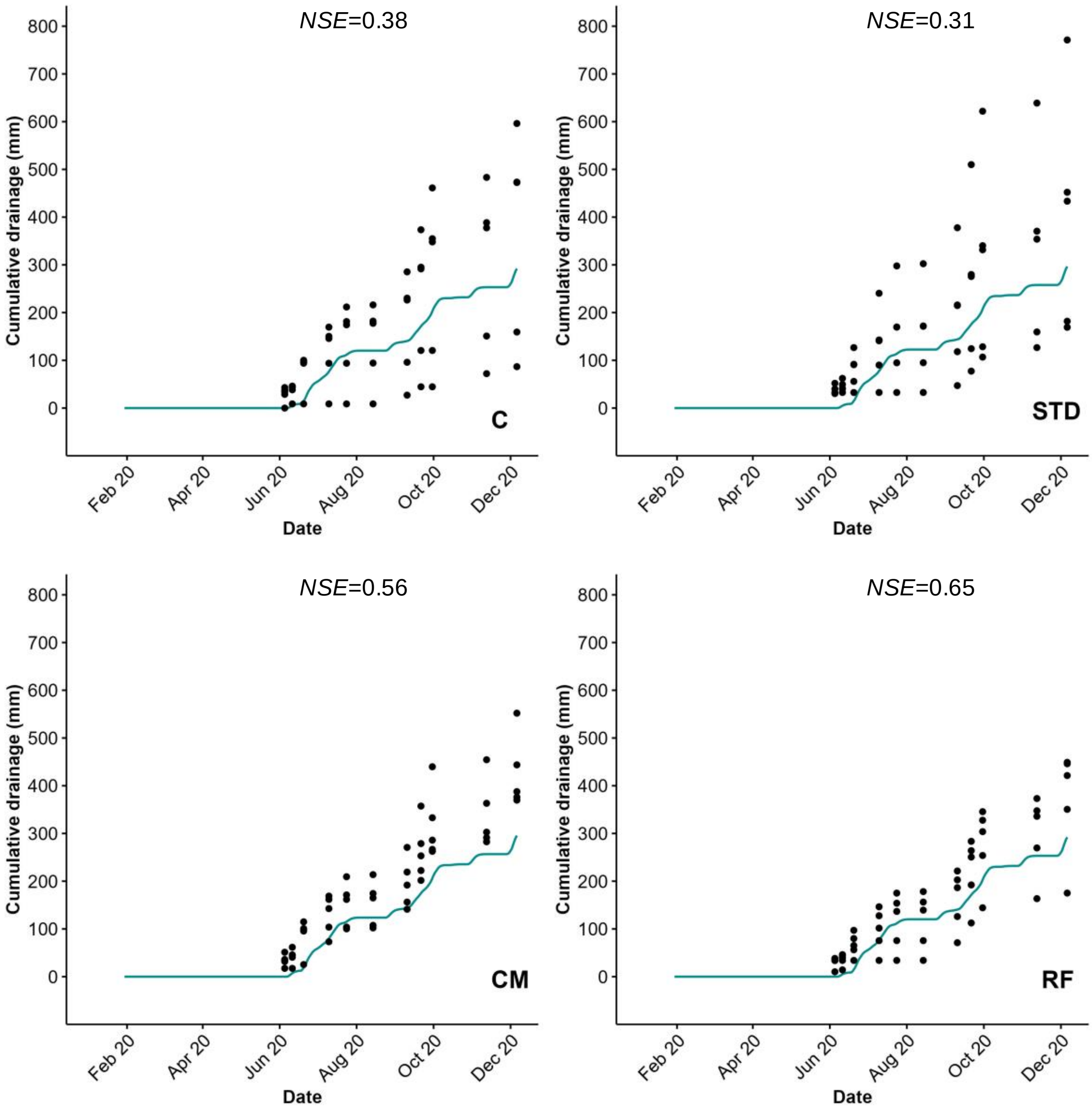

Figure 4.20. Cumulative drainage volumes (mm) for the first year of sampling for each treatment in the lysimeters (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertilisation). Black dots indicate observed data and blue line indicates predicted values.

Most of the nitrate-N leaching in the first year of measurements in the Green vegetables site occurred between mid-June and the end of September 2020. The model's cumulative nitrate-N leaching losses predictions were 'satisfactory' when compared to the measured values, with the greatest *NSE* value for the CM treatment at 0.73, and the lowest for the STD at 0.28 (Figure 4.21). The highest simulated cumulative nitrate-N leaching loss was 240 kg ha^{-1} in the STD treatment, followed by 206, 185 and 176 kg ha^{-1} in the RF, CM and C treatments, respectively. Leaching losses simulated by APSIM were similar to the measured mean cumulative nitrate-N leaching losses (Table 3.28), with only 4% mean increase.

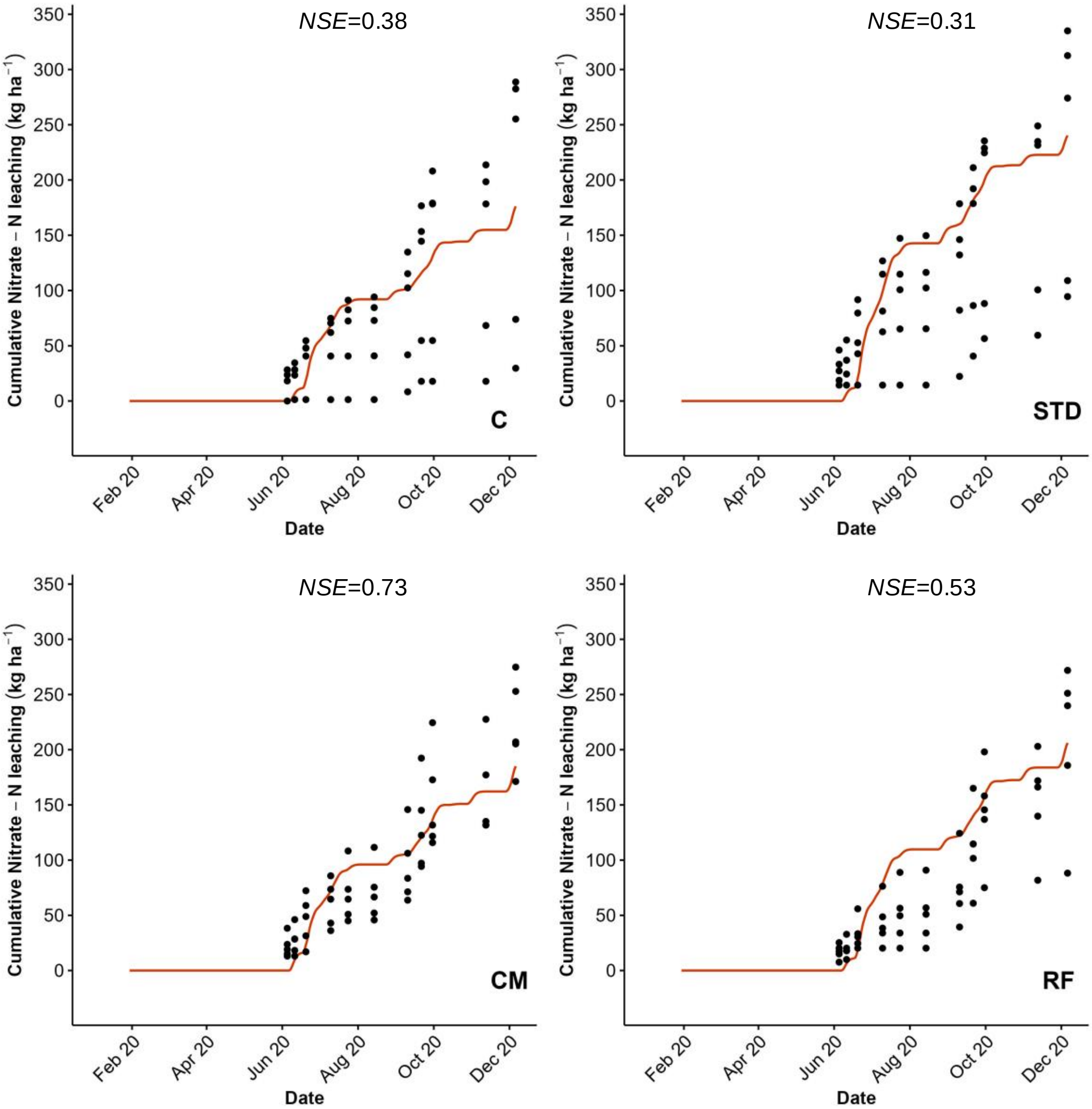

Figure 4.21. Cumulative nitrate-N leaching losses (kg nitrate-N ha^{-1}) for the first year of sampling for each treatment in the lysimeters (C for control; STD for standard practice; CM for chicken manure and RF for reduced fertilisation). Black dots indicate observed data, and the orange line indicates predicted values.

4.4.2.4 Drainage and nitrate-N leaching fluxes – second year (2021)

Similar to the first year (2020), drainage values were highly variable across the lysimeters. APSIM's ability to simulate cumulative drainage was 'satisfactory' across all treatments (Figure 4.22). The highest *NSE* (0.7) was calculated for the EXC treatment and the lowest (0.4) in the C treatment. Total simulated cumulative drainage was 512 mm for the C treatment, 512 mm for the STD treatment, 533 mm for the CM treatment, and 508 mm for the EXC treatment. Simulated cumulative drainage in the lysimeters in the second year was 44% of the cumulative rainfall for the same period and most of the drainage happened between July and October 2021.

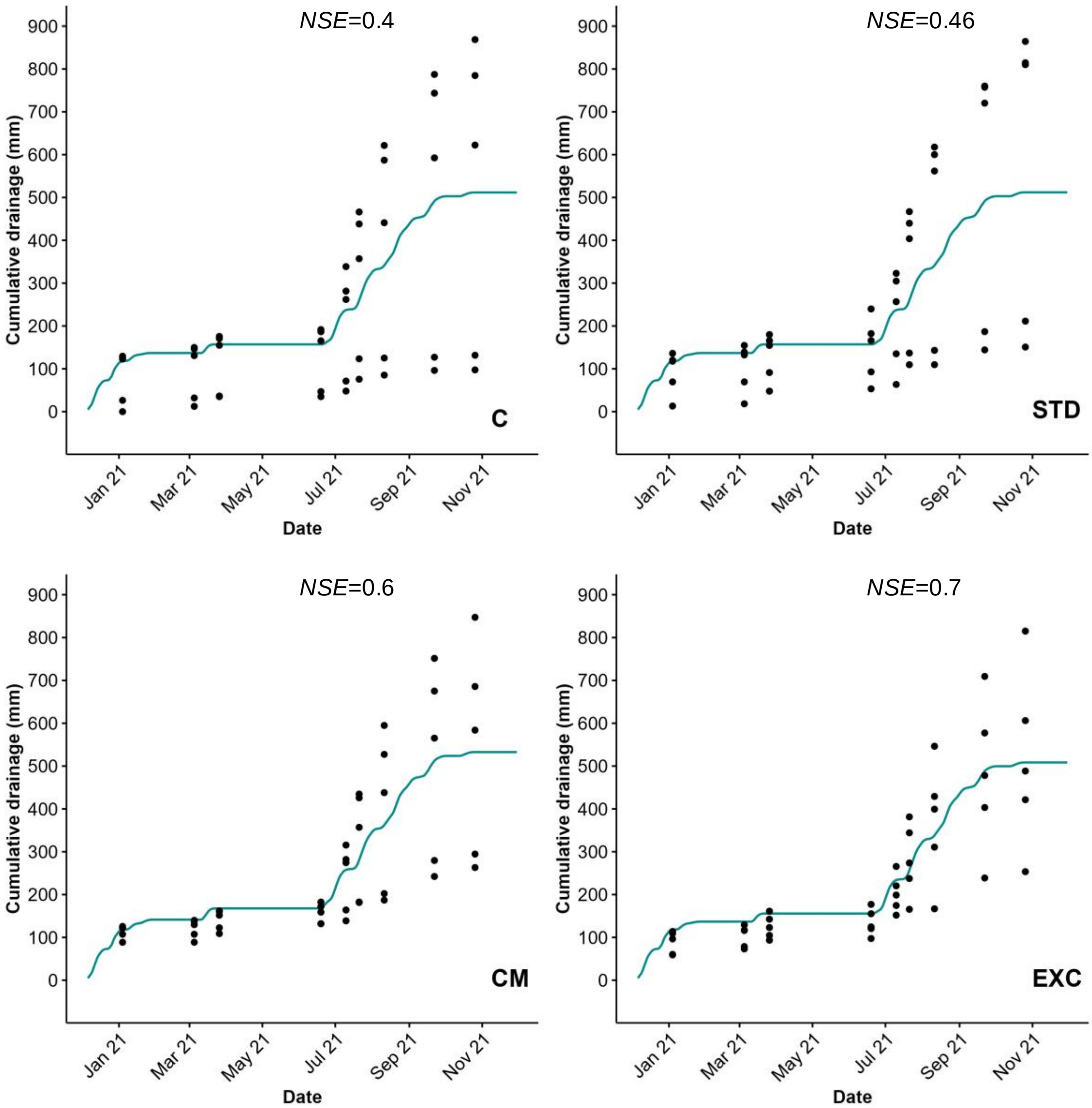

Figure 4.22. Cumulative drainage (mm) for the second year of sampling for each treatment in the lysimeters (C for control; STD for standard practice; CM for chicken manure and EXC for excess). Black dots indicate observed data, and the blue line indicates predicted values.

Similar to the measured drainage amounts, cumulative nitrate-N losses showed very large variability across the lysimeters. APSIM simulated cumulative losses agreed well with the observed data for the second year of the experiment (Figure 4.23). Most of the simulated and observed leaching losses took place in the winter and spring of 2021. Total simulated cumulative nitrate-N losses were 181 kg ha^{-1} for the C treatment, 196 kg ha^{-1} for the STD treatment, 220 kg ha^{-1} for the CM treatment, and 213 kg ha^{-1} for the EXC treatment. Leaching losses simulated by APSIM were approximately 19% greater than the measured mean cumulative nitrate-N leaching losses (Table 3.31).

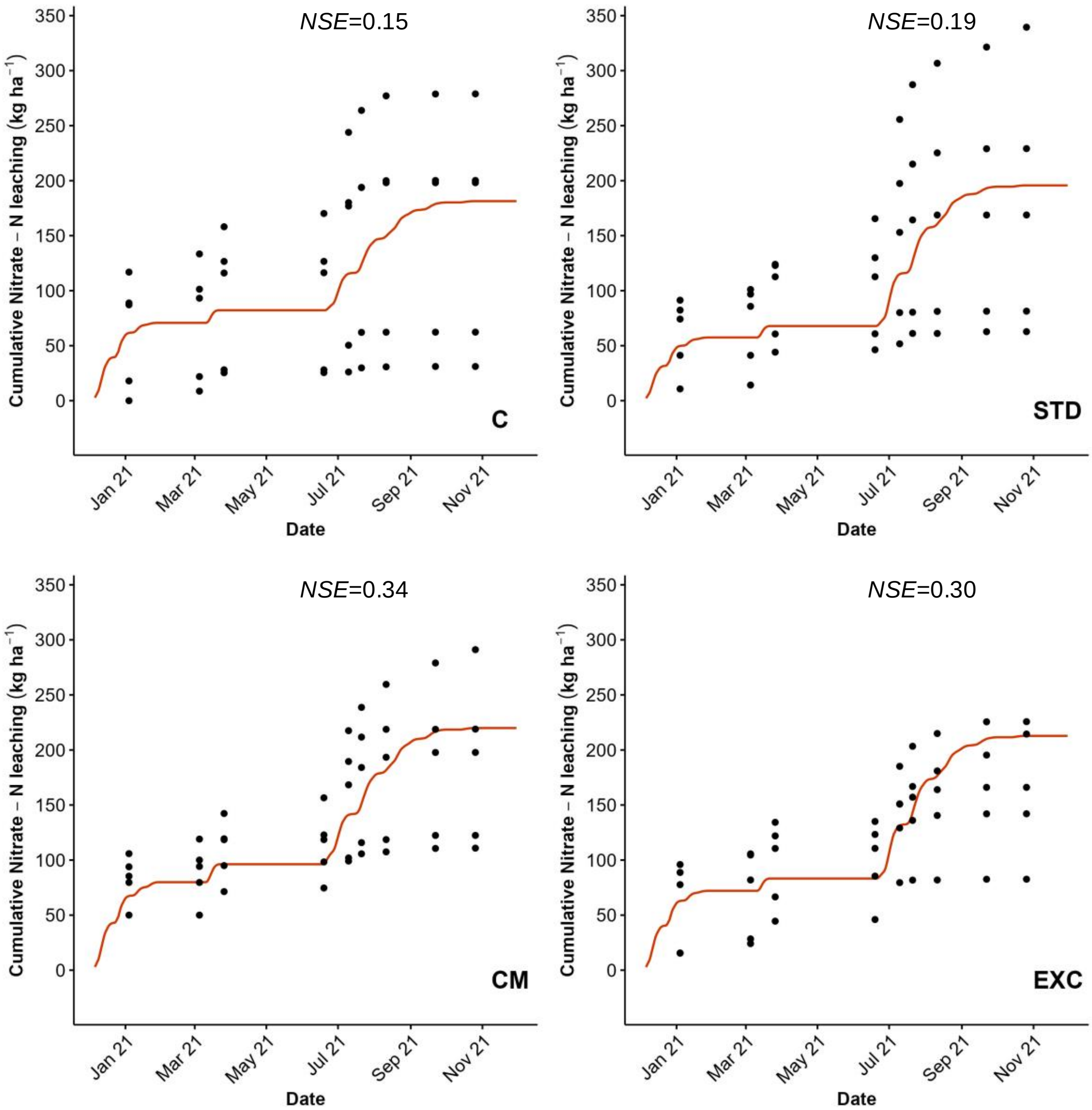

Figure 4.23. Cumulative nitrate-N leaching losses (kg nitrate-N ha^{-1}) for the second year of sampling for each treatment in the lysimeters (C for control; STD for standard practice; CM for chicken manure and EXC for excess). Black dots indicate observed data, and the orange line indicates predicted values.

4.4.3 Green vegetables site – field

Due to differences in soil physical properties between the lysimeters and the nearby field where green vegetables were grown, a parallel APSIM simulation was created to describe the field surrounding the lysimeters. Where necessary, soil physical parameters (such as *DUL*) were adjusted to reflect the properties of the field soil.

4.4.3.1 Soil moisture

There were no important differences in either measured or simulated volumetric soil moisture contents between treatments in the field (Figure 4.24). Visually, simulated values seem to fit well with observed data. However, *NSE* coefficients for C and RF/STD2 treatments where less than -1, which indicates a poor performance of APSIM to predict soil moisture values in the field. The soil moisture values in the field were variable.

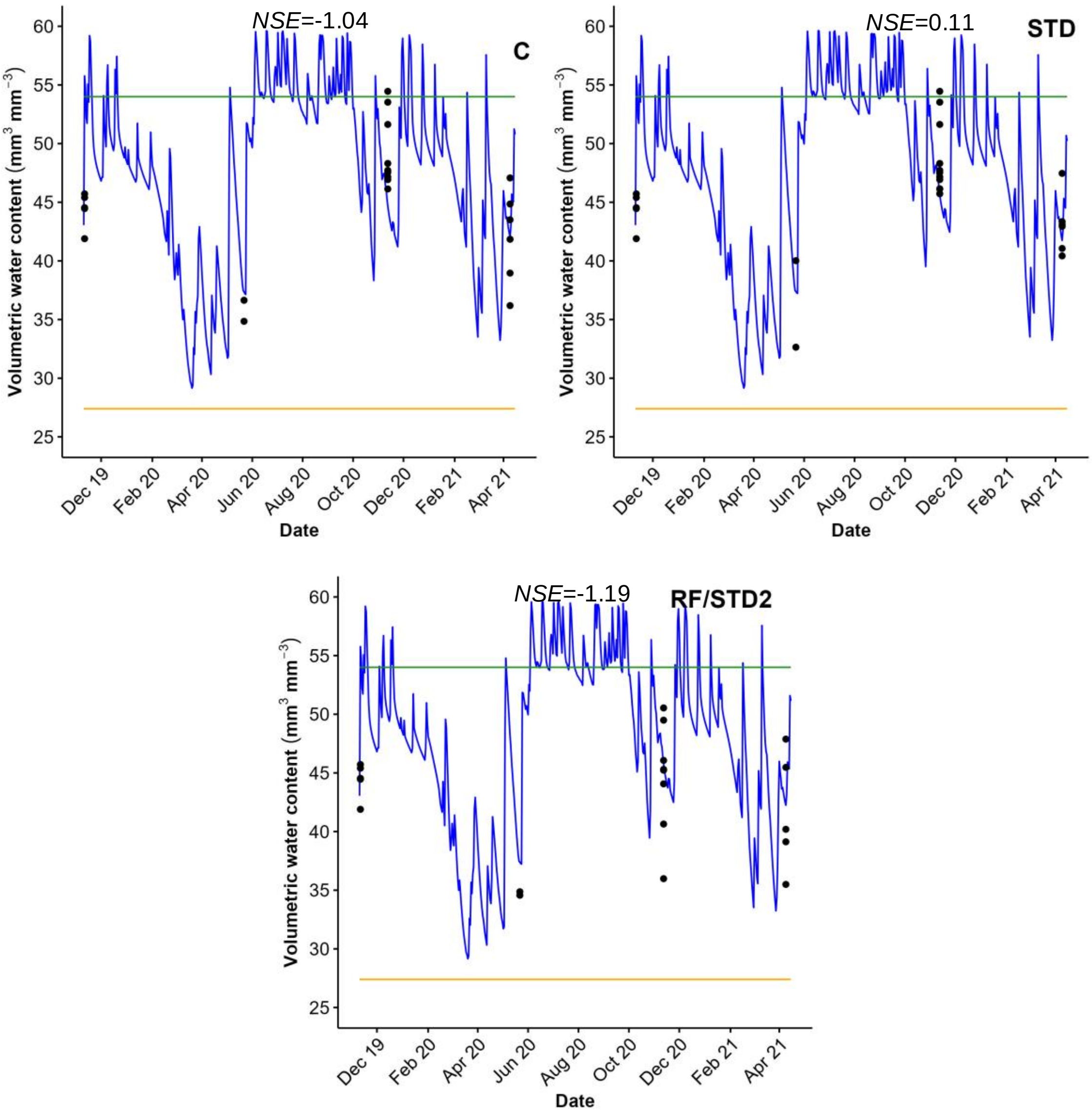

Figure 4.24. Volumetric moisture content in the 0-20 cm soil depth for each treatment (C for control; STD for standard practice; RF for reduced fertilisation and STD2 for standard fertiliser rate applied upfront at sowing). Black dots indicate observed data, the blue line indicates predicted values, the green line indicates the field capacity value, and the orange line indicates permanent wilting point value.

4.4.3.2 Soil nitrate-N concentrations

According to the *NSE* values, APSIM was able to predict the soil nitrate-N concentrations and trends in the soil after crop harvesting and at crop sowing (Figure 4.25).

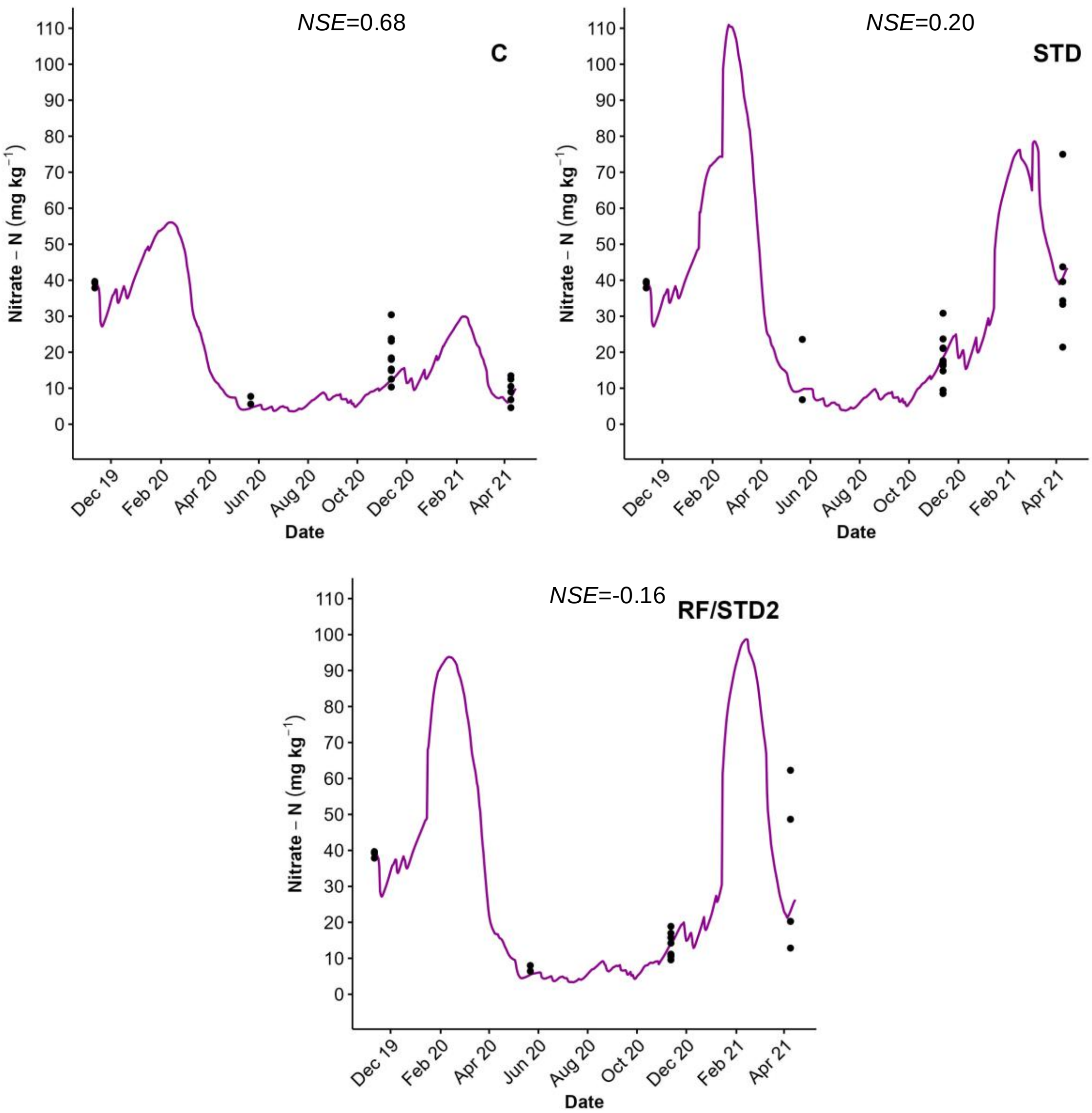

Figure 4.25. Nitrate-N concentrations (mg kg^{-1}) in the 0-20 cm of soil from each treatment in the field (C for control; STD for standard practice; RF for reduced fertilisation and STD2 for standard fertiliser rate applied upfront at sowing). Black dots indicate observed data, and the magenta line indicates predicted values.

4.4.3.3 Drainage and nitrate-N leaching fluxes

Similar to the results from the lysimeters, drainage in the field was predicted to occur mostly between the months of June and September, with a significant amount of drainage also occurred in December 2020 (Figure 4.26).

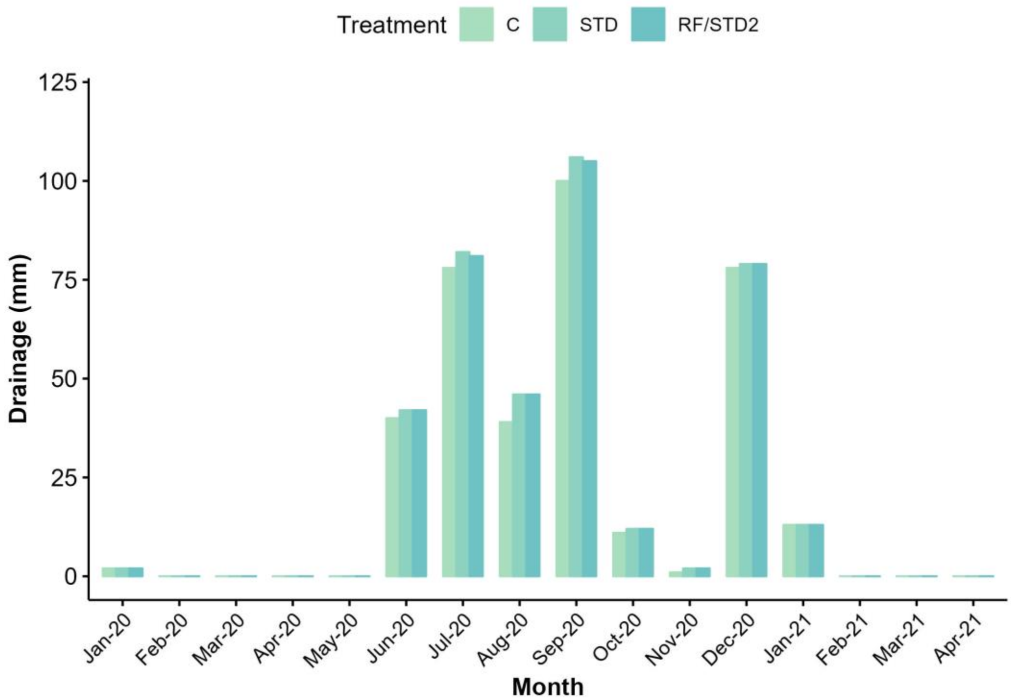

Figure 4.26. Simulated monthly drainage volumes (mm) at 100 cm from each treatment in the field (C for control; STD for standard practice; RF for reduced fertilisation and STD2 for standard fertiliser rate applied upfront at sowing).

In general, there were similar values of simulated cumulative drainage (mm) between treatments (Table 4.18). The C treatment had slightly smaller simulated drainage depths (6%) than the fertilised treatments. The total simulated drainage represented 33-35% of the rainfall recorded in the same period.

Table 4.18. Cumulative simulated drainage (mm) for the year 2020 at 100 cm soil depth from each treatment (C for control; STD for standard practice; RF for reduced fertilisation and STD2 for standard fertiliser rate applied upfront at sowing).

Treatment	Cumulative drainage at 100 cm soil depth (mm)	Proportion of rainfall (%)
C	351	33
STD	372	35
RF/STD2	373	35

According to APSIM, most nitrate-N leaching happened between June and September 2020, and then in December 2020 (Figure 4.27).

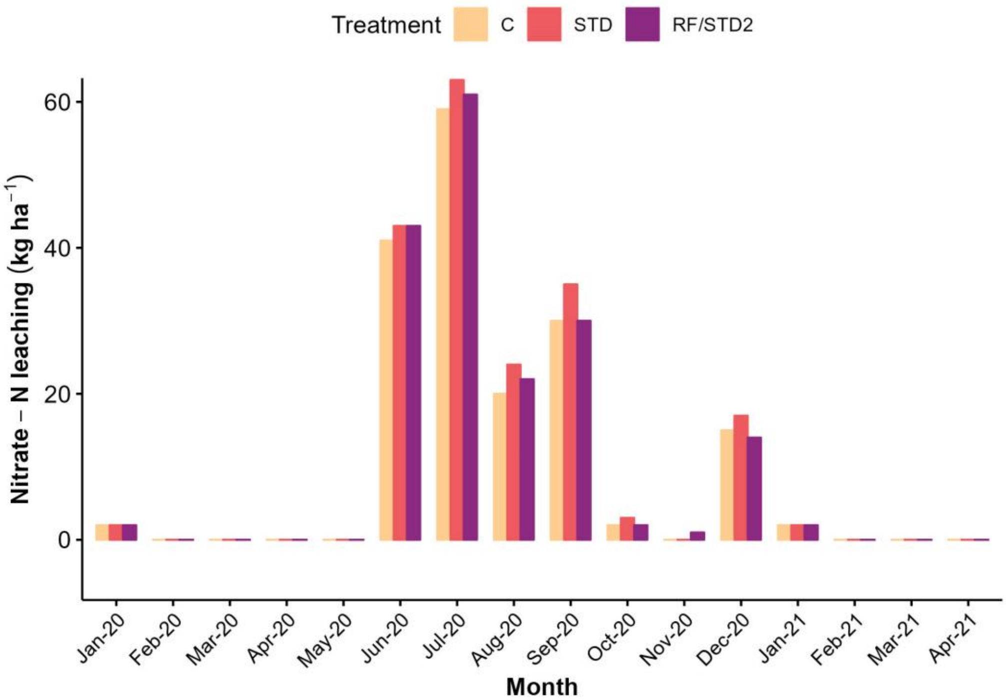

Figure 4.27. Simulated monthly nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 100 cm from each treatment in the field (C for control; STD for standard practice; RF for reduced fertilisation and STD2 for standard fertiliser applied upfront at sowing).

Despite having different N fertiliser rates, there were no noticeable differences in estimated nitrate-N leaching between the treatments (Table 4.19.). The STD and RF/STD2 treatments were 11 and 4% greater than the C treatment, respectively.

Table 4.19. Cumulative simulated nitrate-N leaching losses (kg nitrate-N ha^{-1}) for the year 2020 from each treatment in the field (C for control; STD for standard practice; RF for reduced fertilisation and STD2 for standard fertiliser applied upfront at sowing).

Treatment	**Cumulative nitrate-N leaching at 100 cm soil depth (kg ha^{-1})**
C	169
STD	187
RF/STD2	175

4.4.4 Long-term and in-field mitigation practices simulations

4.4.4.1 Potato-onion rotation

Results from long-term simulations (1991 to 2021) indicated that for a potato-onion rotation, half the years would have annual leaching losses (calculated at 100 cm soil depth) equal to or less than 25 kg nitrate-N ha^{-1} (Figure 4.28). The maximum value of nitrate-N leaching loss was 106 kg ha^{-1} in 2021 and the minimum was 7 kg ha^{-1} in 2003. Mean nitrate-N leaching loss for the rotation at this site was 47 kg ha^{-1}.

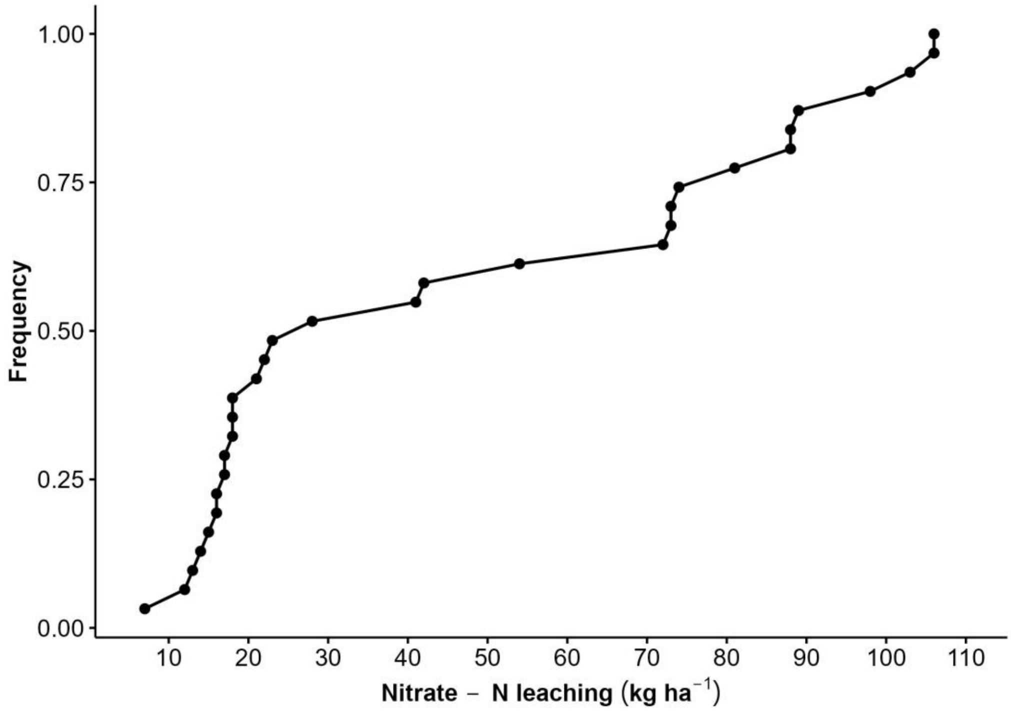

Figure 4.28. Frequency distribution of simulated cumulative annual nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 100 cm soil depth for years 1991 to 2021 in a potato-fallow-potato-fallow-onion-ryegrass-onion-ryegrass rotation.

Scenario 1. Change in soil type

When the soil type in the simulation was changed from Waitohu silt loam to the Shannon silt loam soil, which is poorly drained, the long-term mean annual nitrate-N leaching losses decreased by approximately 7.5% (Table 4.20).

Table 4.20. Cumulative annual nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 100 cm soil depth for years 1991 to 2021 under a potato-fallow-potato-fallow-onion-ryegrass rotation in a Shannon silt loam soil.

Rotation	**Max nitrate-N leaching losses (kg ha^{-1})**	**Min nitrate-N leaching losses (kg ha^{-1})**	**Median nitrate-N leaching losses (kg ha^{-1})**	**Mean nitrate-N leaching losses (kg ha^{-1})**
Potato-fallow-potato-fallow-onion-ryegrass	106	10	28	44

Scenario 2. Potato harvest at maturity and ryegrass cultivation

According to APSIM, much of the leaching losses occurred during the winter of the potato growing years, when the soil is bare, potato tops have died, and potato tubers are ground-stored. If the harvesting time is changed to (approximately) March every year, and ryegrass is grown until the next sowing, the mean annual nitrate-N leaching losses are reduced by approximately 47% (Figure 4.29).

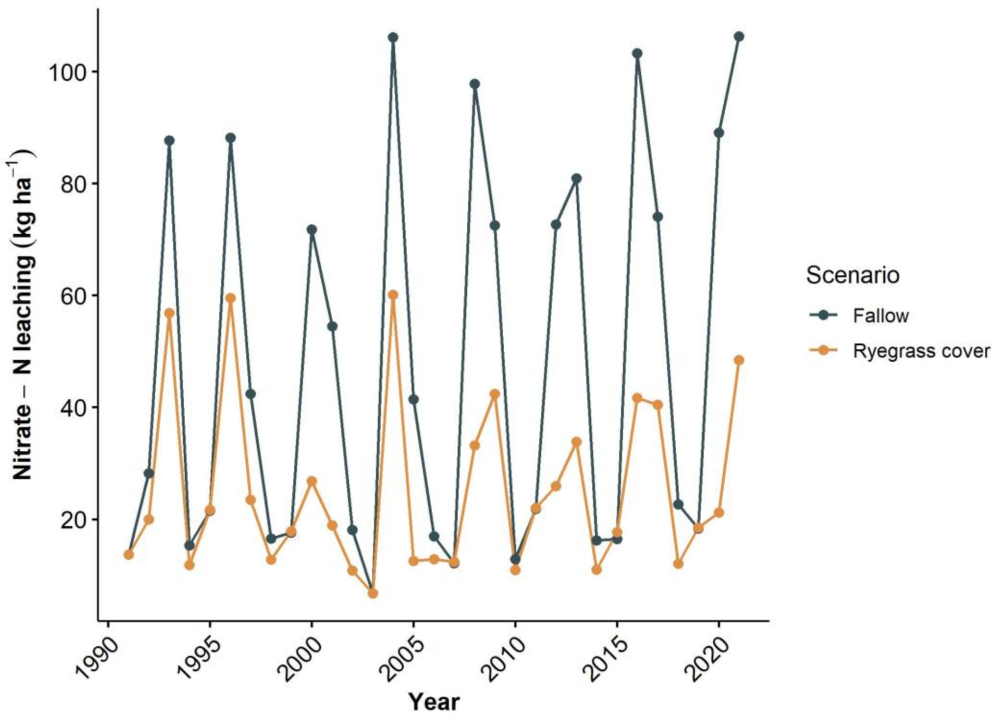

Figure 4.29. Annual nitrate-N leaching losses (kg nitrate-N ha^{-1} yr^{-1}) at 100 cm soil depth for years 1991 to 2021 with a bare fallow period (potato-fallow-potato-fallow-onion-ryegrass-onion-ryegrass rotation), and a with ryegrass cover (potato-ryegrass-potato-ryegrass-onion-ryegrass-onion-ryegrass rotation) during winter-spring.

Scenario 3. Potato harvest at maturity and winter crops cultivation

If ryegrass was replaced by beetroot or fodder beet over the winter period, APSIM predicted that, relative to the bare fallow, mean cumulative annual leaching losses would be reduced by 7% and 17%, respectively (Figure 4.30, Table 4.21).

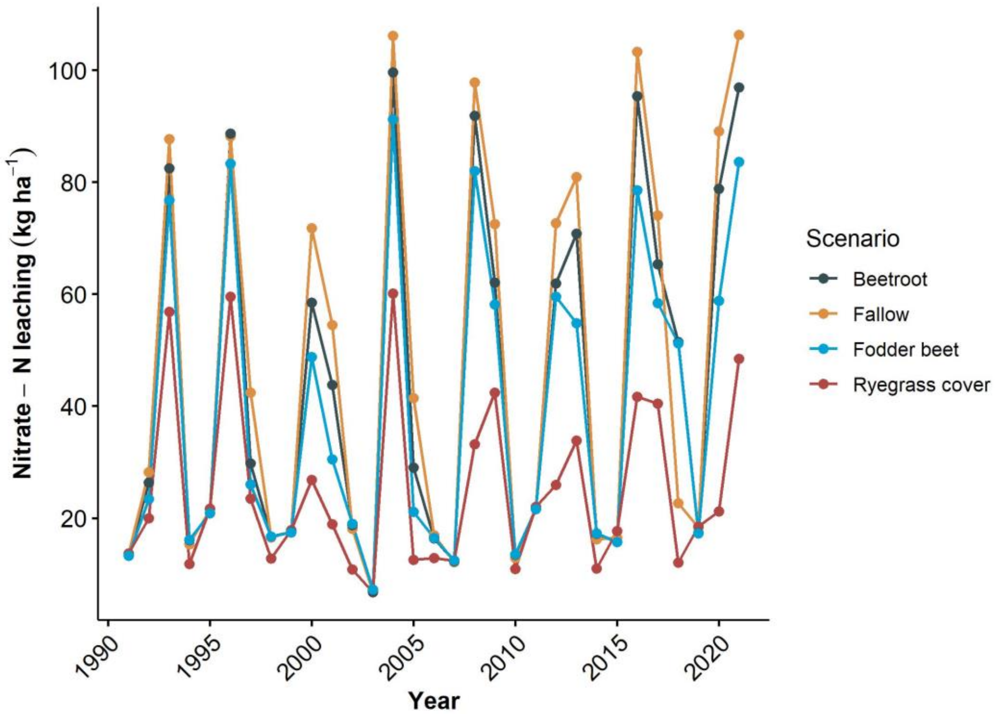

Figure 4.30. Annual nitrate-N leaching losses (kg nitrate-N ha^{-1} yr^{-1}) at 100 cm soil depth for the years 1991 to 2021 with scenarios bare fallow period (potato-fallow-potato-fallow-onion-ryegrass-onion-ryegrass rotation), beetroot cropping (potato-beetroot-potato-beetroot-onion-ryegrass-onion-ryegrass rotation), fodder beet cropping (potato-fodder beet-potato-fodder beet-onion-ryegrass-onion-ryegrass rotation), and ryegrass cover (potato-ryegrass-potato-ryegrass-onion-ryegrass-onion-ryegrass rotation) during winter-spring.

Table 4.21. Cumulative annual nitrate-N leaching losses (kg nitrate-N ha^{-1} yr^{-1}) with different catch crops cultivated after potato harvest from years 1991 to 2021.

Catch crop	Max nitrate-N leaching losses (kg ha^{-1})	Min nitrate-N leaching losses (kg ha^{-1})	Median nitrate-N leaching losses (kg ha^{-1})	Mean nitrate-N leaching losses (kg ha^{-1})
Bare soil fallow	106	7	28	47
Ryegrass	60	7	20	25
Beetroot	100	7	29	44
Fodder beet	91	7	23	39

4.4.4.2 Green vegetable crops rotation

If the initial soil N concentration measured in the lysimeters are used in a long-term simulation, as the initial and re-set values, along with the field soil parameters (Tables 4.5 and 4.6), the mean annual nitrate-N leaching rates are 131 kg ha^{-1}. Maximum, median and minimum annual nitrate-N leaching values for this scenario were 185, 137 and 61 kg ha^{-1}, respectively. If more typical values for initial soil N are used for initial and re-et values, then half of the years in the long-term simulation have equal or less than 79 kg nitrate-N ha^{-1} of leaching losses per annum under beetroot-ryegrass-Pak choi-ryegrass cultivated on a Shannon silt loam soil (Figure 4.31). The maximum value was 117 kg nitrate-N ha^{-1} in 2019 and the minimum was 43 kg nitrate-N ha^{-1} in 2005. Mean annual nitrate-N leaching losses for this site were 79 kg ha^{-1}.

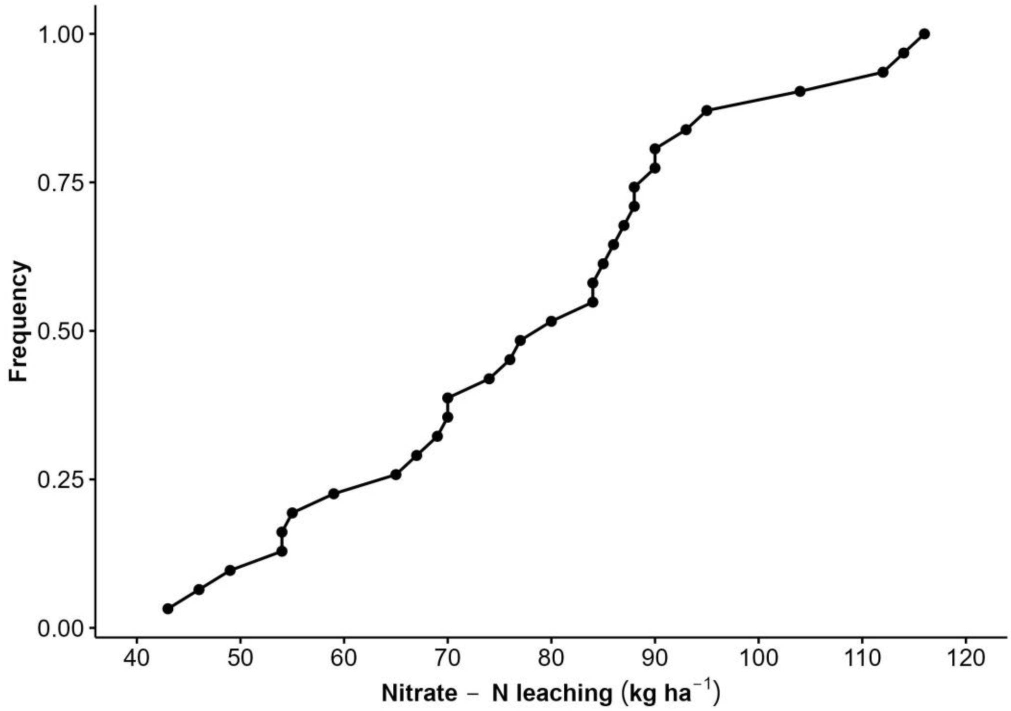

Figure 4.31. Frequency distribution of simulated cumulative annual nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 100 cm soil depth for years 1991 to 2021 in a beetroot-ryegrass-Pak choi-ryegrass rotation.

Scenario 1. Change in soil type

When changing the soil type in this simulation from a Shannon silt loam to a Waitohu silt loam, which is moderately well drained, long-term mean cumulative annual nitrate-N leaching losses decreased by 9% (Table 4.22).

Table 4.22. Cumulative annual nitrate-N leaching losses (kg nitrate-N ha^{-1} yr^{-1}) from the farmers rotation in a Waitohu silt loam soil.

Rotation	Max nitrate-N leaching losses (kg ha^{-1})	Min nitrate-N leaching losses (kg ha^{-1})	Median nitrate-N leaching losses (kg ha^{-1})	Mean nitrate-N leaching losses (kg ha^{-1})
Beetroot-ryegrass-Pak choi-ryegrass	110	39	71	72

Scenario 2. Potential of drainage management

When a tile drain is simulated with 70 and 90% of drainage water interception, mean cumulative annual nitrate-N losses at 100 cm soil depth are reduced by 45 and 52%, respectively (Figure 4.32, Table 4.23). However, 54 and 60 kg nitrate-N ha^{-1} are lost via the drainage systems and would need an edge-of-field mitigation practice.

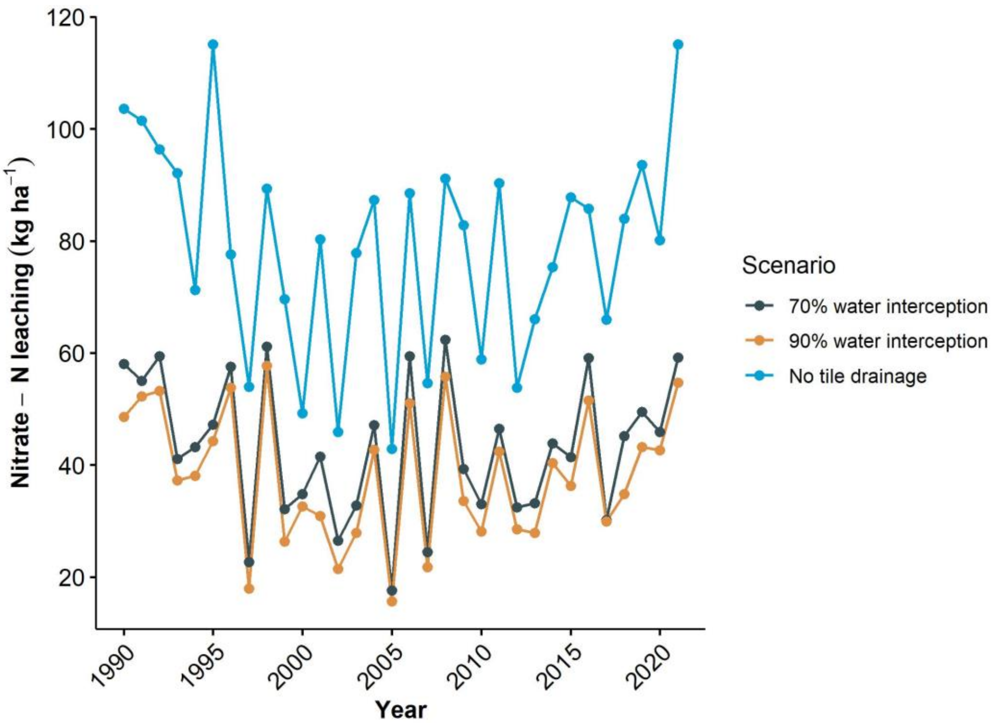

Figure 4.32. Cumulative nitrate-N leaching losses (kg nitrate-N ha^{-1} yr^{-1}) at 100 cm soil depth for 0, 70 and 90% of drainage water interception.

Table 4.23. Mean cumulative annual nitrate-N leaching losses (kg nitrate-N ha^{-1}) with 0, 70 and 90% of drainage water interception by a tile drain system.

Rotation	**No tile drainage**	**70% of drainage water interception**		**90% of drainage water interception**	
	Mean annual nitrate-N flux at 100 cm (kg ha^{-1})	Mean annual nitrate-N flux at 100 cm (kg ha^{-1})	Mean annual nitrate-N flux in tile drain (kg ha^{-1})	Mean annual nitrate-N flux at 100 cm (kg ha^{-1})	Mean annual nitrate-N flux in tile drain (kg ha^{-1})
Beetroot-ryegrass-Pak choi-ryegrass	79	43	54	38	60

Scenario 3. Other alternative cover crops and catch crops instead of ryegrass

According to APSIM simulations, there were no important variations in mean annual nitrate-N leaching losses between the use of different cover/catch crops (Table 4.24). The most efficient catch crop was mustard for seed, with a 6% reduction of nitrate-N leaching losses when compared to the base scenario (i.e., ryegrass) (Figure 4.31).

Table 4.24. Simulated nitrate-N leaching losses (kg nitrate-N ha^{-1} yr^{-1}) from the farmer's rotation incorporating different cover crops.

Crop type	Cover/catch crop	Max nitrate-N leaching losses (kg ha^{-1})	Min nitrate-N leaching losses (kg ha^{-1})	Median nitrate-N leaching losses (kg ha^{-1})	Mean nitrate-N leaching losses (kg ha^{-1})
Grass	Normal	116	43	80	78
	Early	112	36	77	74
Broadleaf	Chicory	111	38	79	75
	Fodder beet	126	45	83	81
	Lettuce	127	44	82	82
	Mustard for seed	106	36	77	73
	Mustard for forage	126	43	81	80
	Plantain	119	43	81	78
	Radish for seed	115	42	82	77
	Rapeseed	114	38	79	75
	Turnip	114	41	81	76
	Winter broccoli	124	43	81	81
	Winter cauliflower	126	44	82	82
	Winter cabbage	126	44	82	82
Cereals	Barley	125	44	82	81
	Oats	126	44	82	82
	Rye	126	44	82	82
	Triticale	126	44	82	82
	Wheat	125	44	82	81

Scenario 4. "In-season" versus "out-of-season" production

Despite the differences in soil type between both simulations, both models showed similar trends when comparing "out-of-season" and "in-season" scenarios, with lower nitrate-N leaching losses under "out-of-season" simulations (32% and 45% reduction under broccoli and cauliflower scenarios when modelled in Overseer, and 31% and 38% reduction under broccoli and cauliflower scenarios when modelled in APSIM) (Table 4.25).

Table 4.25. Simulated mean cumulative annual nitrate-N leaching losses (kg nitrate-N ha^{-1}) for years 1991 to 2021 in different crop rotation scenarios.

Crop rotation	Scenario	Simulated mean cumulative annual nitrate-N leaching losses at 100 cm depth in a Shannon silt loam soil (kg ha^{-1})	Overseer simulated mean annual nitrate-N leaching losses in a Waitohu silt loam soil (Stout, 2021) (kg ha^{-1})
Broccoli-cabbage	Out-of-season	64	42
Broccoli-lettuce		46	65
Cabbage-broccoli	In-season	84	72
Lettuce-broccoli		75	79
Cauliflower-cabbage	Out-of-season	81	60
Cauliflower-lettuce		62	45
Cabbage-cauliflower	In-season	123	89
Lettuce-cauliflower		105	89

Scenario 5. Alternative crop rotations

From the nine representative rotations (Table 4.9), rotation 9 was the one that had the least mean cumulative annual nitrate-N leaching losses, with 41 kg ha^{-1} yr^{-1} (Figure 4.33).

This meant a reduction of 48% compared to the base green vegetable crop rotation scenario (Figure 4.31). The rotation with the greatest mean cumulative annual nitrate-N leaching losses was rotation 2 with 117 kg ha^{-1} yr^{-1}. Most rotations had losses between 80 and 100 kg nitrate-N ha^{-1}, with an overall mean of 79 kg nitrate-N ha^{-1} yr^{-1} (standard deviation of 25 kg nitrate-N ha^{-1} yr^{-1}).

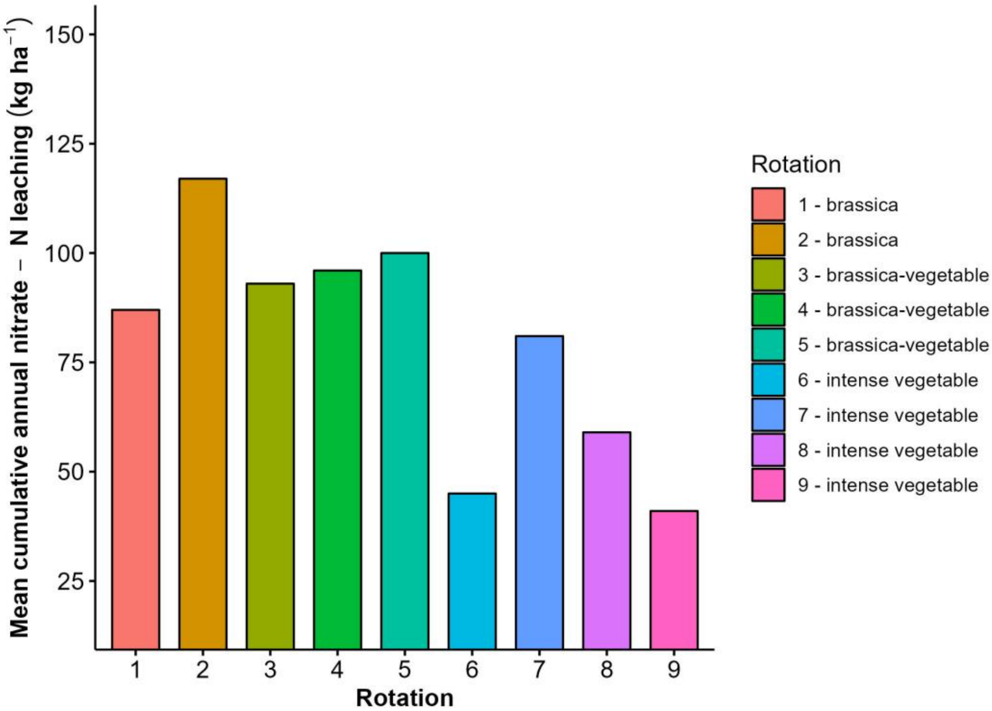

Figure 4.33. Simulated mean cumulative annual nitrate-N leaching losses (kg nitrate-N ha^{-1}) for different crop rotations of the farmers of the area from years 1991 to 2021. For details of crop rotations see Table 4.9.

4.5 Discussion

4.5.1 Model calibration and testing

4.5.1.1 Soil moisture

Overall, APSIM was successful in predicting volumetric soil moisture contents across different fertiliser treatments, crop rotations, soil depths and climatic seasons, with the *NSE* values usually greater than 0.35 (Figures 4.2, 4.3, 4.4, 4.5 and 4.18). *NSE* coefficients were normally greater at the Potatoes site than in the Green vegetables site, possibly because there was more detailed and frequent data for the 2-years period of sampling. Similarly, Vogeler et al. (2017) and Vogeler et al. (2019) had excellent agreement between both observed and predicted soil moisture values in a field cultivated with ryegrass-clover in the Otago area, with *NSE* values greater than 0.5 (Vogeler et al., 2019). Other research in New Zealand that have reported soil moisture content have also indicated a positive performance of APSIM, with *NSE* values often between 0.7 and 0.9 (Teixeira et al., 2018; Trolove et al., 2019b)

In their study of pasture, Snow et al. (2007) tested the ability of APSIM to predict soil water deficit, drainage, and runoff volumes using the SWIM module in a silt loam soil from the Manawatu region. A comparison of the simulated values with three databases revealed that APSIM was able to predict these variables well, particularly drainage (Snow et al., 2007).

In a more recent and comprehensive study, Vogeler and Cichota (2018) used measured volumetric and gravimetric soil moisture data from the NZ-NSD and SMAP databases for soils of the Waikato, Canterbury, and Otago regions to compare with APSIM's soil moisture predictions. The SoilWater module had correlation coefficients higher than 0.6 and relatively good predictive power, with mostly positive or near zero *NSE* coefficients

(Vogeler and Cichota, 2018). Good agreement between predicted and observed soil moisture contents has also been shown extensively in Australian conditions, including long-term data (Keating et al., 2003; Huth et al., 2012). These results demonstrate that APSIM can be used with confidence to estimate the soil water balance across a variety of conditions (i.e., soil types and climate regimes). In addition, in concordance with the results for cumulative drainage measured in Levin, published dataset also show the large variability of individual lysimeters and the difficulties in interpreting these results (Table 3.38, Chapter 3). The level of agreement between measurements and predictions of soil water variables found in this work is to those studies cited above and provide support for the use of APSIM to help interpret measured the data and to extrapolate results via scenario analyses.

4.5.1.2 Soil nitrate-N concentrations and yields

Generally, soil nitrate-N concentrations and nitrate-N leaching losses are difficult to predict due to their dynamic nature, large variability, and dependency on a range of factors; both in space and time. Furthermore, measuring nitrogen in soil and drainage can be technically challenging, and requires a large sample size to be representative. Therefore, it is not measured often in many studies, or may only include a small sample size. Numerous studies using different models have emphasised the difficulties in modelling nitrate-N transport in the soil and water (Hu et al., 2006; Huth et al., 2012; Yang et al., 2014; Liang et al., 2015; Roelens, 2018; Liang et al., 2020Alderkamp et al., 2022). For instance, Kersebaum et al. (2007) compared the performance of 10 models to predict mineral N in different soils of Germany, for which they obtained positive *NSE* values in only 3 of them. The natural variability is accentuated in vegetable crop farms, where soil is constantly subjected to tillage, non-uniform fertiliser application and other management actions that further change soil conditions. However, one of the advantages of modelling is that it provides a general picture of conditions over time and in an integrative manner; this can be very useful to help with management decisions and environmental planning. In comparison, while measurements of parameters such as soil nitrate-N may be more reliable, they often only represent a snapshot of a situation with specific conditions, a static take on a system that is variable and dynamic.

For most of the comparison made here, APSIM predicted nitrate-N concentrations in a ‘satisfactory’ manner (see sections 4.3.1.4, 4.3.1.5, 4.4.2.2, and 4.4.3.2), and only in a few cases was its performance considered ‘unsatisfactory’ based on the *NSE* coefficient. Still, APSIM’s estimations were visibly reasonable in most cases, following a realistic trend in time, and responding appropriately to N inputs and the weather or management actions.

The specific location and timing of soil sampling in crop beds was found to be highly influential when comparing observed data against predicted data. This is particularly the case in the Potatoes site. Topsoil nitrate-N predictions of APSIM had a poor performance according to the *NSE* coefficient, particularly in the early season, most likely because samples were taken from the top of the ridge, whereas the fertiliser was applied banded next to the potatoes at planting, in the “shoulder” of each bed (Figure 4.34). Nitrate-N would not have been moved uniformly through the soil by the time of the early soil measurements; this would have created more concentrated “pockets” of nitrate-N on one side while the rest of the soil volume had a lower quantity compared to expectations. This explains the large variability in the early measurements of the same sampling date (Figure 4.6) and the fact that the peak of N concentration measurements was later than that predicted by APSIM, which simulates perfectly mixed fertiliser. In this regard, samples taken later on, after the nitrate-N from the fertiliser had been distributed more evenly through the soil, showed less variation, and were considered to be better suited to an evaluation of the model’s performance. Model coefficients improved in fertilised treatments when only the latter samples were considered (after February 2020), but unfertilised treatments remained unchanged. This is most prominent for topsoil samples, as subsoil samples’ *NSE* coefficients did not improve when considering only measurements from February 2020 onwards. APSIM’s capability to predict topsoil nitrate-N concentrations improved considerably in the second potato growing season (Figure 4.8), as samples were taken from different locations in the cultivation ridge.

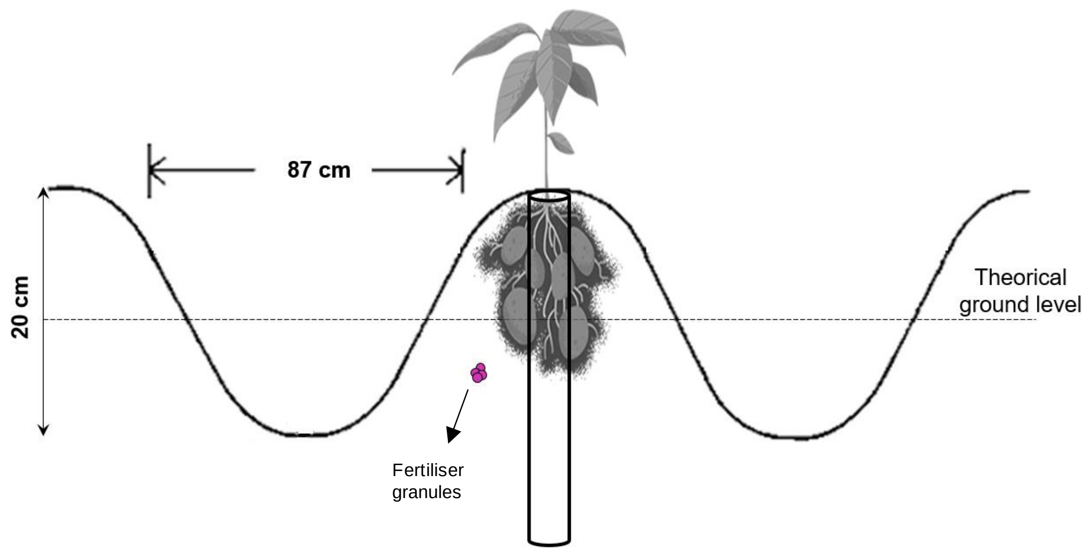

Figure 4.34. Schematic representation of the soil sampling in the first season of the Potatoes site, with fertiliser granules on only one side of the ridge, while sample cores were taken from the middle.

APSIM tended to slightly underpredict nitrate-N concentrations in the C treatment at the Potatoes site. One possible reason that could explain this, is the generally greater potato tuber yields obtained in the simulations compared to the measured data of the C treatment (which would mean a higher simulated N uptake). Several studies have pointed out the implications of lacking N fertilisation on tuber yields, especially during the first stages of the potato growth, which the Potato module from APSIM could perhaps be underestimating (Sparrow and Chapman, 2003; Rens et al., 2018; Zotarelli et al., 2021). Another possible reason is that APSIM could be describing the release of N in small concentrations from mineralisation of organic material unsatisfactorily. Soil nitrate-N in the C treatment were relatively low and stable while APSIM predicted more variation. Nonetheless, as with the other treatments, soil nitrate-N trends over time were predicted by the model with reasonable accuracy.

When applying excess N fertiliser, APSIM's ability to predict nitrate-N concentrations in the subsoil (30-60 cm) was considered unsatisfactory (Figure 4.9). Possible reasons to explain this could be a priming effect triggered by the abundance of mineral N in a localised volume of the soil and greater denitrification rates (van Groenigen et al., 2010). APSIM would not capture these properly as it assumes that the fertiliser nutrients are evenly distributed in the soil volume. APSIM may benefit from further development so that it can better simulate denitrification and the effects of excess nitrate-N in concentrated volumes of the soil and nitrate-N leaching losses from the soil profile. Work has been done for N deposited via urine from grazing animals which has shown that this consideration is an important one (Cichota, Snow, et al., 2013).

Predicted potato tuber yields in the fertilised treatments were similar to the measured yields (Tables 4.16 and 4.17). As the model assumes that N additions are distributed uniformly through soil layers and that the plants can readily use all N in the soil, APSIM did not predict large differences in potato yields between fertilised treatments. In contrast, the measured data shows some difference in the potato yield between the treatments, particularly in situations where the rate of N release and availability differed.

It is quite likely that N fertiliser impacts some of the inputs to APSIM, for example, the number of buds per seed. If different 'numbers of buds per seed' had been used for the C compared with the fertilised treatments to capture the possible effect of nitrogen fertiliser on this parameter, then there would have been excellent agreement between measured and predicted yields for both C and the treatments. However, in the interests of simplicity, there was no further adjustment of the model used here. Further attention should be given to the interactions between N fertiliser and input parameters to APSIM to improve the model's ability to predict potato yield.

In general, the soil nitrate-N predictions of the model at the Green vegetables site had better *NSE* coefficients (Figure 4.25) than at the Potatoes site (Figures 4.6 and 4.8), probably because soil samples were taken after each crop harvest, when the variability associated with recent fertilisation was minimal.

APSIM's soil nitrate-N and leaching losses estimations were found to be particularly sensitive to plant N uptake, and therefore, an appropriate estimation of crop yields is crucial to the performance of the model. This is especially important when using the SCRUM module, which does not use population parameters and relies on the expected yield as an input. In this study, SCRUM was found to largely underpredict yields, and expected yields had to be set to much greater values than their typical expected yields so as to match the measured data from the field.

There is a suggestion that, in some cases, the default mineralisation rates of APSIM may underpredict real mineralisation rates, as also indicated by Sharp et al. (2011a), Sharp et al. (2011b), Vogeler et al. (2019), and Vogeler et al. (2020). For instance, in the Green vegetables site, predicted soil nitrate-N concentrations in November 2021 were considerably lower than the observed values in all treatments. With default parameters, APSIM tended to distribute more N towards the microbial biomass pool and organic matter (i.e., greater immobilisation). Therefore, mineralisation and decomposition rate parameters had to be adjusted with expert assistance (R. Cichota, 2021; personal communication). In addition, the physical state of the residues in APSIM is not considered as an input (i.e., chopped or whole), which can potentially make a difference to decomposition time as the surface area and contact with soil varies with residue form.

Overall, studies in New Zealand rated APSIM's predictive power for soil nitrate-N content as satisfactory, though literature reporting this variable for vegetable crops was not found except for Sharp et al (2011a) and Sharp et al. (2011b). These authors indicated that APSIM underpredicted soil mineral N content in the 0-100 cm of soil, and extensive research should be conducted to adapt APSIM's simulation of mineralisation to temperate climates (Sharp et al., 2011b).

In Western Australia, using the Wheat, SoilNitrogen and SoilWater modules, Asseng et al. (1998) found good agreement between observed and predicted soil nitrate-N concentrations in the 10-30 cm of depth, but a slight overestimation by APSIM in the 0-10 and 30-150 cm. In a similar study in the Netherlands, Asseng et al. (2000) reported an acceptable performance by APSIM in predicting soil mineral N content, despite achieving a somewhat low coefficient of determination (R^2 = 0.46). Archontoulis et al. (2014), in Midwest US conditions, carried out an extensive comparison between several observed and predicted parameters in a 2-year experiment in soils under maize crop, which were fertilised with manure and compost. Similar to the findings reported here for the topsoil of the first potato season, they indicated that APSIM simulated soil mineral N content matched closely the observed data in the first 20 cm of soil. However, they also reported high variability in soil-N concentrations and an initial overestimation that biased the overall model's performance.

Given the results from the comparisons here, which were in line with published work, it can be concluded that the APSIM model is suitable for the very complex task of simulating the N balance in vegetable crop systems. The results here and those from the literature also highlight the need for critical analyses of the results as the data is quite variable and results can be sensitive to parameters difficult to measure as well as to the setup that represents the management in the model.

4.5.1.3 Drainage and nitrate-N leaching fluxes

High confidence can be placed in APSIM's estimations of drainage depth. As previously indicated, very satisfactory agreement was found between measurements and simulations of soil water content in the field with potato-onion rotations. In the Green vegetables site, despite the noticeably large variability of the measured data, APSIM predicted the mean cumulative drainage water volumes reasonably well, with *NSE* values higher than 0.3. This predictive ability is more obvious in treatments with lower variability.

As expected, the measured and simulated drainage and leaching losses at both experimental sites, tended to occur mostly in winter and spring, although intense rainfalls, and consequently drainage, also occurred during December 2020. Intense drainage events during summer have also been reported for this area by Norris et al. (2017). In addition, estimations of cumulative drainage from APSIM were very similar for both field sites, which indicates consistency and reliability in APSIM simulations.

Large within-treatment variability for suction cups measurements led to an overall 'unsatisfactory' to 'satisfactory' rating of APSIM's ability to reproduce nitrate-N concentrations in drainage water. In their study of APSIM's ability to predict solute transport in SWIM3, Huth et al. (2012) have emphasised that, in cases where spatial variation is greater than temporal variation, model indicators such as *NSE* may not be ideal, because of the high variability in observed data. Other authors also agree that, instead of focusing on only one test, several statistic parameters should be used to evaluate the overall model performance, as this would reveal different strengths and weaknesses of the model (Bellocchi et al., 2010; Vogeler & Cichota, 2018).

When simulated drainage volumes are combined with nitrate-N concentrations from suction cups to calculate cumulative nitrate-N losses from the potato trial, there was a 'satisfactory' to very 'satisfactory' agreement between measured and predicted data.

Studies that have used APSIM to estimate nitrate-N leaching losses have reported both satisfactory and unsatisfactory model performance. Sharp et al. (2011a) and Sharp (2011b) monitored nitrate-N concentrations in drainage water from different potato fields using suction cups for three years and found that an older version of APSIM had a generally poor performance in predicting nitrate-N leaching losses (*NSE*= -1.33 to -58.9), and that when a default mixing factor was modified, predictions improved for the first year (*NSE*= 0.64). In contrast, Khaembah et al. (2015), and Khaembah & Brown (2016) found that a newer version of APSIM was highly successful in predicting cumulative nitrate-N leaching losses under a potato-fallow-pea-potato crop rotation (*NSE*= 0.84). Several other studies carried out in soils with pasture and urine depositions, have reported successful predictions by APSIM for cumulative drainage and cumulative nitrate-N leaching losses (Cichota et al., 2010a; Cichota et al., 2010b; Romera et al., 2010).

Outside of New Zealand, Asseng et al. (1998) compared simulated with measured nitrate-N leaching losses in fields cultivated with wheat in Western Australia. They concluded that both values were similar. Stewart et al. (2006), evaluating nitrate leaching losses in fields cultivated with sugarcane in Australia, concluded that APSIM predicted the magnitudes of cumulative nitrate-N leaching losses successfully, but not the variations over time. Finally, Alderkamp et al. (2022), in their study in soils under ley-arable systems in two experimental sites in Denmark and Germany, found that APSIM had a poor to reasonable ability to estimate nitrate-N leaching losses, leading them to suggest that improvements to the simulation of mineralisation are required if the model is to provide better predictions.

In general, cumulative nitrate-N leaching losses simulated in APSIM responded to the increase in N fertiliser rate applied to soils with the potato-onion rotation. Simulated cumulative nitrate-N leaching losses at a 100 cm obtained in the present study (30-65 kg nitrate-N yr^{-1}) were smaller than the 116 to 170 kg N ha^{-1} range reported by O'Brien (2019) who used Overseer for the same area, and they were at the smaller end of the range of 50-300 kg N ha^{-1} reported by Khaembah et al. (2015) who used APSIM to simulate losses under a range of fertiliser rates in other areas of New Zealand. Nitrate-N leaching rates in the potatoes site were between 26-49% of the N applied, suggesting that a significant amount of N fertiliser can be lost as leaching in these areas.

The model suggested that in-field practices that change the way N is delivered, such as controlled release fertiliser and liquid fertiliser, did not reduce substantially nitrate-N leaching losses for a potato-onion rotation. This was most likely because by the start of the drainage season (July) the nitrate-N concentrations in the soil were essentially the same across the different fertiliser treatments. This suggests that these in-field mitigation techniques could be more valuable where the crop is grown during the winter-spring seasons rather than during summer. Mitigation practices for crop rotations that grow only in summer, such as potato-onion rotations, could perhaps focus on managing crop residues and reducing or avoiding fallow periods.

APSIM has also performed well in estimating larger cumulative nitrate-N leaching losses in soils growing the green vegetable crops. However, once again, APSIM tended to underpredict the cumulative nitrate-N leaching losses in the C treatment, which might also be related to the previously mentioned limitations in APSIM's ability to estimate N mineralisation and residue decomposition or its overestimation of plant uptake of N. Differences in soil parameters, management setup, and the profile depth (60 cm versus 100 cm) may also help to explain why estimations of nitrate-N losses were larger when using input data related to the lysimeter trial than the input values related to the field trial.

According to APSIM, much of the nitrate-N leaching losses in soils with vegetable crop rotations were caused by the initial soil nitrate-N concentrations (residual N), and by the incorporation of the whole beetroot crop into the soil later in the season rather than by fertiliser management. The incorporation of the whole crop, which was triggered by factors related to the COVID-19 pandemic, was unusual but it provides evidence that residue management is important and can be of concern for crops with a low harvest index. The impact of residue management is also supported by the relatively large leaching losses values found in the treatment with no fertiliser. Cumulative nitrate-N leaching values found in the lysimeters were similar to those reported previously for the area at 100 cm in fluxmeters (Norris et al., 2017).

4.5.2 Long term simulations and in-field mitigation practices

With the model verified for the conditions and crop rotations of the Arawhata catchment, APSIM was then used to investigate a variety of scenarios for reducing N losses over long-term simulations. Long-term simulations were run to look at the likely range of nitrate-N leaching rates from the different rotations under a range of climatic conditions. As noted, farmers would not practise these rotations continuously over a 31-year period, not least of all, because of the adverse effect on organic matter content. To overcome this particular problem, input parameters in APSIM simulations were reset at the conclusion of each cycle of the rotation to minimise the effects of changes in soil organic matter. APSIM assumes that the soil organic matter is split into three pools with different turnover rates (a 'fast', a 'slow' and an 'inert' fraction), the proportion and the turnover rates of these fractions are input parameters and are seldom measured (in fact, the proportions are not really measurable). The APSIM community typically use default parameters for the turnover rates, with the proportions of each pool being set by experience and trial-and-error, and depending on the scenario being described. These parameters are generally not highly important in short-term runs, but can have considerable impact in the longer-term. In these cases, simulations can be used to find the equilibrium situation in a given combination of crop, climate and soil. Alternatively, simulations can be reset periodically to avoid build-up of bias over time. Defining the time-span at which to reset simulations can be problematic, as nitrate-N fluxes can be affected by the frequency of the reset, therefore, a medium-to-short reset period is usually considered in order to maintain stability over time. In the case of the present work a two-year period was used.

The variability in leaching rates between years in the long-term simulations is mostly associated with changes in rainfall amounts (and the resulting drainage depths), the type of crop grown from the crop rotation (i.e., whether it was a first season potato, second season potato or onion crop), and the amount of crop biomass (N uptake).

4.5.2.1 Potato-onion crop rotation

An interesting result from the long-term simulations was the fact that half of the simulated years (15) had cumulative annual nitrate-N losses less than 25 kg ha^{-1} which indicates that, normally, nitrate-N leaching losses for this site are small. On the other hand, the frequency distribution graph (Figure 4.28) revealed that there is an approximate 25% chance of leaching to exceed 80 kg nitrate-N losses ha^{-1} yr^{-1}. It seems that, both years of the field study were in this upper end of the range due the above average rainfall experienced over this period, particularly during winter and summer. For example, the rainfall experienced between June and September 2021 was 120 mm greater than the long-term average (Chappell, 2015), and the amount of rainfall in December 2021 was more than 2.5 times greater than the long-term average.

When simulating a change in soil type, both in potato-onion rotation and green vegetables crop rotation simulations, there was a small difference compared to the base scenario. This difference was mostly associated with a difference in immobilisation and mineralisation rates in the soil.

The long-term simulated mean cumulative annual nitrate-N leaching loss (47 kg nitrate-N ha^{-1}) was greater than values reported by Jolly (2020) for a potato crop rotation from the same area, but considerably less than the 116-170 kg N ha^{-1} reported by O'Brien (2019), both simulated in Overseer. This difference could be attributed to mechanistic differences between APSIM and Overseer models, including differences in drainage and yield estimations, as suggested by Stout (2021).

APSIM was then used to quantify the reductions of nitrate-N leaching losses under different in-field mitigation scenarios. Harvesting potatoes earlier than usual and growing ryegrass until the next potato growing season, was the most effective scenario investigated here, with an average reduction in annual nitrate-N leaching of 47%. This average reduction can be subjected to high variability, and therefore, must not be taken as a fixed value. Replacing the fallow period with ryegrass reduced the quantity of nitrate-N at risk to leaching.

APSIM was found to be a useful tool to describe the variations in N losses over time. It can provide insights to when and under which circumstances the majority of nitrate-N losses are likely to occur. It can be a helpful model to estimate mean cumulative annual nitrate-N leaching losses under different scenarios and test variations in management practices under different environmental conditions.

4.5.2.2 Green vegetables crop rotations

In the Green vegetables site, mean and median cumulative annual nitrate-N losses over the longer-term were higher than in the Potatoes site, despite having lower fertilisation rates. Possible explanations to this include the fact that potatoes have typically longer growing periods (5-6 months) than green vegetable crops (3-4 months) and have an overall greater N uptake. Furthermore, green vegetable crops tend to have shallow roots, and high amounts of fast decomposing residues, which tend to leave high levels of residual N in the soil.

In terms of in-field mitigation practices to decrease nitrate-N leaching losses, there was little scope to improve the current situation, as the farmer is already using relatively low N fertilisation rates and cover crops. For example, in the cover crop analysis, there was no substantial difference in N reductions when including the most efficient cover crop in the crop rotation compared to the typical rotation (Table 4.24). This could be explained by a similar biomass production between different catch/cover crops in-between seasons, which implies similar rates of N uptake. APSIM's ryegrass model in SCRUM was found to be the most efficient in intercepting leaching losses, however, field trials should be conducted to further evaluate differences between various cover crop options.

One of the best ways found in the simulated scenarios to reduce leaching losses in this field was to intercept drainage water with artificial drains for further edge-of-field mitigation techniques. This was represented in APSIM by a relatively simple approach whereby a percentage of drainage water and its associated nitrate-N was captured at 60 cm soil depth. However, these results are largely exploratory for scoping the potential for edge of field mitigation (i.e., what is the best case). The selection of values of 70 and 90% drainage captured and 100% reduction in this captured drainage are somewhat arbitrary. The exact values here are not so important, as the point of this exercise was to assess the likely potential of edge of field techniques to reduce N leaching (i.e., by the simulated 45 to 52%). As noted, if drainage water can be captured and treated at the edge of the field then drainage water quality could be improved substantially.

One of the disadvantages found when using APSIM to simulate in-field mitigation practices was it is not possible to combine crops in one single field, neither represent in much detail the interactions between them. Hopefully, this will be included in future updates of the model, as such complex systems and intercropping are common on vegetable farms and software to model them is needed. Examples of these complex systems where APSIM can be limited will be further analysed in Chapter 6.

In order to compare APSIM's nitrate-N leaching results with Overseer, a widely used farm-scale model in New Zealand, brassica crop rotation simulations conducted by Stout (2021) were replicated in APSIM using the Shannon silt loam series and evaluated in long-term simulations. Results indicated that APSIM and Overseer had relatively similar mean cumulative annual nitrate-N leaching losses, with an absolute mean difference of 18 kg nitrate-N ha^{-1}. In general, APSIM estimated greater nitrate-N leaching losses, particularly in scenarios involving cauliflower instead of broccoli. This difference, besides the difference in soil type modelled, could well be due to the yield estimate difference between the two models, as indicated by Stout (2021).

Despite this difference, both models indicated a similar trend when cultivating crops "in-season" versus "out-of-season". According to Stout (2021), the difference between both scenarios in nitrate-N leaching could be explained by greater rates of soil inorganic N during the drainage season (winter) under the "in-season" scenario. These larger soil inorganic N rates were associated with greater amounts of N from crop residues in the "in-season" production of broccoli and cauliflower, driven by larger yields. Further elaboration about comparing these two models and their mechanisms is out of the scope of this study, but the interested reader can refer to documents such as Stout (2021), Khaembah & Brown (2016), Anastasiadis et al. (2013) and Vibart et al. (2015). Therefore, as noted by Stout (2021), there seems to be little merit to the claim that growing crops out of season increases the risk of N leaching.

Other crop rotations that were deemed representative from farmers of the area were also simulated in APSIM using the Shannon silt loam soil over 31 years (1991-2021). Expected yields were set as recommended by literature and guidelines, however, as seen previously, APSIM-SCRUM tends to underestimate simulated yields and, in some cases, above-normal expected yields must be set as an input to estimate realistic yields and N uptake rates. Some rotations that included certain crops, such as spinach or lettuce, had small simulated harvested yield in APSIM, and thus resulted in greater simulated nitrate-N leaching losses. With this in mind, the results of these simulations are best used for comparative purposes and are likely to vary from region to region.

According to APSIM, representative rotations that were strongly based on brassica cultivation tended to produce more nitrate-N leaching losses than other intensive vegetable rotations, possibly because brassica rotations often need greater amounts of N fertiliser, and do not have a cover crop between growing seasons. In contrast, rotations that have a ryegrass cover crop every year and include crops with low inputs of N will produce lower N leaching amounts (maximum reduction of 48%). An interesting result is that rotation 9 (Table 4.33), though somewhat similar to the rotation used in present study, had smaller N losses, which is likely due to shorter fallow periods and greater biomass, and thus higher N removal by the Pak choi crop.

Based on this set of results from APSIM, it can be emphasised the importance of managing residues, perhaps by taking the residues' N content into consideration when determining fertilisation levels. Reducing the duration of fallow periods is another important factor to mitigate nitrate-N leaching losses under vegetable productions systems.

4.6 Conclusion

APSIM Next Generation successfully predicted soil moisture and drainage in a variety of conditions, such as soil type, crop rotations and climate regimes, in two sites in the Arawhata catchment. However, when using the *NSE* coefficient to measure goodness-of-fit between simulated and observed nitrate-N concentrations in soil and drainage water, the model had difficulties in matching the observed values. These difficulties were mostly associated with the high spatial variability of the data and, possibly, to the inaccurate representation of mineralisation rates in vegetable crop systems. Despite this, APSIM could accurately predict cumulative (i.e., annual) nitrate-N leaching losses.

Overall, the APSIM model was useful to identify the drainage (and leaching) season and provided key insights into long-term leaching losses. APSIM also was able to depict the effect of different climatic regimes on nitrate-N leaching losses in long-term simulations. It revealed that the conditions in which the measurements for this study were made were wetter than usual, and that in 50% of the years, the expected N-nitrate leaching losses were less than 25 kg N ha^{-1} yr^{-1} and 79 kg N ha^{-1} yr^{-1} in a potato-onion and green vegetable fields, respectively.

Furthermore, after calibration and validation, APSIM was also helpful in quantifying differences in nitrate-N leaching losses under a suite of in-field mitigation practices assessed in the long-term. According to APSIM, nitrate-N leaching reductions from these scenarios ranged from 6 to 52% compared to the normal practice of growers of the area. The reductions presented here can be subjected to variability, and must not be taken as a fixed value. However, based on this set of results from APSIM, we can emphasise the importance of managing residues, perhaps by taking the residues' N content into consideration when determining fertilisation levels. Reducing the duration of fallow periods is another important factor to mitigate nitrate-N leaching losses under vegetable productions systems.

APSIM was then used to quantify the reductions of nitrate-N leaching losses under different in-field mitigation scenarios. Harvesting potatoes earlier than usual and growing ryegrass until the next potato growing season, was the most effective scenario investigated here for a potato-onion rotation fields, with an average reduction in annual nitrate-N leaching of 47%. Replacing the fallow period with ryegrass reduced the quantity of nitrate-N at risk to leaching. For a green vegetable crops rotation, the use of rotations that have a ryegrass cover crop every year and include crops with low inputs of N was one of the most effective scenarios investigated here, with an average reduction in annual nitrate-N leaching of 48%.

Chapter 5: Up-scaling evaluation of in-field practices on nitrate losses reduction in intensive vegetable production catchments

5.1 Abstract

Intensification of land use has severely affected the water quality of Lake Horowhenua in the Manawatu-Whanganui region of New Zealand. Its main tributary, the Arawhata stream, is surrounded by intensive vegetable crop farms which have been described to largely influence in the lake's water quality. In the present study, the main flow and nutrient pathways were identified in the Arawhata catchment. This was achieved by both, a field characterisation of the nitrate-N concentration and flow rate in the drainage network of the catchment, and the coupling of field-scale models, APSIM Next Generation and Overseer, and a catchment-scale model, SWAT+. Flow rates and nitrate-N loads predicted by the coupled models were compared against observed values, and goodness-of-fit was evaluated using *NSE* and R^2 coefficients. Due to uncertainty in the crop rotation pattern in areas with green vegetable crops, two scenarios were assumed: 'Model 1': assumes that the vegetable rotations containing brassica crops generate 65% of the nitrate-N leaching losses of the vegetable rotations, and 'Model 2': assumes that the vegetable rotations containing brassica crops generate 80% of the nitrate-N leaching losses of the vegetable rotations. Furthermore, after calibration and model evaluation, five scenarios with different in-field practices to reduce nitrate-N leaching were evaluated using the APSIM-Overseer-SWAT+ model. These scenarios included: a) Scenario 1: a ryegrass cover crop is grown in fallow periods of the potato-onion rotations, b) Scenario 2: brassica rotations were replaced by green vegetable crops rotations with small nitrate-N leaching rates, c) Scenario 3: combination of scenarios 1 and 2, d) a decrease of 10% in the quantity of N fertiliser applied to all crops and e) a decrease of 20% in the quantity of N fertiliser applied to all crops. Measured flow rates in the sampling sites of the Arawhata catchment ranged from 0.002 to 251.825 L s^{-1}, and nitrate-N concentrations ranged from 1.27 to 8.29 g m^{-3}. The coupled APSIM Next Generation-Overseer-SWAT+

models had a *NSE* and R^2 of 0.68 and 0.74 for predicting monthly stream flow of the Arawhata stream, respectively. Predictions of monthly nitrate-N loads had *NSE* values between 0.68 and 0.71, and R^2 values of 0.7 and 0.73, respectively. These parameters indicated that the coupled models were successful in predicting both stream flow and nitrate-N loads. Nitrate-N attenuation factors estimated using the SWAT+ model predictions indicated that the Arawhata catchment has a low capacity to reduce nitrate-N loads, which was associated with the low residence time of the nitrate-N in the groundwater. From the simulated scenarios, a change in crop rotations that are less oriented to cultivating brassicas showed to be the most effective single practice among those evaluated in this study, with reductions between 17 – 21% of mean annual nitrate-N load in Arawhata stream.

5.2 Introduction

The wellbeing of rivers and lakes in Aotearoa (New Zealand) has been key to societies since its foundations, being considered as ‘the blood ways of Papatūānuku (mother earth)’ by Māori people (LEARNZ, 2020). Productive farming systems are key for New Zealand to ensure the nation’s food security, and to social and economic wellbeing (Ball, 2020; Soliman & Greenhalgh, 2020). However, intensive farming activities, including vegetable production, have been identified as a major source of nutrients (nitrogen and phosphorus) runoff affecting quality of receiving freshwaters. In fact, nutrient losses to waterways remains one of the top priorities of New Zealand’s Ministry for the Environment and Regional Councils (Cosgrove, 2019; Ministry for the Environment and Stats NZ, 2022), as reflected in the recently revised National Policy Statement for Freshwater Management 2020 (Ministry for the Environment & New Zealand Government, 2022) and Essential Freshwater Package (Ministry for the Environment & Ministry for Primary Industries, 2020). Therefore, a sound understanding and management of nutrient losses from farming systems is very important to maintain a healthy ecosystem and biodiversity in Aotearoa’s environment.

Lake Horowhenua, and its most important tributary, Arawhata stream, have been facing eutrophication, despite consistent efforts to ameliorate it (Roygard et al., 2015). Studies have found that nutrient enrichment has been associated with nitrogen (N) leaching from surrounding horticulture, market gardening and intensive dairy farming (Gibbs, 2011; Thomas & Gibbs, 2014). In this context, identification of targeted and effective nutrient management and mitigation practices is essential to reduce nutrient losses and improve water quality in the Lake Horowhenua (Land and Water Forum, 2010; MGM Governance Group, 2015; Burbery et al., 2020).

Field observations to identify and map critical water and nutrient losses pathways is key to develop appropriate management and mitigation practices in agricultural landscapes. The use of farm-scale nutrient losses and catchment-scale hydrologic models can be a useful tool to simulate the impacts of agricultural practices on spatial and temporal dynamics of water flow pathways and nutrient losses to receiving streams, rivers and lakes (Legarth et al., 2022; Singh et al., 2017). The APSIM model has been demonstrated to accurately quantify the amount of water that flows into the soil profile, simulate soil-water-nutrient interactions, and be able to predict nutrient losses from agricultural fields (Chapter 4). However, field-scale models, such as APSIM and Overseer, are limited in simulating water and nutrient flows at catchment-scale. Coupling such field-scale models into catchment-scale models, such as SWAT+, could help to quantify potential impacts of in-field management practices on nutrient losses at catchment-scale.

SWAT is an eco-hydrologic catchment-scale model that is the product of more than 45 years of development efforts (Bieger et al., 2019). Previous studies have shown that SWAT, and its newly restructured version, SWAT+, models can be successful in simulating nutrient losses in catchments with predominantly agricultural land use (Akhavan et al., 2010; Halecki & Bedla, 2022). SWAT+ has been designed to allow more flexibility to the user, opening the possibility of coupling external estimations of nutrient losses into the mode, however, this has not been yet tested. Although having been tested extensively globally, only a few research articles have been published regarding SWAT's ability to model flow and nutrient load of New Zealand's rivers (Hoang, 2019), and none has yet used the newly developed SWAT+ version.

The main objectives of this study are to (1) analyse water and nutrient flow pathways in the Arawhata catchment; (2) to develop a model that can predict N loads to the Arawhata stream from the farms in the catchment. This overarching model will be a combination or coupling of models (APSIM and Overseer) capable of providing good estimates of N leaching from soils with a model that simulates hydrological processes at the catchment scale (SWAT+ model), 3) to calibrate and validate the coupled APSIM-Overseer and SWAT+ models for nitrate losses in the Arawhata catchment; and (4) apply the calibrated and validated APSIM-Overseer and SWAT+ models to assess the performance of a range of in-field strategies to reduce nitrate-N leaching from intensive vegetable farming in the Arawhata catchment. The study aims to up-scale, to the catchment level, the evaluation of in-field mitigations to reduce N leaching losses under intensive vegetable production. This will facilitate the process of identification and development of targeted and effective nutrient management and mitigation practices to help improve water quality in Lake Horowhenua and similar catchments in New Zealand and other countries.

5.3 Materials and Methods

5.3.1 Study area

The Arawhata catchment covers an area of 1,714 ha located between Levin and Ohau, in the Manawatu-Whanganui region of New Zealand. The catchment has several drains and streams flowing north-westwards and it is hydrologically defined by the Arawhata stream at its lowest point in elevation (Figure 5.1). The Arawhata stream is one of the main inlets of Lake Horowhenua (Gibbs, 2011). The Arawhata stream and Lake Horowhenua have shown high concentrations of nitrate-N in recent decades, making the Lake Horowhenua amongst the most nutrient-rich in the country, and rising major concern in the community (Gibbs, 2011, Roygard, 2015).

The average annual rainfall in Levin is 1163 mm and the months of June-October tend to be the wettest (Table 5.1) (Chappell, 2015). However, large rainfall events and wet days can also be expected during December. The catchment has two weather stations recording daily climatic variables (Figure 5.1), including those necessary for parameterisation of the catchment analysis.

Table 5.1. Monthly average rainfall amounts (a), percentage of total annual per month (b), monthly average number of rain days (c) and wet days (d) observed in Levin (Chappell, 2015).

Weather station		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Levin, Winchester St	a (mm)	69	85	114	83	90	114	106	91	99	111	99	104
	b (%)	6	7	10	7	8	10	9	8	8	10	9	9
Levin AWS	c (0.1 mm rain day)	11	10	11	12	14	18	18	18	17	16	15	16
	d (1 mm wet day)	8	7	8	8	10	12	12	12	13	12	10	11

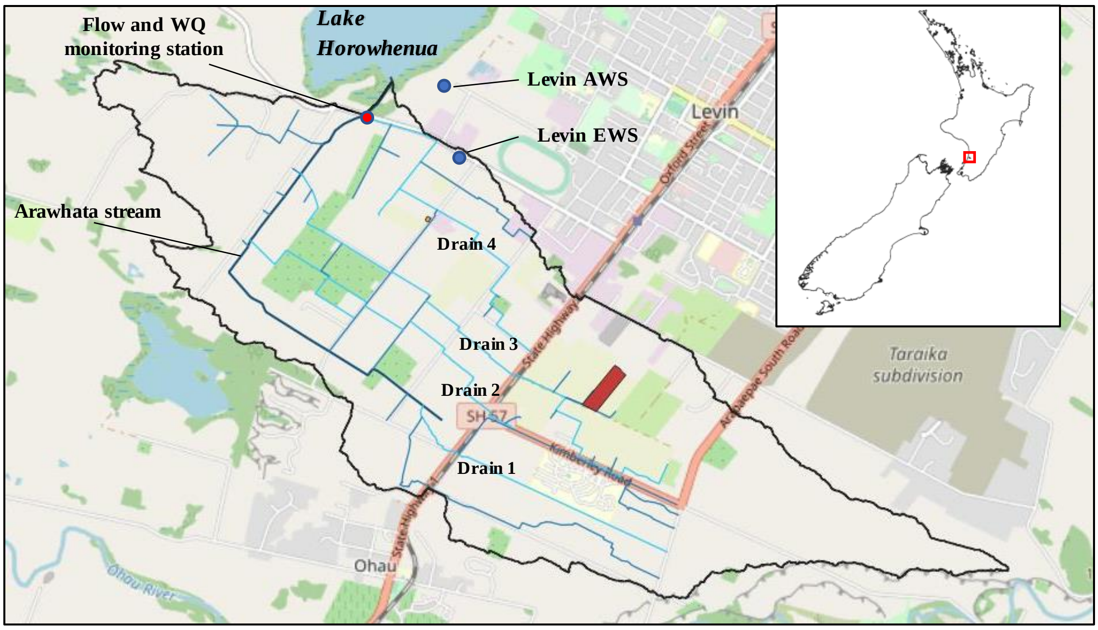

Figure 5.1. Arawhata catchment complex (black line for catchment boundary, blue lines for drains and culverts, light blue for main drains, dark blue line for Arawhata stream).

The main land-uses in the Arawhata catchment include pastures grazed by dairy cows (34%), outdoor vegetable crop farms (29%), and grasslands –within lifestyle blocks and vegetable farms– (28%) (Figure 5.5) (Horizons Regional Council, personal communication, September 2021). The landscape in the Arawhata catchment is mainly flat (0-3°) and soils in the area are heterogeneous. Commonly, soils are derived from loess, moderately deep to deep, with silt loam texture and moderate to poor drainage (Chapter 3). Due to constant ploughing and cultivation, the first soil layers (<40 cm) are generally permeable, while deeper soil layers (at approximately 40 cm depth) are compacted, causing a perched water table in the soil profile for much of the year (A. Palmer, personal communication, May 2022). This surplus water slowly permeates into deeper groundwater and laterally into the stream and lake (Figure 5.2) (Thomas & Gibbs, 2014). As a consequence, baseflow of the Arawhata stream remains high and surface runoff contributions to streamflow are generally limited to high rainfall events.

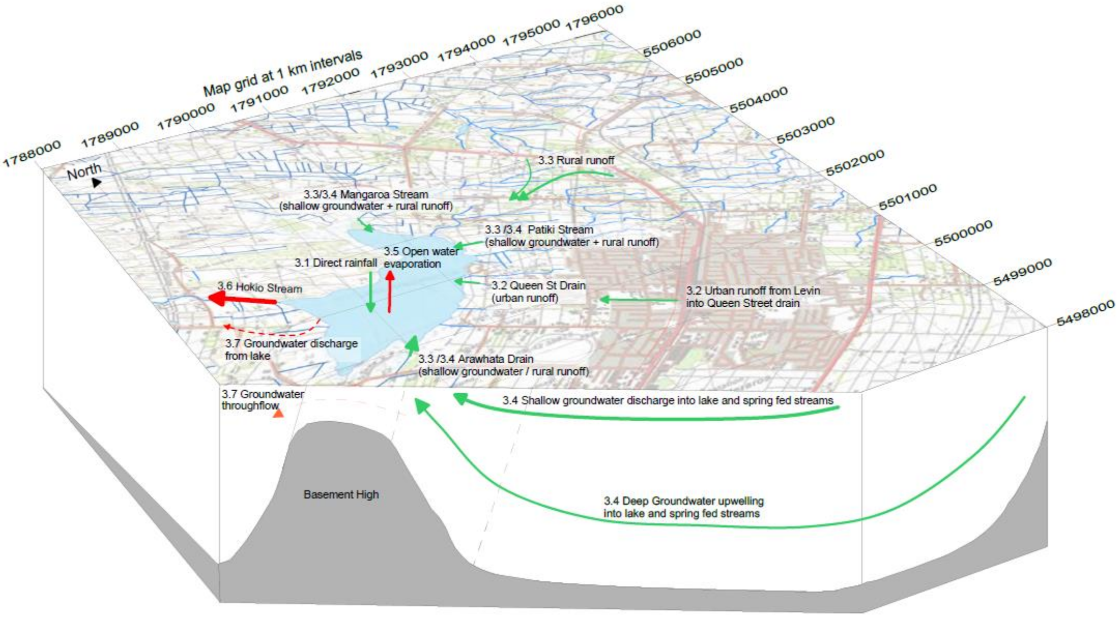

Figure 5.2. Arawhata catchment water balance (all units in mm) (Thomas & Gibbs, 2014)

Flow discharge and nutrient concentrations in the Arawhata stream outflow (Figure 5.1) have been continuously monitored by Horizons Regional Council since July 2017. Streamflow discharge has been estimated by using hourly observations of the stream stage, and a stage-discharge relationship. Nitrate-N concentrations have also been monitored approximately once a month using grab samples from the same gauging station from 1998 to 2022 (Figure 5.1). However, only concentration data monitored by Horizons Regional Council from 2017 to 2022 were used to calculate nitrate-N loads. This period was chosen based on the availability of flow data. Both flow and nitrate-N concentrations measurements were obtained from Horizons Regional Council (Horizons Regional council, personal communication, August 2022).

5.3.2 Characterisation of nitrate-N concentrations and flow discharge in the drainage network

In addition to Horizons existing flow and water quality data at the Arawhata stream outflow, nitrate-N concentrations and water flow were monitored at nine sampling sites in the main drains of the Arawhata catchment (Figure 5.3). The sampling sites were located in the middle and lower parts of the catchment as these were the areas with most of the vegetable crops (Figure 5.3). Drain number 4 was not monitored because the main land uses in this area were pasture or lifestyle blocks.

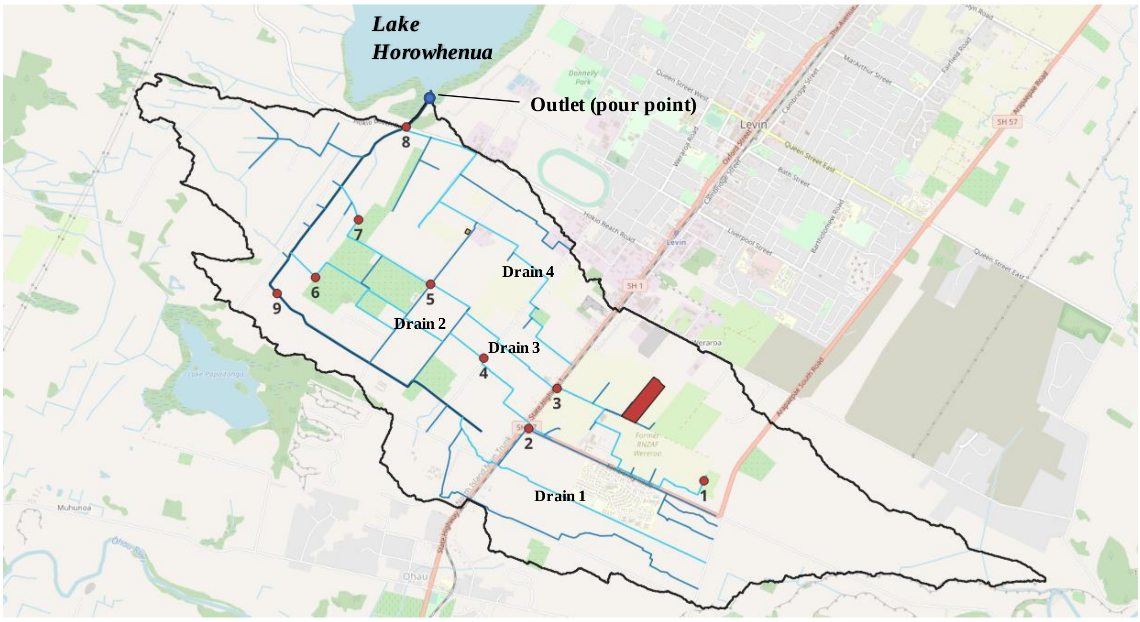

Figure 5.3. Arawhata drainage network (black line indicates the catchment boundary, blue lines for drains and culverts, light blue for main drains, dark blue line for Arawhata stream, red dots with numbers for the sampling sites and blue point for outlet).

Field visits were made (25 in total) to the sampling sites after every intense rainfall event between July 2019 and April 2021. Water quality samples and flow rate were measured every time that flowing water was observed in these drains. Once water samples were collected in 1 L bottles, they were immediately chilled in a chilly bin. They were filtered through 0.45 µm diameter membrane filters and stored in a freezer (<0°C) to decrease the microbiological and chemical activity. The samples were analysed for nitrate and ammonium forms of nitrogen using a QuikChem 8500 flow injection analyser (detection limit of 0.25 mg L^{-1} for both forms) at the Analytical Chemistry Laboratory at Massey University (Table 5.2).

Table 5.2. Nitrogen forms analysis in the water samples of drains of the Arawhata catchment.

N form	Range (mg L^{-1})	Detection limit (mg L^{-1})	Method
Nitrate-N	0-12	0.25	Reduction via copperised cadmium, NEDD coupling and colorimetry at 520 nm (Wendt, 2000).
Ammonium-N	0-12	0.25	Reaction with sodium hydroxide, potassium tartrate, buffer solution, working buffer solution, sodium nitroprusside, sodium hypochlorite and a saline diluent. Use of colorimetry at 660 nm (Wendt, 2000).

Drain flow observations were measured after intense rainfall events with a Valeport EM Flow Meter model 80, using the velocity-area method, whenever the water level was sufficient for measurement (>2 cm). At sampling site 8, the flow rate measurements were made by Horizons Regional Council as they have a monitoring station at this site.

The sampling sites 1, 3, 5 and 7 were located in the drain 3 (from top to bottom), while the sampling sites 2, 4 and 6 were located in the drain 2 (from top to bottom). Water flows in a north-westwards direction in the catchment draining to Lake Horowhenua (Figure 5.3). The sampling sites 9 and 8 were located in the Arawhata stream before and after the discharge of main drains, respectively.

The mean values of N concentrations from sampling sites were fitted to ANOVA and then compared for statistical differences between the sampling sites, following the Tukey HSD method for adjusting the p-value.

5.3.3 Modelling of stream flow and nitrate-N in the drainage network

Field sampling and analysis (section 5.3.2) was complemented with catchment-scale modelling of nitrate-N loads and flow discharge in the drainage network of the catchment. In this study, a procedure to couple the field-scale model APSIM Next Generation (Holzworth et al., 2018) and Overseer (Overseer Ltd., 2019) with the catchment-scale SWAT+ was developed simulate flow discharges and nitrate-N loads at the catchment-scale. The calibrated coupling of APSIM, Overseer and SWAT+ models was then used to quantify the potential impacts of various in-field management practices to reduce nitrate-N loads from the vegetable production areas in the Arawhata catchment.

The Soil and Water Assessment Tool (SWAT) is a semi-distributed, process-based catchment model (Arnold et al., 2012). It has been extensively used in different climatic, hydrologic, and environmental scenarios and situations world-wide, with more than a thousand peer-reviewed publications supporting it (Gassman et al., 2007; Fu et al., 2019). The SWAT model can simulate the complex spatial and temporal dynamics of water and nutrient flows across spatially heterogeneous landscapes.

The Soil and Water Assessment Tool Plus (SWAT+) is a completely restructured version of the SWAT model, which works with QSWAT+, a more user-friendly interface developed for QGIS (Bieger et al., 2017; Kakarndee & Kositsakulchai, 2020). The SWAT+ model has been developed to provide more flexibility to the model and enhance the spatial representation of water and nutrient transport processes in a catchment (Bieger et al., 2017). The SWAT+ model represents the hydrology of a catchment by using a combination of streamlines and elevation inputs to create subbasins and landscape units (LSU), and a combination of soil, land use and elevation inputs to create hydrologic response units (HRU) inside each LSU. Similarly, the model creates routing units (RU) as objects that rule interactions between HRU's, aquifers, streams, and each other (Tech, 2019).

5.3.3.1 SWAT+ inputs and assumptions

The SWAT+ requires delineation of subbasins, LSUs and HRUs to capture spatial heterogeneity in the hydrogeological environment of the catchment. To delineate subbasins in the Arawhata catchment, a Digital Elevation Model (DEM) from LIDAR with 5-m resolution was sourced from the local regional council (Figure 5.4) (Horizons Regional Council, personal communication, September 2021). In this process, an existing drain network was burned into the DEM, which was obtained from the regional council and refined with field and satellite observations (Figure 5.3).

A streamline in SWAT+ was defined as a water body draining more than 120 ha, while a channel drains an area greater than 12 ha. Landscape units were then defined by a DEM inversion, creating floodplains of streams and landscape ridges.

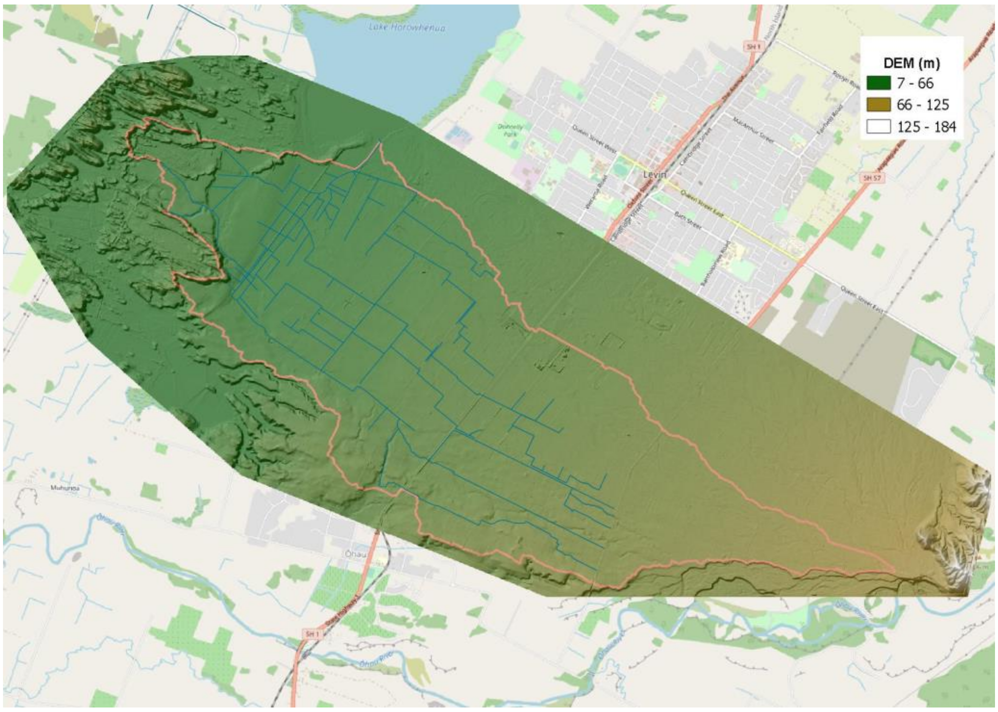

Figure 5.4. Digital elevation model, catchment delineation (orange line) and burnt streamlines of the Arawhata catchment.

Soil and land-use layers from the Arawhata catchment were used to generate HRUs (Figure 5.5) (Horizons Regional Council, personal communication, September 2021; A. Palmer, personal communication, May 2022; Landcare Research, 2018). The soil map was created based on expert description and classification (A. Palmer, personal communication, May 2022), and the Fundamental Soil Layer (FSL) (Landcare Research, 2018). The soils of the area are highly heterogeneous, therefore, the soil type assigned to each polygon was determined by the soil type with the largest coverage in that area (Figure 5.5).

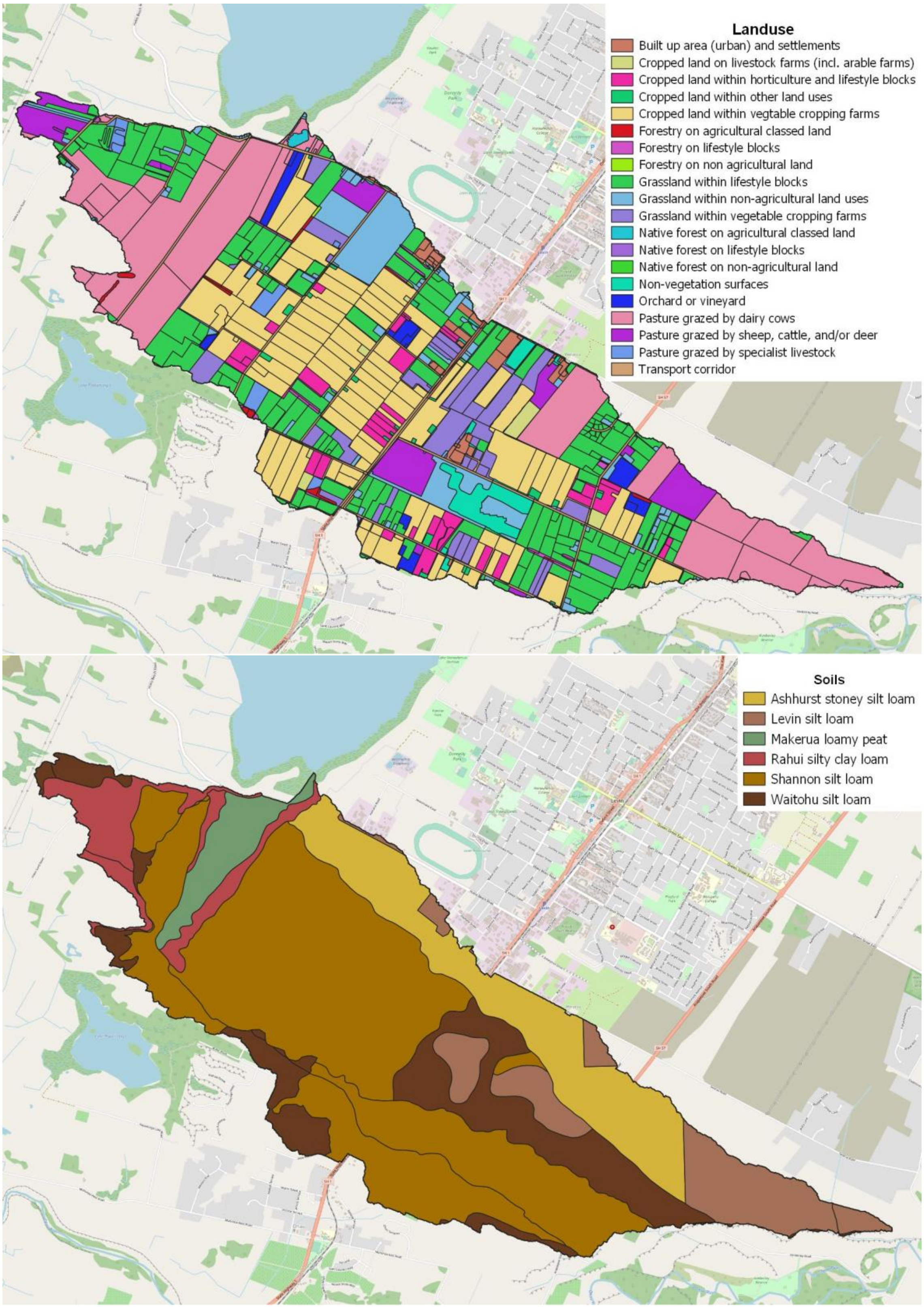

Figure 5.5. Land use map (Horizons Regional Council, personal communication, September 2021) and soil map (A. Palmer, personal communication, May 2022; Landcare Research, 2018) of the Arawhata catchment.

The SWAT+ model requires a range of soil parameters (Table 5.3). Soil parameters such as the soil saturated hydraulic conductivity (SOL_K), moist soil albedo (SOL_ALB), and the USLE soil erodibility factor were calculated from pedotransfer functions, based on the soil physical parameters of the National Soil Database (NSD) (Landcare Research, 2022; Palmer & Wilde, 2007), whilst others had to be inferred from the physical properties of the soils (Table 5.3).

Table 5.3. Soil parameters required by SWAT+.

Soil parameter	Description	Source
HYDGRP	Soil hydrologic group (A, B, C or D), classification from the U.S. Natural Resource Conservation Service	Inferred from permeability and drainage classification (NSD)
SOL_ZMX	Maximum rooting depth of soil profile (mm)	NSD
ANION_EXCL	Fraction of porosity from which anions are excluded	Default
SOL_CRK	Potential or maximum crack volume of the soil profile	Default
Texture	Texture of the soil layer	NSD
SOL_Z (layer #)	Depth from soil surface to bottom layer (mm)	NSD
SOL_BD (layer #)	Moist bulk density (g cm^{-3})	NSD
SOL_AWC (layer #)	Available water capacity of the soil (mm/mm)	Plant available water from NSD
SOL_K (layer #)	Saturated hydraulic conductivity (mm hr^{-1})	Pedotransfer functions using parameters from NSD (Cichota et al., 2013)
SOL_CBN (layer #)	Organic carbon content (%)	NSD
SOL_CLAY (layer #)	Clay content (% soil weight)	NSD
SOL_SILT (layer #)	Silt content (% soil weight)	NSD

SOL_SAND (layer #)	Sand content (% soil weight)	NSD
SOL_ROCK (layer #)	Rock fragment content (% total weight)	NSD
SOL_ALB (top layer)	Moist soil albedo	Calculated from Munsell chart colour value of the layer (Post et al., 2000)
USLE_K (top layer)	USLE equation soil erodibility factor	Calculated from the USLE K equation proposed by Williams (1995) (as cited in Neitsch et al., 2002)

For the purpose of modelling soil water balances, the land uses of the Arawhata catchment were aligned with the SWAT+ plant and urban databases. Vegetable crops were inputted as a generic arable crop (agrl), pasture inputs were based on SWAT+'s standard procedure for this plant, and lifestyle blocks with dry stock were inputted as areas planted with grassland (gras). Furthermore, channels that covered less than 2% of their subbasin, and HRU's smaller than 10% of their LSU in terms of land use, soil or elevation were merged with nearby ones. In this process, a total of 7 subbasins, 33 channels, 66 LSU's and 370 HRU's were created to simulate the spatial heterogeneity in; elevation, soils, and land uses across the Arawhata catchment.

To simulate the amount of water flowing through the catchment, SWAT+ uses the water balance equation (Equation 5.1 and Figure 5.6):

$$SW_t = SW + \sum_{t=1}^{t}(R_i - Q_i - ET_i - P_i - QR_i) \qquad (5.1)$$

where, SW is the soil water content minus the permanent wilting point water content, t is time in days, R is the amount of precipitation, Q is the amount of runoff, ET is the amount of evapotranspiration, P is the amount of percolation and QR is the return flow; all units are in mm (Arnold et al., 1998).

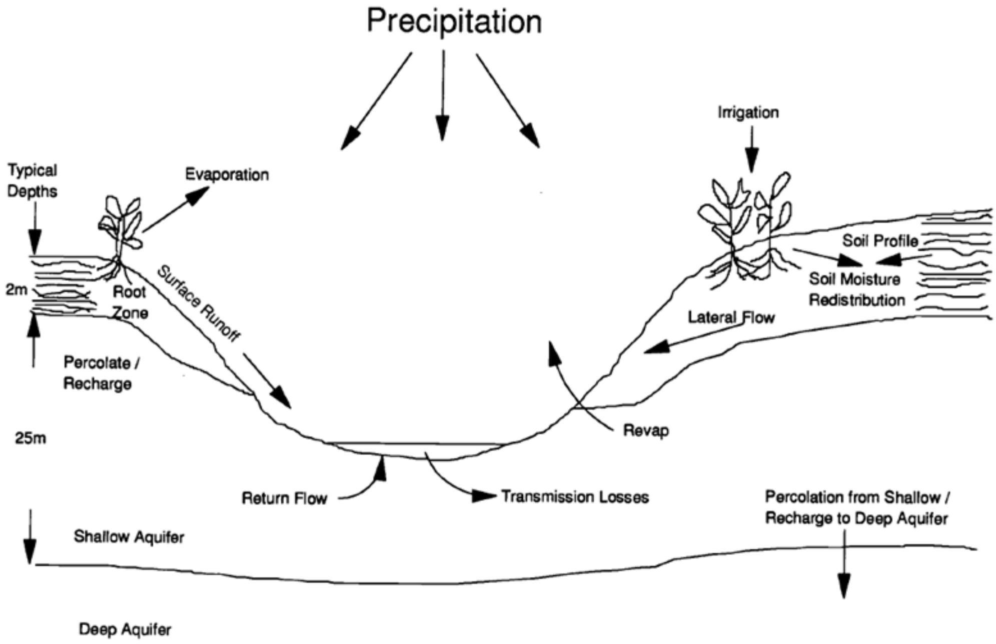

Figure 5.6. Schematic representation of the components of the water balance calculated by SWAT within a subbasin (Arnold et al., 1998).

The potential evapotranspiration is modelled in SWAT+ using the Penman-Monteith method (Monteith, 1965). Runoff is estimated using the SCS curve number method (USDA-SCS, 1972 as cited by Arnold et al., 1998). Other internal water flow processes include: a) soil moisture redistribution, which happens when the soil moisture is larger than field capacity, b) lateral flow, which depends on soil moisture, conductivity and slope, c) percolation to shallow aquifer, d) revap, which is the movement of water from shallow groundwater to the soil above in dry conditions, and d) return flow, which is the contribution from shallow aquifer to streamflow (Arnold et al., 1998). The contribution of groundwater to streamflow is calculated as a function of the water table height (m above the stream bottom) and the specific yield of the shallow groundwater (Arnold et al., 1998). Further information about the methods and equations used in SWAT and SWAT+ can be found in Arnold et al. (1998) and Arnold et al. (2012).

The SWAT+ model was run for the years 2016 and 2022 (including a one-year period to equilibrate and get the hydrologic cycle fully operational) using climate data obtained from the neighbouring weather stations (Figure 5.1). Daily rainfall (mm), solar radiation (MJ m^{-2}), maximum and minimum temperatures (°C) were obtained from the weather station “Levin AWS” whilst daily mean wind speed (m s^{-1}) and relative humidity (%) were obtained from the weather station “Levin EWS”.

Sensitivity analysis and water balance calibration

The SWAT+ model’s predictions of stream flow were calibrated by using the observed monthly streamflow data of the Arawhata stream, monitored by the regional council between 2017 and 2022 (Figure 5.1).

A set of parameters that could affect the streamflow was identified from previous research articles and guidelines (Table 5.4) (Donmez et al., 2020; Malagó et al., 2019; Zettam et al., 2020; Zhang et al., 2008). The sensitivity analysis was run in SWAT+ Toolbox v0.7.6 (Chawanda, 2021) using the Sobol sensitivity analysis, which included 12 parameters and 3000 simulations. The most sensitive parameters were then selected to perform the calibration process.

Table 5.4. Identified set of hydrologic parameters, definition, type of change and calibration range for the sensitivity analysis of the Arawhata catchment.

Parameter	Definition	Type of change	Calibration range
CN2	Condition II curve number for moisture condition	Percent change	-50 – 50
OVN	Overland Manning roughness	Percent change	-50 – 60
FLO_MIN	Minimum aquifer storage to allow return flow (m)	Percent change	-30 – 30
ESCO	Soil evaporation compensation factor.	Absolute value	0 – 1
EPCO	Plant uptake compensation factor	Absolute value	0 – 1
CN3_SWF	Pothole evaporation coefficient	Absolute value	-50 – 50

K	Saturated hydraulic conductivity (mm h^{-1})	Percent change	-30 – 30
AWC	Available water capacity of the soil (mm mm^{-1})	Percent change	-50 – 50
ALPHA_BF	Baseflow alpha factor (1 $days^{-1}$).	Absolute value	0 – 0.98
CHN	Manning coefficient for main channel	Absolute value	0 – 0.1
REVAP_CO	Groundwater "revap" coefficient.	Absolute value	0.02 – 0.2
REVAP_MIN	Threshold depth of water in the shallow aquifer for “revap” or percolation to the deep aquifer to occur (mm)	Absolute value	0 – 500

To estimate the model’s performance, Nash-Sutcliffe efficiency coefficients (*NSE*) was estimated as follows:

$$NSE = 1 - \frac{\sum_{i=1}^{m}(P_i - O_i)^2}{\sum_{i=1}^{m}(O_i - \bar{O})^2} \qquad (5.2)$$

where, m is number of the observed values, P_i and O_i are the predicted and observed values, and $\bar{O}$ is the mean of the observed values (Nash & Sutcliffe, 1970).

Though the calibration process aimed to maximise the *NSE* coefficient, the coefficient of determination (R^2) between the predicted and observed data was also considered in the model’s performance.

The calculated *NSE* and R^2 are interpreted based on the rankings developed by Moriasi et al. (2007) and Ayele et al. (2017) (Table 5.5) to assess a model’s performance.

Table 5.5. Model performance interpretation for flow (Moriasi et al., 2007; Ayele et al., 2017) (NSE indicates Nash-Sutcliffe efficiency coefficients and R^2 indicates coefficient of determination).

Interpretation	R^2	NSE
Very good	>0.7-1.0	>0.75-1.00
Good	>0.6-0.7	>0.65-0.75
Satisfactory	>0.5-0.6	>0.50-0.65
Unsatisfactory	≤0.5	≤0.50

To calibrate the model and find the best values for hydrologic parameters, an automatic calibration using the Dynamically Dimensioned Search (DDS) method was used with 500 iterations in SWAT+ Toolbox, maximising the NSE coefficient. In this process, a total of nine hydrologic parameters were identified (from the previous sensitivity analysis) and their fitted values incorporated into the model (Table 5.6).

Table 5.6. The hydrologic parameters with the greatest sensitivity and their fitted values for SWAT+ calibration in the Arawhata catchment.

Parameter	Type of change	Fitted value
PERCO	Absolute value	0.96261
CN2	Percent change	-16.55939
ESCO	Absolute value	0.02835
CN3_SWF	Absolute value	37.73374
K	Percent change	-17.10774
AWC	Percent change	-39.36457
ALPHA	Absolute value	0.80738
REVAP_CO	Absolute value	0.02563
REVAP_MIN	Absolute value	12.67206

The data from 2017 to 2022 was used to calibrate the model for streamflow. Unfortunately, the period of observed streamflow data was considered short, and the use of this data to obtain a robust model calibration was preferred instead of validating the model. However, once calibrated, the model's water balance was checked with the preliminary study results (section 5.4.1) and the previously identified catchment's water balance (Figure 5.2).

5.3.3.2 Coupling of field-scale models' nitrate-N losses into catchment-scale SWAT+ model

SWAT+ models transformations and transportation of N as a combination of inorganic and organic pools. Nutrient can be taken up by the crop or lost in surface runoff, lateral flow, leaching from the rootzone, denitrification and volatilisation (Gassman et al., 2007).

In this study, field-scale models of nitrate-N losses (leaching from the rootzone) were incorporated in the catchment-scale SWAT+ model to simulate the spatial and temporal dynamics of nitrate-N flows, including its potential attenuation, in different flow pathways across the Arawhata catchment. APSIM Next Generation (Holzworth et al., 2018) was used to simulate rootzone nitrate losses for intensive vegetable production systems and Overseer (Overseer Limited, 2019) was used to simulate rootzone nitrate losses for pasture and grasslands. However, the pattern of outdoor vegetable crop land use (i.e., the area of each crop grown) in the Arawhata catchment is highly variable for each year, and the composition of the crop rotations and fertiliser management is kept as confidential information by the growers. These intensive vegetable crop systems in the area are highly complex, with multiple rolling crop rotations in cultivation blocks as small as 0.1 hectares (Bloomer et al., 2020).

For quantification of nitrate-N losses, the vegetable crop land in the SWAT+ model (agrl) was separated into two main areas (Figure 5.7). Based reports, recommendations (L. Posthuma, personal communication, 2022; Horticulture NZ, 2017; Jolly, 2020), and field observations, the Levin and Waitohu silt loam soils were assumed to have a potato-fallow-potato-fallow-onion-ryegrass-onion-ryegrass (potato-onion) rotation (125.6 ha). Areas with other soils series were assumed to have a green vegetable crops rotation pattern (372.1 ha). This green vegetable crops rotation pattern, though unknown, was assumed to be strongly based on brassica crops (Horticulture NZ, 2017; Stout, 2021).

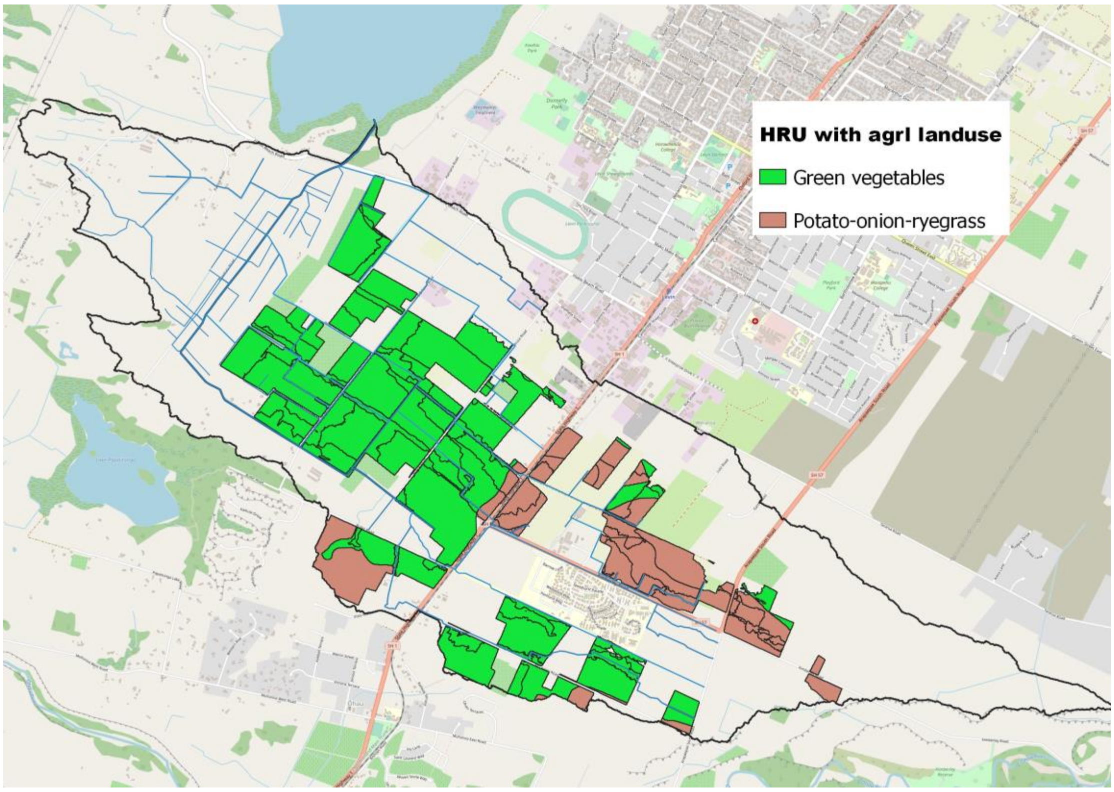

Figure 5.7. Predominant crop rotations in intensive vegetable areas of the Arawhata catchment (based on L. Posthuma, personal communication, 2022; Horticulture NZ, 2017; Jolly, 2020).

APSIM's daily nitrate-N leaching estimations for potato-onion rotations and green vegetable crop rotations for years 2016 to 2021 were used as input to the SWAT+ model (Chapter 4). In the case of the vegetable crops, nine typical rotations of vegetable crops were selected (Table 5.7, Figure 5.8). However, as the areas under vegetable crops/rotations were unknown, two different scenarios were considered. Although the exact area under each crop is not known, it is well understood that brassica crops occupy the greatest area in the catchment. According to local agronomists and different sources, it is likely that the area of brassica crop is between 65% and 80% of the catchment area (L. Posthuma, personal communication, 2022; Horticulture NZ, 2017). This information was used to inform two scenarios: (1) 'Model 1' assumes that the vegetable rotations containing brassica crops (rotations 1 to 5, Table 5.7) generated 65% of the nitrate-N leaching losses of the vegetable rotations (Br_65), and (2) 'Model 2' assumes that the vegetable rotations containing brassica crops generate 80% of the nitrate-N leaching losses of the vegetable rotations (Br_80) (these are labelled "Model' here to distinguish from scenarios which are introduced below).

Table 5.7. Representative vegetable rotations for simulation of nitrate-N losses in the Arawhata catchment.

Rotation	Type	Year 1												Year 2											
		J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D
1	Brassica rotation		Broccoli			Broccoli				Broccoli					Broccoli			Broccoli				Broccoli			
2			Cauliflower			Cauliflower				Cauliflower					Broccoli			Broccoli				Broccoli			
3	Brassica -vegetable rotation		Spinach			Broccoli				Broccoli					Spinach			Broccoli				Broccoli			
4		Lettuce				Broccoli				Broccoli				Lettuce				Broccoli				Broccoli			
5		Squash				Cauliflower				Broccoli				Squash				Cauliflower				Broccoli			
6	Intensive vegetable rotation		Spring onion				Lettuce				Spinach			Maize				Ryegrass				Cabbage			
7			Spinach			Beetroot				Ryegrass				Cabbage					Lettuce				Spinach		
8		Maize				Oats				Ryegrass					Lettuce			Cabbage					Spinach		
9		Pak choi					Spinach			Ryegrass				Beetroot				Ryegrass					Spring onion		

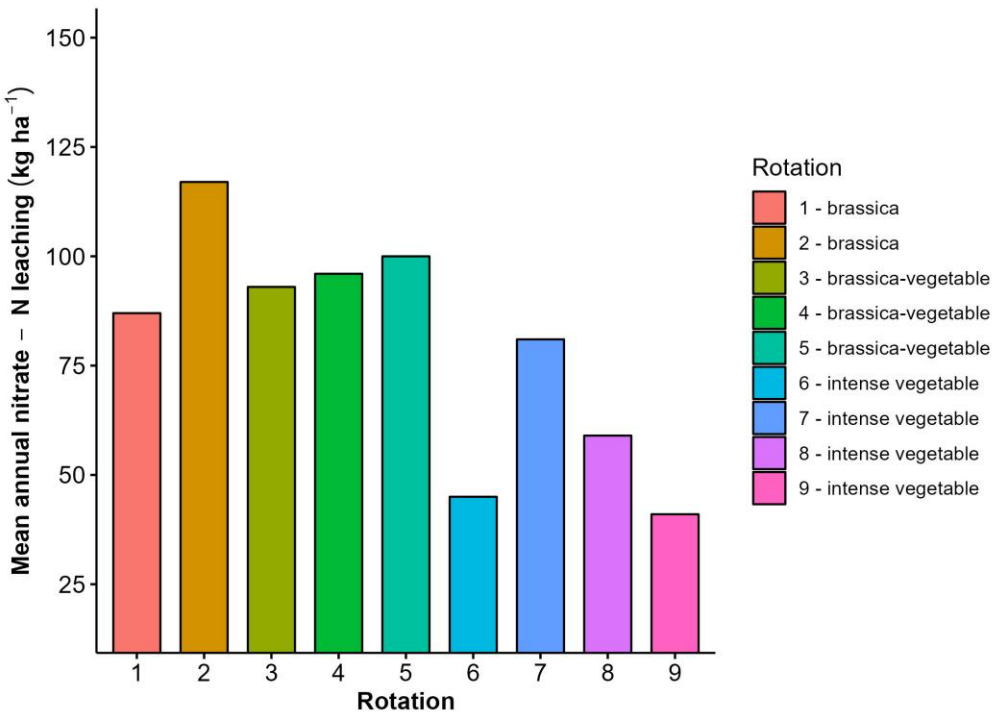

Figure 5.8. Mean annual nitrate-N leaching losses simulated in APSIM for different vegetable crop rotations over a period of 31 years (from 1991 to 2021) in the Arawhata catchment.

For the dairy farm land use in the catchment, long-term average nitrate-N leaching estimations were made in the Overseer model. Five typical farms were modelled to identify nitrate-N leaching rates from dairy farms in the upper and lower catchment (Figure 5.9). Input data for Overseer simulations of nitrate-N leaching from these representative farms was supplied by DairyNZ (A. Duker, personal communication, 2022)

Figure 5.9. A map of dairy farms simulated in Overseer in the Arawhata catchment.

The mean nitrate-N leaching losses predicted by Overseer ranged between 17 and 46 kg nitrate-N ha^{-1}, depending on the soil (Table 5.8). The mean annual nitrate-N leaching estimations from lifestyle blocks containing dry stock were estimated in Overseer as approximately 10 kg ha^{-1}. Overseer losses were expressed on a daily basis (uniformly)and associated with soils of the Arawhata catchment as described in SWAT+.

Table 5.8. Mean annual nitrate-N leaching losses (kg ha^{-1}) from soils under dairy farm predicted by Overseer, and the equivalent soil series in SWAT+.

Dairy farm	SWAT+ soil	SMAP-Overseer soil	Mean annual nitrate-N leaching losses (kg ha^{-1})
1	Makerua	Wind_16a.1	17
1	Shannon	Ngam_22a.1	35
1	Rahui	Ymai_53a.1	18
2	Shannon	Ngam_22a.1	35
2	Waitohu	Bend_4a.1	28
2	Levin	Heat_1a.1	35
3	Makerua	Wind_16a.1	18
3	Rahui	Long_27a.1	30
3	Rahui	Ymai_53a.1	19
4	Ashhurst	Mand_86a.1	46
4	Levin	Kirk_36a.1	32
5	Ashhurst	Mand_86a.1	46
5	Levin	Kirk_36a.1	32

As SWAT+ calculates its own nutrient leaching losses inside the model, the "point source" elements in the model were used to 'override' these values and insert the field-scale nitrate-N leaching simulated by APSIM for vegetable areas and Overseer for dairy and lifestyle/dry stock blocks. Though point source objects (or "recall" objects) were originally made for simulating point-source pollution into streams, these objects can also be set up to outflow to other objects of the catchment (aquifers, streams, routing units). In this case, point source objects were created to simulate the model's HRU, incorporating the field-scale nitrate-N leaching losses, using the following steps:

1. Routing units (or LSU's) and their respective outlet aquifers were identified.
2. HRUs connected to each respective RU were identified.

3. HRUs were filtered depending on land-use and soil, and their areas and simulated water flow pathways (daily percolation and surface runoff volumes) were extracted.
4. Vegetable crops and pasture grazing land-uses were identified and targeted.
5. Creation of point source objects resembling original model's HRU's water balance.
6. Water flow (percolation, surface runoff and lateral flow) for each day per point source was processed, conserving the same amounts of water outflowing to the same objects as the original HRU's.
7. Daily nitrate-N loads (kg nitrate-N ha^{-1}) (from respective land uses) were incorporated in the point source elements and so mixed with the water outflowing to their respective aquifers.
8. Manual incorporation of point source objects into the project's database.
9. Set fraction of original model's HRU with dairy, dry stock and vegetable crop land-use to 0.

Figure 5.10 schematises the modelling process used for coupling the field-scale nitrate-N leaching losses as point source elements into the SWAT+ model.

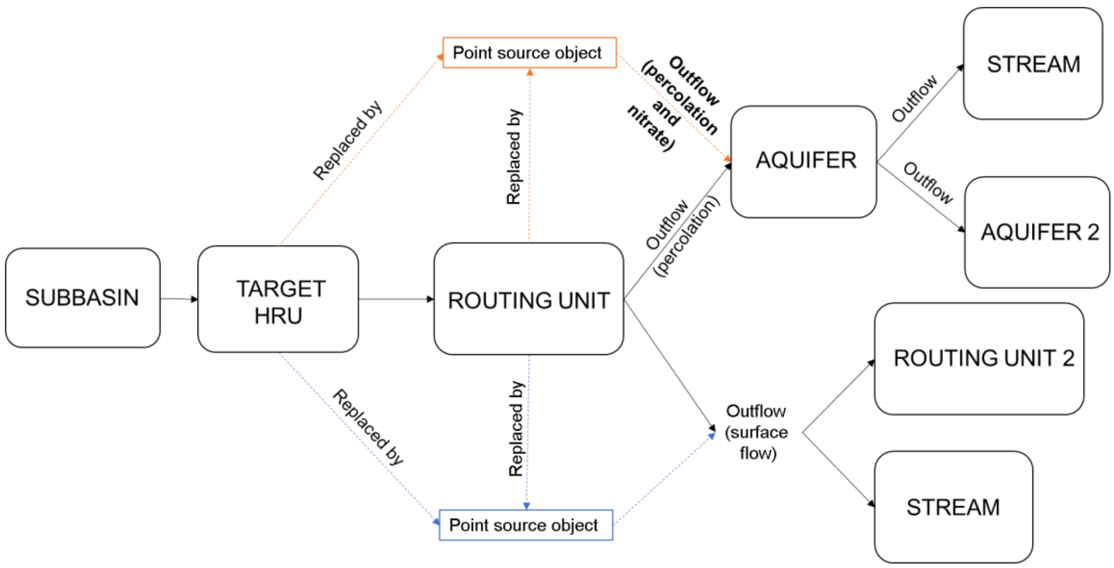

Figure 5.10. Coupling of the field-scale nitrate-N leaching losses as point source elements into the SWAT+ model.

Nitrate-N amounts in surface runoff could not be predicted by the field-scale models and incorporated into the SWAT+ model. However, nitrate-N losses in surface runoff were considered negligible for this catchment, as most of the nitrate-N is transported as percolation through groundwaters in the catchment. Studies conducted by Burkitt (2014) and (Min & Shi, 2018) have shown that nitrate-N losses through surface runoff are typically low when compared to leaching losses in pastures and vegetable fields.

5.3.3.3 Nitrate-N load calibration

Testing the combined models' (SWAT+ and APSIM/Overseer) predictions with measured water quality data in the Arawhata stream was not straightforward, because only one grab sample was taken monthly to monitor nitrate-N concentrations. Furthermore, not all months from 2017 to 2022 had water quality data, and therefore only months when the observed nitrate-N data was available were used for comparison between the observed and predicted mean monthly nitrate-N loads in Arawhata stream.

The APSIM/Overseer coupled with the SWAT+ predicted mean monthly nitrate-N loads and mean annual nitrate-N loads in Arawhata stream. The predicted mean monthly nitrate-N loads were compared with the observed data in the Arawhata stream over 4 years from 2017 to 2021. The observed monthly nitrate-N loads (in t) were calculated as:

$$L = F * C * 1{,}000{,}000 \qquad (5.3)$$

where, L is the observed monthly mean nitrate-N loads in the Arawhata stream (in t), F is the observed monthly mean flow (in m^3) C is the observed mean nitrate-N concentrations (in g m^{-3}) (Elwan et al., 2018).

From the predicted nitrate-N river loads and the predicted nitrate-N leaching losses for each land use, the nitrate-N attenuation factor (%) for the Arawhata catchment was estimated following the procedure described by (Elwan, 2018):

$$AF_N = \left[\frac{(N_{Rootzone} - N_{River\ load})}{N_{Rootzone}}\right] * 100 \qquad (5.4)$$

where, AF_N is the nitrate attenuation factor, $N_{Rootzone}$ is the predicted nitrate-N leaching from the rootzone (t yr^{-1}) and $N_{River\ load}$ is the predicted nitrate-N loads in Arawahata stream (Elwan, 2018)

Field-scale nitrate-N leaching losses can potentially be attenuated in the flow pathways in the subsurface environment (Elwan, 2018; Collins et al., 2017; Rivas, 2018). As field-scale nitrate-N leaching amounts were simulated externally to SWAT+, nitrate-N calibration parameters that rule at the HRU/soil level in SWAT+ (such as denitrification exponential rate coefficient (CDN), or denitrification threshold water content (SDNCO)) were considered negligible and set to a minimum. However, nitrate-N attenuation in groundwaters was considered using the parameter HL_NO3 (or H_LIFE), which sets the time that it takes to reduce nitrate-N concentrations to a half in aquifers of the model (i.e., HL is the half-life). In the Model 1 (Br_65) and Model 2 (Br_80) of the field-scale nitrate-N leaching losses, the value of the nitrate attenuation parameter HL_NO3 was varied to best fit. The best fit was identified using the model performance parameters to compare the predicted and observed monthly mean nitrate-N loads in Arawhata stream (Table 5.9). In this process, the HL_NO3 parameter was set equally to all aquifer objects of each model, assuming an '*effective*' parameterisation of uniform nitrate attenuation in shallow groundwater across the catchment. This is a simplification applied in absence of more detailed information on any potential spatial variability of nitrate attenuation in the catchment. The initial nitrate-N concentration in the shallow groundwater was set to 11 g m^{-3}, which was the mean observed nitrate-N concentrations in the Arawhata stream during the three months with the lowest average rainfall (January, March and October), representing baseflow (groundwater) as the major contributor to stream flow and so the stream nitrate-N concentration is likely to reflect groundwater concentration.

Table 5.9. Model nitrate-N parameter and fitted values for each scenario.

Parameter	Model 1 (Br_65)	Model 2 (Br_80)
HL_NO3 (days)	1100	800

5.2.3.4 Scenarios simulating in-field management practices to reduce nitrate-N loads

The calibrated Model 1 (Br_65) and Model 2 (Br_80) were used to simulate the ability of five scenarios of in-field crop and fertilizer management practices to reduce nitrate-N loads. These in-field practices were first simulated in APSIM and then their field-scale nitrate-N leaching losses were incorporated in the SWAT+ model to evaluate their effects on nitrate-N loads in the Arawhata stream:

- Scenario 1: A ryegrass cover crop is grown in fallow periods of the potato-onion rotations (125.6 ha).
- Scenario 2: The brassica rotations (1 - 5) (372 ha) which have large nitrate-N leaching rates are replaced by green vegetable crops rotations with small nitrate-N leaching rates (6 and 9) (Table 5.7, Figure 5.8).
- Scenario 3: A combination of scenarios 1 and 2 described above.
- Scenario 4: A decrease of 10% in the quantity of N fertiliser applied to all crops.
- Scenario 5: A decrease of 20% in the quantity of N fertiliser applied to all crops.

5.4 Results

5.4.1 Characterisation of flow and nitrate-N concentrations in the drainage network

There were only a few rainfall events that led to a measurable water level in the drains during the monitoring period (July 2019 to April 2021). Therefore, a qualitative monitoring of the drain flow was also undertaken. Most of the drains did not have significant water flow, suggesting that the drainage network had been designed for exceptionally large rainfall events (Figure 5.11).

From the field visits to characterise flow qualitatively (21 flow visits), drains were found as 'dry' 38% of the time. On 62% of the visits that water was found in these drains, only 37% of the time this water was flowing and measurable. Therefore, water in the drains was usually stagnant in puddles. The drain monitoring sites in the lower parts of the catchment (i.e., sampling sites 6, 7 and 9) were more likely to be flowing than the drains in the middle parts of the catchment (i.e., sampling sites 1, 2, 4 and 5).

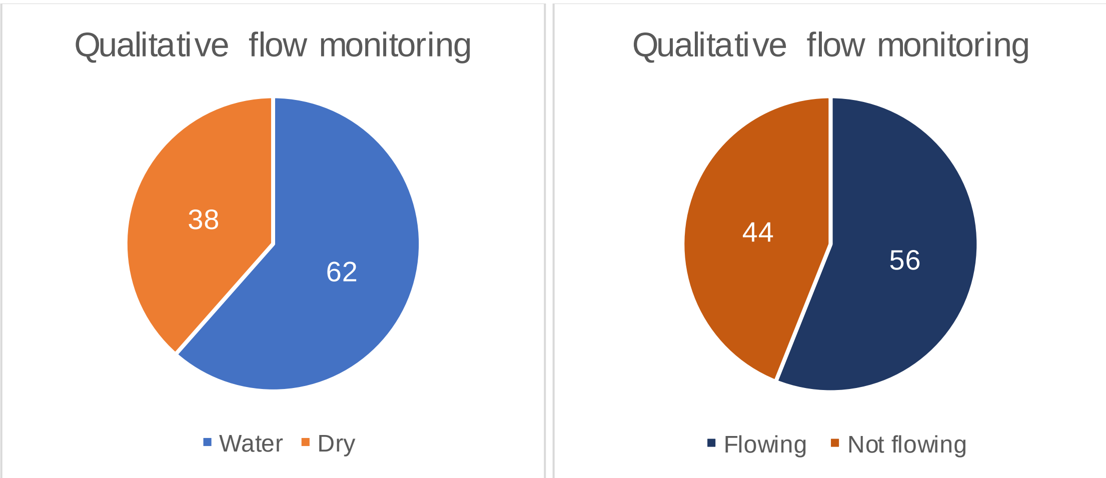

Figure 5.11. Descriptive monitoring of the flow in drains of the Arawhata catchment observed from 2020 to 2021.

The largest mean drain flow was 23.4 L s^{-1} at the sampling site 9, while the smallest mean measurable drain flow rate was 0.002 L s^{-1} at the sampling site 4 (Table 5.10). Flow rates in drains that flow to the Arawhata stream ranged from 0.002 to 23.357 L s^{-1}. In the drain 3, there was a decrease in flow rate of 60% between sampling sites 3 and 5, but an increase of 72% between sampling sites 5 and 7 (Table 5.10). In the drain 2, there was a substantial increase in the flow rate from middle to bottom of the catchment, with a thousand times more flow rate in the sampling site 6 than in the sampling site 4 (Table 5.10). Overall, sampling sites in the lower parts of the catchment had a greater number of observations.

Table 5.10. Mean flow rate in the sampling sites of drains of the Arawhata catchment observed from June 2020 to April 2021.

Sampling site	Mean flow rate (L s^{-1})	Samples	Field visits
3	2.193	3	21
4	0.002	2	21
5	0.878	3	21
6	2.367	4	21
7	1.510	5	21
8*	251.825	-	21
9	23.357	6	14

*Monitored by Horizons Regional Council (as described in section 5.2.2)

The analysed mean nitrate-N concentrations between different drain sampling sites were fitted to ANOVA and further Tukey test. This analysis found a statistical difference in the means of nitrate-N concentrations between the drain sampling sites (Figure 5.12). The sampling site 8 had the greatest nitrate-N concentrations with 8.29 g m^{-3}, whilst the sampling site 4 had the lowest nitrate-N concentration with 1.27 g m^{-3}. Nitrate-N concentrations generally had a large standard deviation, suggesting a high temporal variability in nitrate-N concentrations in the drains. Ammonium-N concentrations were not reported as they were below the analysis detection limit (<0.25 mg L^{-1}), suggesting that most of the soluble inorganic nitrogen was flowing in the catchment's drainage network in nitrate-N form.

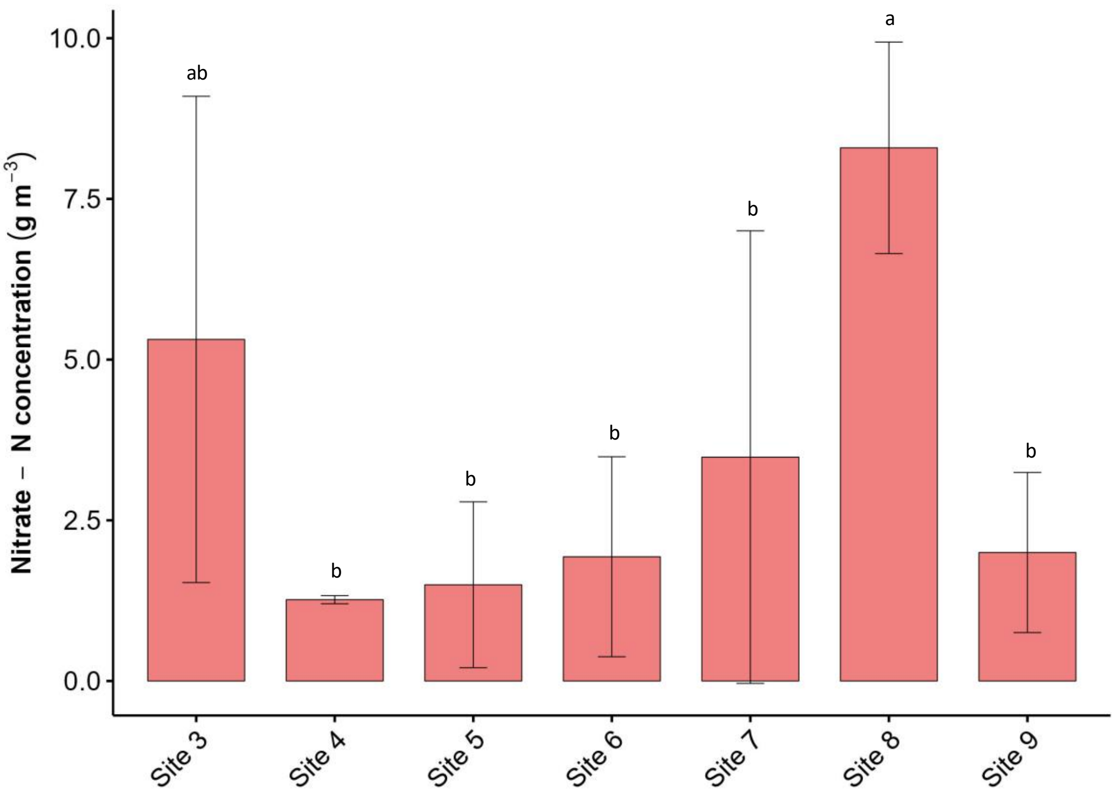

Figure 5.12. Mean nitrate-N measured in the drain sampling sites of the Arawhata catchment after intense rainfall events, from June 2020 to April 2021. Bars indicate standard deviations for each sampling site. Different letters indicate statistically significant differences between means of the sampling sites ($p<0.05$, Tukey method for p-value adjustment).

The sampling sites 9 and 8 in the Arawhata stream represent nitrate-N concentrations before and after the influence of the main drains of the catchment (Figure 5.3), respectively. Interestingly, the sampling site 8 had four times greater nitrate-N concentrations than sampling site 9. Also, The flow from the sampling site 9 to the sampling site 8 increased from 23.4 to 251.8 L s^{-1}. The large increase in flow rate in site 8, in comparison to flow rates observed main drains (sites 6 and 7), confirms that the upwelling of groundwater in lower parts of the landscape along the stream is the main flow pathway in the catchment.

5.4.2 Prediction of stream flow and catchment water balance

Flow rates in Arawhata stream for this period were higher during the winter months (July-September), as compared to the summer months (October-May) (Figure 5.13). The largest flow rate recorded in the Arawhata stream was 0.51 m^3 s^{-1} in August 2017, and the smallest flow rate was recorded at 0.09 m^3 s^{-1} in January 2018.

The SWAT+ prediction of mean monthly flow rate in Arawhata stream was rated as 'good' to 'very good' with an *NSE* score of 0.68 and R^2 of 0.74 (Figure 5.13). However, there appears to be a mismatch between the observed and predicted stream flows from January 2022 to July 2022.

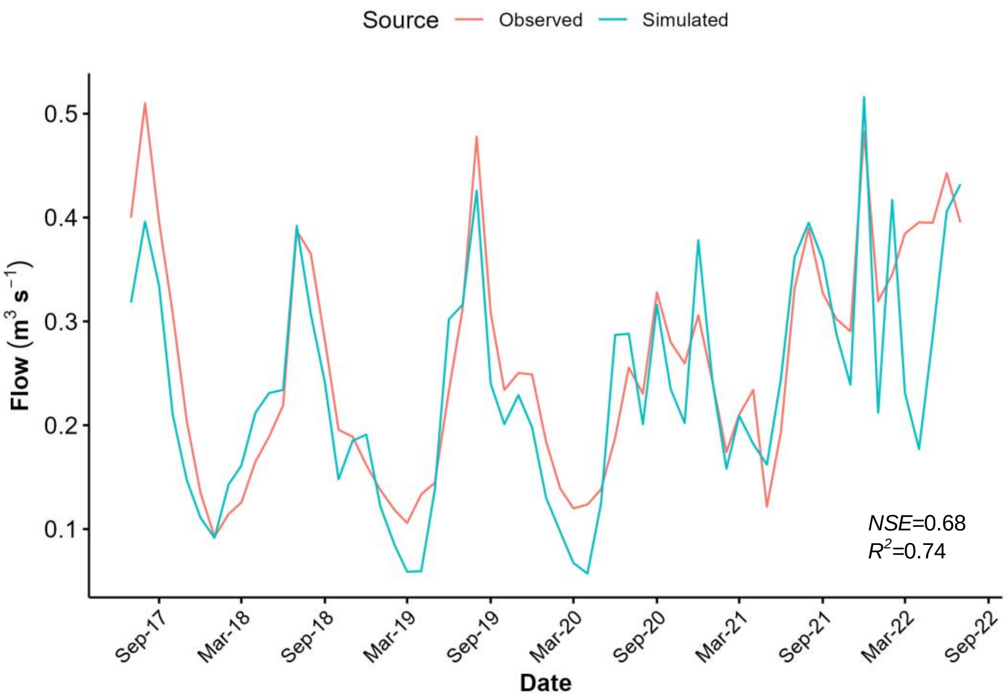

Figure 5.13. A comparison of the mean monthly observed and predicted stream flow (m^3 s^{-1}) of Arawhata stream from 2017 to 2022.

The average annual water balance (mm) of the Arawhata catchment predicted by the calibrated SWAT+ model is summarised in Figure 5.14. The influence of irrigation in the water balance was deemed as negligible, as most growers in the area do not need to irrigate unless facing drought conditions. Percolation estimations from SWAT+ to shallow aquifers were in the normal range, as simulations from APSIM Next Generation showed an overall mean drainage of 439 mm for years 2017 to 2022 (Chapter 4). In addition, similar evapotranspiration values to those reported by SWAT+ were obtained in APSIM Next Generation for years 2017-2022, with values ranging between 280 and 591 mm, and an overall mean of 500 mm.

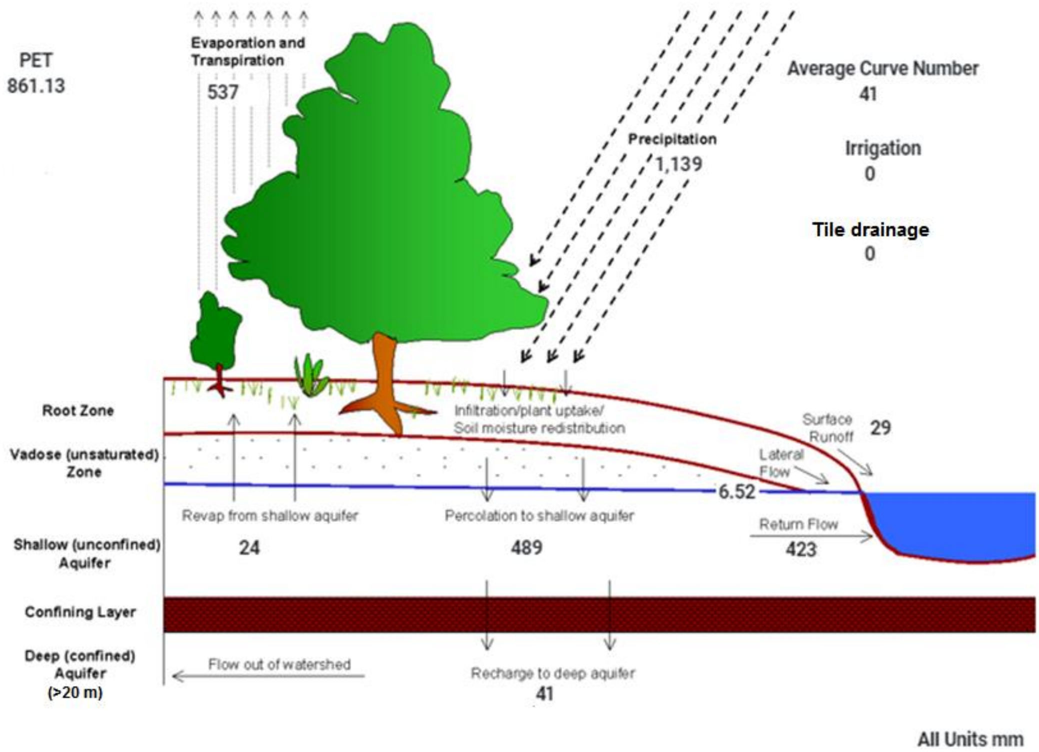

Figure 5.14. Average annual water balance (mm per year) predicted by SWAT+ for the Arawhata catchment over 5 years, from 2017 to 2022.

5.4.3 Prediction of nitrate-N loads

Initially the SWAT+ simulations were found to provide abnormally small nitrate-N loads in Arawhata stream. Technical issues with flow nitrate-N contributions from aquifers to streams were found and reported to the SWAT+ team who are located in College Station, Texas, USA. These problems were kindly fixed, and an upgraded version of the model was provided (Bieger, Arnold & Sammons, personal communication, August 2022). As the water balance of the original model (before incorporating point source objects) was copied into the new model (after incorporating point source objects), there were essentially no differences in both models' performance in prediction of stream flow and both models showed the same *NSE* and R^2 coefficients for predicted mean monthly flow rate of the Arawhata stream over the 4 years from July 2017 to December 2021.

Comparisons of the observed and predicted mean monthly nitrate-N loads in the Arawhata stream showed that the performance of both the Model 1 (Br_65) and the Model 2 (Br_80) was "good" and "very good" for *NSE* and R^2, respectively (Figure 5.15). The model 2 (Br_80), that assumes a greater area of brassicas rotation in the catchment, had a slightly better fit to the observed mean monthly nitrate-N loads in Arawhata stream (Figure 5.15).

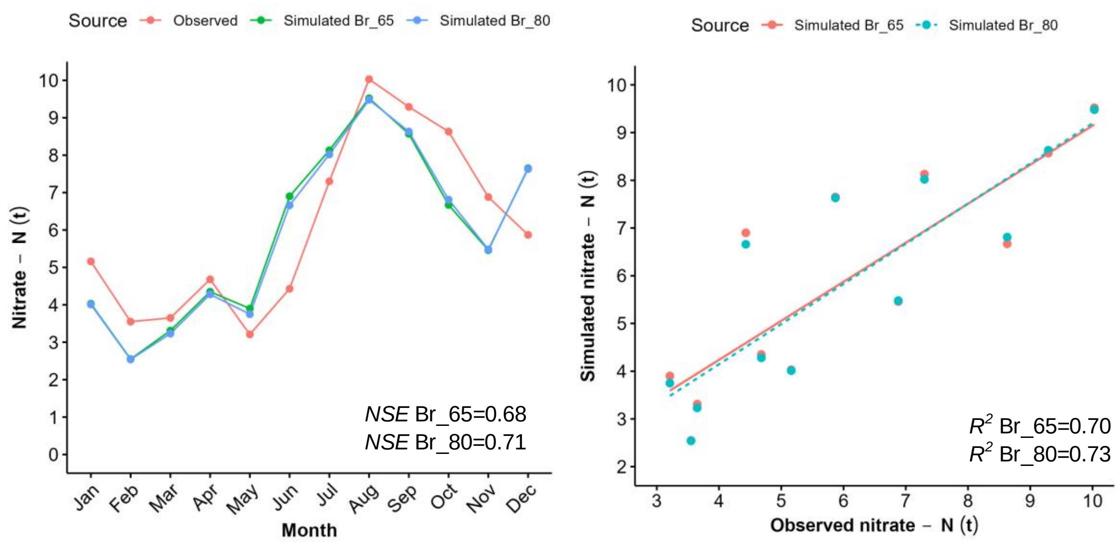

Figure 5.15. A comparison of the predicted and observed mean monthly nitrate-N loads in the Arawhata stream over a period of 4 years, from 2017 to 2021. Model 1 (Br_65) considers that brassica crops represented the 65% of nitrate-N leaching losses of vegetable crop rotations, and Model 2 (Br_80) considers that brassica crops represented 80% of nitrate-N leaching losses of vegetable crop rotations.

The observed nitrate-N loads between 2017 to 2021 were slightly larger than the predicted nitrate-N loads, especially in the year 2017 (Table 5.11). The smallest difference between the observed and predicted nitrate-N loads was in the year 2021, where the nitrate-N load predicted by the Model 1 (Br_65) and Model 2 (Br_80) was 0.01 t and 0.5 t smaller than the observed load, respectively. Note that predicted values obtained in periods when there were no observed values were omitted for comparison purposes.

Based on the statistical parameters, the SWAT+ model performed very good for both models, Model 1 (Br_65) and Model 2 (Br_80) (Table 5.11). The model showed no consistent under- or over estimation of the mean monthly nitrate-N loads (Figure 5.15) and nitrate-N loads between 2017 to 2021 in the Arawhata stream. Overall, the predicted mean nitrate-N load was only 3% (2 t) and 4% (2 t) smaller than the observed mean nitrate-N load in Arawhata stream (Table 5.11). This supports the coupling of the field-scale estimates (by APSIM and Overseer) of nitrate-N leaching into the SWAT+ model for prediction of mean monthly and annual nitrate-N loads of the Arawhata stream.

Table 5.11. A comparison of observed and predicted nitrate-N loads in Arawhata stream over a period of 5 years, from 2017 to 2021. The Model 1 (Br_65) considers that brassica crops represented the 65% of nitrate-N leaching losses of vegetable crop rotations, and Model 2 (Br_80) considers that brassica crops represented the 80% of nitrate-N leaching losses of vegetable crop rotations.

Year	Observed nitrate-N loads (t)	Predicted nitrate-N loads – Model 1 (Br_65) (t)	Predicted nitrate-N loads - Model 2 (Br_80) (t)
2017	53.2	37	37.4
2018	15.5	19	18.2
2019	56.6	54.4	54.1
2020	54.2	62.0	61.1
2021	81.3	81.3	80.8
Mean	**52.2**	**50.6**	**50.3**
NSE		**0.85**	**0.86**
R^2		**0.86**	**0.87**

According to SWAT+, Model 1 (Br_65) had a mean annual leaching rate of nitrate-N from the farms of 65,753 kg (Table 5.12) and a mean annual nitrate-N load to Arawhata stream of 62,720. The difference of 3,033 kg suggests a mean annual attenuation of 5.4% (Table 5.12). For the Model 2 (Br_80), a difference of 5,931 (Table 5.13) suggests a mean annual attenuation of 9.3%. The Model 2 (Br_80), simulating relatively larger nitrate-N leaching (as compared to the Model 1 (Br_65)), representing a greater area under the brassicas rotation, resulted in smaller values of HL_NO3 (Table 5.9), meaning greater nitrate-N attenuation in groundwater (Table 5.13). This was because both models, the Model 1 (Br_65) and Model 2 (Br_80) were calibrated to the observed load of nitrate-nitrogen in the Arawhata stream and resulted in similar total nitrate-N outputs (Tables 5.12 and 5.13).

Attenuation has been described above in terms of half-life and the ratio of N input to N output. The biophysical hydrologic link is the mean transit time of N in the catchment system. For Br_65, if the half-life is 1100 days, and there is a 5.4% loss of N, then the mean transit time for this loss to occur must have been 88 days. For Br_80, with a half-life of 800 days and a loss of 9.3% of N, then the mean transit time must have been 113 days.

Table 5.12. Total annual nitrate-N input, output, their difference and calculated attenuation factor for the Model 1 (Br_65) for years 2017 to 2021 in Arawhata catchment. The Model 1 (Br_65) considers that brassica crops represented 65% of nitrate-N leaching losses of vegetable crop rotations.

Year	**Total nitrate-N input (kg)**	**Total nitrate-N output (kg)**	**Difference (kg N)**	**Attenuation (%)**
2017	45,552	36,890	8,662	
2018	61,261	62,930	-1,669	
2019	74,666	67,480	7,186	
2020	65,247	65,030	217	
2021	82,037	81,270	767	
Mean	**65,753**	**62,720**	**3,033**	**5.4**

Table 5.13. Total annual nitrate-N input, output, their difference, and calculated attenuation for the Model 2 (Br_80) for years 2017 to 2021 in Arawhata catchment. The Model 2 (Br_80) considers that brassica crops represented 80% of nitrate-N leaching losses of vegetable crop rotations.

Year	**Total nitrate-N input (kg)**	**Total nitrate-N output (kg)**	**Difference (kg)**	**Attenuation (%)**
2017	47,221	37,480	9,741	
2018	62,978	62,230	748	
2019	77,994	66,960	11,034	
2020	67,684	64,110	3,574	
2021	85,348	80,790	4,558	
Mean	**68,245**	**62,314**	**5,931**	**9.3**

5.4.4 Impact of different in-field management strategies on nitrate-N loads

All five simulated scenarios were found to reduce nitrate-N loads in Arawhata stream (Figure 5.16).

The results of Model 1 (Br_65) and Model 2 (Br_80) followed similar trends, where the most to least efficient in-field management practices for reducing nitrate-N loads were predicted as Scenario 3, followed by Scenario 2, Scenario 5, Scenario 1 and Scenario 4, respectively. The single practice that produced the most impact in reducing nitrate-N loads in Arawhata stream was Scenario 2, with a reduction of 17 and 21% of mean annual nitrate-N load predicted by the Model 2 (Br_65) and Model 2 (Br_80), respectively.

The smallest reduction in mean nitrate-N loads was produced when simulating a decrease of 10% of N fertiliser applied to all crops. This option resulted in only 1 to 4 % decrease in the mean annual nitrate-N load in the Model 1 (Br_65) and the Model 2 (Br_80), respectively. However, simulation of a decrease of 20% of N fertiliser use in areas with crops resulted in 6 to 8% decrease in the mean annual nitrate-N load for the Model 1 (Br_65) and the Model 2 (Br_80), respectively. However, according to APSIM estimations, crop yields were reduced in approximately 3% and 6% between 2017-2022 when decreasing 10 and 20% of N fertiliser application, respectively.

The greatest reduction of 21 to 25% in the mean nitrate-N loads by the models, the Model 1 (Br_65) and Model 2 (Br_80), was achieved by combining two practices (Scenarios 1 and 2); cultivating ryegrass cover crops during the fallow period in areas with potato-onion (Scenario 1), and incorporating rotations of non-brassica vegetables that produce low nitrate-N leaching losses in areas with miscellaneous vegetable crop rotations (Scenario 2). The difference between the Scenario 3 and the baseline was statistically significant (p=0.013). No other statistically significant differences were found between different scenarios.

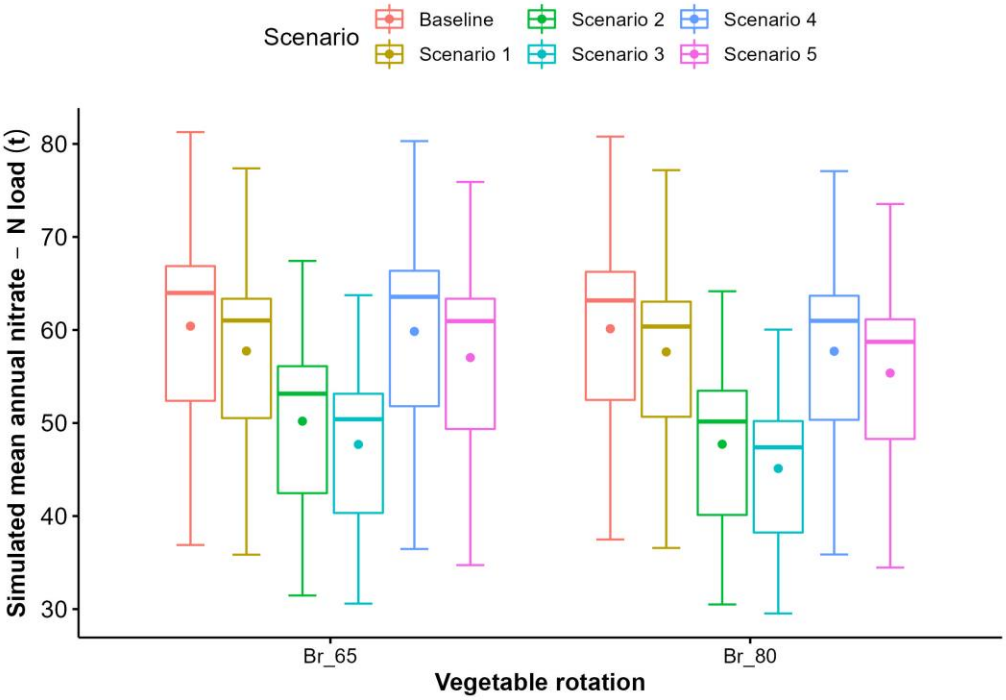

Figure 5.16. Predicted median, mean, quartiles, maximum, and minimum of annual nitrate-N loads in the Arawhata stream in different in-field mitigation strategies scenarios. Model 1 (Br_65) considers that brassica crops represented the 65% of nitrate-N leaching losses of vegetable crop rotations, and Model 2 (Br_80) considers that brassica crops represented the 65% of nitrate-N leaching losses of vegetable crop rotations. Scenario 1 considers ryegrass cover crop in land use with potato-onion rotations when fallow period occurs, Scenario 2 considers a change of rotations in areas with miscellaneous green vegetable crops, Scenario 3 considers a combination of the Scenarios 1 and 2, Scenario 4 considers a decrease in 10% in N fertilisation of all land uses with crops, and Scenario 5 considers a decrease in 20% in N fertilisation of all land uses with crops.

5.5 Discussion

5.5.1 Characterisation of water and nitrate-N flows in the Arawhata catchment

The observations of water and nitrate-N flows from the characterisation of the catchment undertaken between 2020 and 2021 agreed well with the SWAT+ simulations, suggesting that most of the water from rainfalls (recharge) is transported to Arawhata stream through shallow and deep groundwater, while a small portion is transported through surface runoff via drains and smaller streams. Thomas & Gibbs (2014) and Thomas & Garden (2021) suggested that this upwelling of shallow groundwater is due to a basement high (the Poroutawhao High) situated underneath the Arawhata stream. The Poroutawhao High, which is a greywacke basement that appears within 20 m of the surface, is aligned to the west of the Lake Horowhenua and includes a significant fault (the Poroutawhao/Levin Fault) (Figure 5.17) (Thomas & Garden, 2021; Hughes, 2005; Hughes & Kennedy, 2009). The fault helps to impound the lake and brings basement closer to the surface.

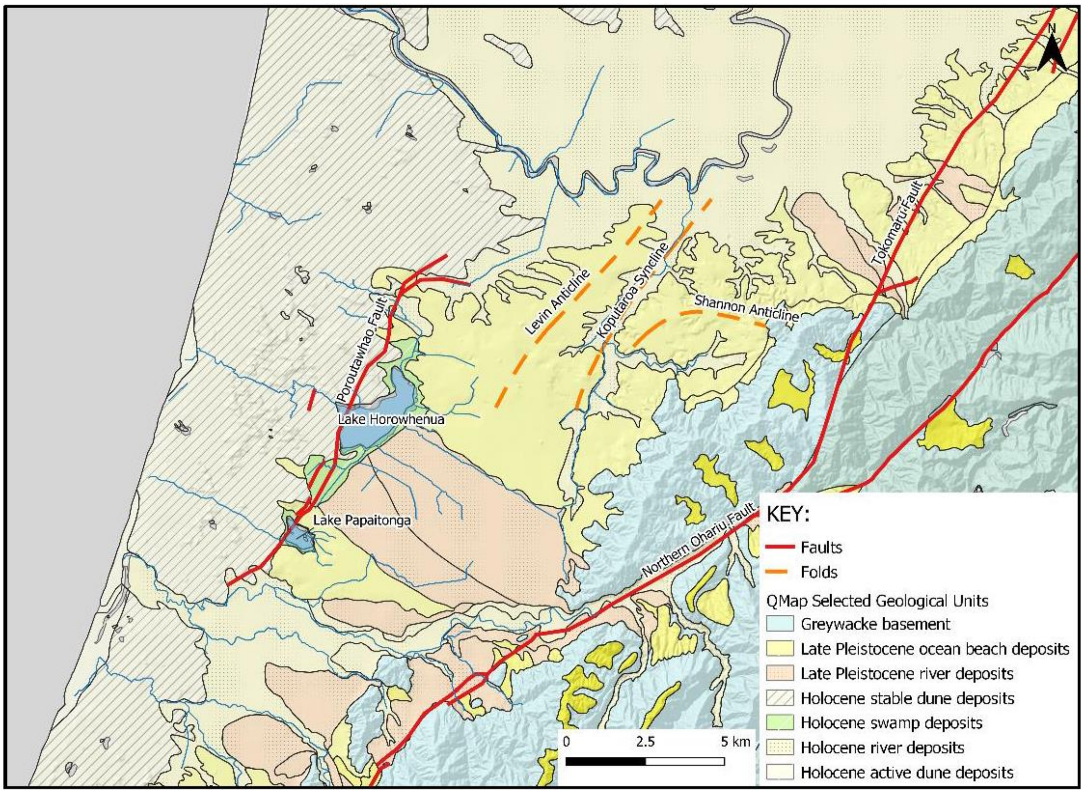

Figure 5.17. Map of the geology of the area (Thomas & Garden, 2021)

The drain flow observations suggests that there is very limited connectivity between drains and shallow groundwater in the middle and upper parts of the catchment, and most of the drain flow occurs in the lower catchment where groundwater is also at its shallowest depth (Chapter 3) (Figure 5.18). Additionally, water sitting in the drains and culverts without flow (which was observed in the sampling sites in field conditions, normally in the middle-catchment) could potentially imply that drain water could be percolating to shallow and deep groundwater.

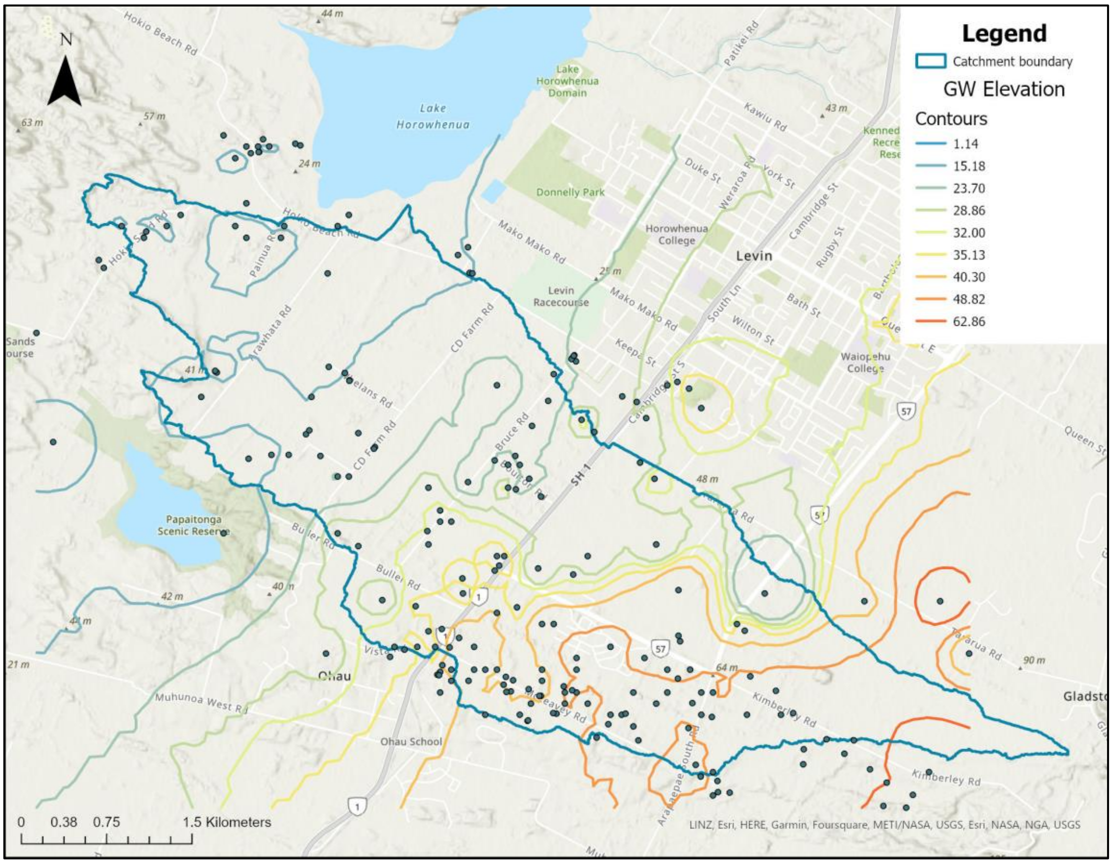

Figure 5.18. Groundwater contour map of 216 boreholes monitored by Horizons Regional Council in the Arawhata catchment area in December 2022 (dark blue line indicates the catchment boundary, points indicate monitored borehole and coloured contour lines indicate groundwater elevation).

Soils in parts of the catchment have been classified as imperfectly drained, which would normally indicate a runoff process dominance. However, these soils are also developed over gravel or sand, which are highly permeable, and surface water can be easily percolated once it has been reached over these well-drained areas. In addition, a large part of the soils in the catchment are also under constant ploughing, with a plough pan at 30-40 cm and natural compaction below (loess compaction), suggesting that the first layer of soils is highly permeable, and water may flow slowly in the subsurface. As a result, water that falls on the soils of the Arawhata catchment is mostly perching, slowly flowing in the subsurface towards the stream and lake or reaching the gravels underneath where it flows rapidly. The common presence of oxidised gravels under these soils supports this statement (A. Palmer, personal communication, May 2022) (Table 3.1, Chapter 3). Therefore, the suggested travel times for nitrate-N in the catchment system of 88 days and 113 days seem reasonable.

The results of N analyses showed that most of the soluble inorganic nitrogen found in drains was in the form of nitrate-N, as ammonium-N was close to <0.25 g m^{-3}. Statistical differences between nitrate-N concentration samples taken before and after the influence of drains and aquifers (the sampling sites 8 and 9) confirmed that most of the nitrate-N contributions occurs along Arawhata stream in the lower part of the catchment. This suggests that a combination of the base flow (shallow groundwater) and the flow from drains 2 and 3 (between the sampling sites 9 and 8) contributes a major portion of nitrate-N flowing from Arawhata stream to Lake Horowhenua. Nitrate-N concentrations found in these drains were comparatively lower than concentrations found in the shallow groundwater (Chapter 3), and therefore, nitrate-N found in the subsurface environment is likely to be the main contribution of nitrate-N loads to the stream.

5.5.2 Flow and nitrate-N load evaluation in SWAT+ model

The observed and predicted data indicated that flow and nitrate-N magnitudes increase in Arawhata stream between June and September (winter-spring) and are generally lower between the months of January and May (summer-autumn).

The coupling of field-scale nitrate-N leaching (simulated by a combination of APSIM for the vegetable areas and Overseer for the pastoral areas) in the catchment-scale SWAT+ model was successful in predicting mean monthly flow rates, mean monthly nitrate-N loads and mean annual nitrate-N loads in Arawhata stream. However, there is some degree of uncertainty under these estimations, as the exact vegetable crop rotation pattern information in the Arawhata catchment could not be sourced (if there is any), and tentative modelling scenarios had to be used instead (Table 5.7, Figure 5.8). In this context, the use of hydrologic models becomes challenging when it comes to situations where there is a high turnover of land use. Furthermore, due to limited data availability, only a calibration process could be undertaken. With this in mind, the results and interpretations of the modelling conducted in this study must be taken with caution. Nonetheless, there is great scope for using SWAT+ model to predict nutrient losses from agricultural areas, especially with a growing availability of high-resolution soil and land use map coverage and observations of flow and water quality in streams and rivers in New Zealand. However, a more intensive monitoring of field-scale nitrate-N leaching and its potential attenuation in subsurface environment is suggested to provide a better comparison and reduce uncertainty in the catchment-scale modelling of nitrate-flow pathways (Elwan, 2018; Collins et al., 2017; Rivas, 2018).

The coupling of field-scale models that had previously estimated nitrate-N leaching losses, though somewhat complicated, was also successful in catchment-scale modelling using SWAT+. This is an improvement compared to the previous SWAT versions, where no intervention was possible within the model. This incorporation of nitrate-N losses from field-scale models into catchment-scale SWAT+ has not been done before. Ghebremichael et al. (2022) used a similar technique in SWAT 2012 to the one used in this study, but point source objects, representing pesticide percolation from unique combinations land use and soils, were only outflowed to streams and not as percolation to aquifers, as this is not a feature of older SWAT versions. This is a significant improvement in up-scaling of field-scale nitrate-N leaching simulations (by the models like APSIM) by coupling with the catchment-scale SWAT+ modelling, accounting for nitrate-N flows and its potential attenuation in different flow pathways (via soil percolation and groundwaters).

In this study, though the SWAT+ model's nitrate-N load prediction was considered satisfactory, there is still room for improvement. For instance, in the calibration process, SWAT+ offers limited capability for automatic calibration or uncertainty analysis, such as in the SWAT-CUP of previous versions. A robust analysis of the model's uncertainty is a very important step into building a robust model that has not yet been addressed properly in the newly develop version of SWAT. Additionally, SWAT+ has been particularly developed to model surface flow, and models groundwater movement through several assumptions that might not represent reality accurately (Bailey et al., 2020a). This could explain why some flow predictions in the year 2022 mismatch the flow observations, as this year had exceptionally large rainfalls and the effect of groundwater in the streamflow could have been underestimated. Only recent efforts have been made into integrating SWAT+ with more complex and representative models that estimate groundwater transport, such as MODFLOW or gwflow (Bailey et al., 2020a; Bailey et al., 2020b).

The half-life parameter of nitrate-N in the catchment (800-1100 days) was within the range reported by other authors (Almasri & Kaluarachchi, 2007; dos Santos et al., 2020; Klein et al., 2013; Puckett et al., 2011; Rivett et al., 2008), but at the higher end of the range. This could suggest that most of the denitrification process was considered in previous field scale models, and groundwater in the catchment was simulated with low nitrate-N attenuation capacity. On the other hand, results from the nitrate-N budgets suggested that the calculated mean nitrate-N attenuation factor of the catchment ranged between 5-9%, which is relatively low when compared to the 14-94% obtained by Elwan (2018) for the Rangitikei and Tararua catchments. This low nitrate-N attenuation factor could be explained by the presence of permeable materials (oxidised sand and gravels) found as main sediments under the soils of the catchment, which decreases the residence time and resulting denitrification (Elwan, 2018). Further research into mapping and modelling spatially-variable nitrate-N attenuation factors (nitrate-N half-life parameter) of different aquifers, such as done in the coupling of Overseer and SOURCE model by Legarth et al. (2022), should be considered to refine the SWAT+ model's estimations.

5.5.3 Impact of in-field management measurements

Considering various sources of uncertainty identified, the results and interpretations of the modelling conducted in this study must be taken with caution. However, the ability of the coupled field-scale nitrate-N leaching (simulated by a combination of APSIM for the vegetable areas and Overseer for the pastoral areas) into the SWAT+ offers a suitable tool to simulate potential effects of various in-field mitigation scenarios on reduction of nitrate-N loads at catchment scale.

In Arawhata catchment, a shift in crop rotation composition in green vegetable crop areas, from heavily based in brassicas rotations (broccoli and cauliflower) to crop rotations that need less N fertiliser and consider the frequent use of cover crops, such as maize and ryegrass, was predicted to decrease mean annual nitrate-N loads by 17 – 21% in Arawhata stream. These results are consistent with findings of Zemek et al. (2020), who suggested that brassica crops normally produce the largest nitrate-N leaching losses from a range of vegetable crops, due to its large N inputs and low N use efficiency. Furthermore, as discussed in the Chapter 3 and 4, this type of rotation usually produces a large amount of residues with low C:N rations, resulting in high N mineralisation rates.

According to the model's predictions, the incorporation of cover crops during fallow periods in areas with a potato-onion rotation did not result in significant reductions in nitrate-N loads in Arawhata stream (Figure 5.16). This was attributed to the relatively smaller area (125.6 ha) that this type of land use covers compared to the green vegetable crop rotation land use area (372 ha) (25 and 75% of the total outdoor vegetable crop land use, respectively). Similarly, a reduction of 10 or 20% in fertiliser applied to vegetable crops did not produce a significant reduction in nitrate-N loads in Arawhata stream, which could suggest that N fertiliser rates simulated in APSIM match the crop demand accordingly. However, this assumption needs further validation with more in-field surveys and observations of crop N fertilization rates in the study area.

Results from the simulated in-field crop and fertiliser management scenarios of this study are relatively consistent with other's work. For instance, Misaghi et al. (2020) simulated the effect of reducing 50% of N fertiliser using the SWAT model in the Zanjanrood catchment, Iran, for years 1996-2013, obtaining a reduction of 17% of nitrate-N leaching in vegetable and arable crops (alfalfa, wheat, onion, and barley). In the present study, reducing the use of N fertiliser by 20% resulted in 6-8% of nitrate-N load reduction in the Arawhata stream.

5.6 Conclusions

The hydrology and nutrient transport through the catchment were correctly identified and characterised in this study, both qualitatively and quantitatively, and with the aid of the SWAT+ model, concluding that most contributions to Arawhata stream are through the groundwater and subsurface flow. The coupling of APSIM-Overseer-SWAT+ was successfully calibrated to represent the flow and nitrate-N loads of the Arawhata stream.

Nitrate-N attenuation factors estimated using the SWAT+ model predictions indicated that the Arawhata catchment has a low capacity to reduce nitrate-N loads, which was mostly attributed to the low residence time of the nitrate-N in the groundwater, indicated by the presence of oxidised sand over gravels and oxidised gravels below soils.

The coupling of field-scale models, such as APSIM Next Generation and Overseer, with catchment scale models, such as SWAT+, is here demonstrated to be a useful tool to quantify the effects of different in-field management strategies to reduce N losses in the Arawhata catchment. A change in crop rotations that are less oriented to cultivating brassicas showed to be the most effective single practice among those evaluated in this study (reduction of 17 – 21% of mean annual nitrate-N load in Arawhata stream). Combining this change in crop rotations with a period of ryegrass cover crop in areas of the catchment with potato-onion rotation led to a reduction of 21-25% of mean annual nitrate-N load in Arawhata stream. Other practices, such as reducing N fertiliser application rates, resulted in a reduction of 1-8 % of mean annual nitrate-N load in Arawhata stream.

Chapter 6: Synthesis and recommendations

6.1 Introduction

One of agriculture's greatest challenges is the tensions between meeting global food demand while maintaining nutrient levels in soils, and protecting biodiversity in rivers, lakes and groundwater (Tilman et al., 2002; Davidson et al., 2015; Houlton et al., 2019; Sutton et al., 2013). Agriculture is considered to be the main source of diffuse pollutants of the aquatic environment (Fageria & Baligar, 2005; Stewart & Lal, 2017; Wang & Li, 2019). In the case of N, large losses occur when high concentrations of N in the soil coincide with large drainage events (Di & Cameron, 2002). Several factors related to climate, crop, soil and N management, described in detail in the Literature Review (Chapter 2), influence the way N is leached to water bodies. These nutrient losses to the environment can cause several adverse effects, such as: eutrophication, loss of biodiversity, loss of recreational value, and health issues if water is ingested (Cameron et al., 2013; Grizzetti et al., 2011; Schullehner et al., 2018).

Among different agricultural land uses, intensive vegetable production is recognised as one of the largest producers of N leaching losses (Di & Cameron, 2002). These large N losses have been linked to intensive N fertiliser and water inputs, low crop nutrient use efficiency, and large residual N and mineralisation rates (Agneessens et al., 2014; Cameira & Mota, 2017; Chaves et al., 2007; De Neve & Hofman, 1998). In this regard, the Literature Review (Chapter 2) suggested that research results that quantify the amount of N lost through leaching from these systems is lacking in the New Zealand context and so there is limited understanding of the magnitude of nitrogen leaching under vegetable crops.

For many years, Lake Horowhenua in the Manawatu-Whanganui region has had major problems with eutrophication and it is currently one of the most degraded lakes in New Zealand (Horizons Regional Council, 2017, Land Air Water Aotearoa, 2022). The main tributary of the lake, the Arawhata stream, is surrounded predominantly by intensive vegetable crop and dairy farms, which are suspected to be the major contributors of nutrients to both the stream and lake (Gibbs, 2011; Thomas & Gibbs, 2014; Horizons Regional Council, 2017). For this reason, Horizons Regional Council have established the 'One Plan' strategy (Horizons Regional Council, 2014). This has become particularly important for growers of the Arawhata catchment due to difficulties in minimising N leaching losses.

Several tools have been developed to assist growers to reduce N losses to the environment. Some process-based models provide a better understanding of the fate of nutrients in the soil profile, including leaching under specific conditions, while others can be used to estimate nutrient loads in rivers and lakes, both of which are essential to the decision-making process and the establishment of policies. As measuring N leaching can be a difficult task, particularly under complex and dynamic systems such as vegetable crops on free draining soils, the use of models is a valuable tool. Linking different agricultural scenarios and the use of GMPs with field scale and catchment scale models may help identify the most effective mitigation strategies to help improve water quality in the Arawhata stream and Lake Horowhenua. Therefore, this study set out to develop a robust modelling procedure for use in the Arawahata catchment. The present study focused on: (1) providing measurements of nitrate-N losses under vegetable crop systems and (2) modification of a field scale model to investigate a number of mitigation strategies to decrease nitrate-N leaching from the soil profile and (3) evaluate catchment scale models coupled with field-scale models to assess the impact of mitigation strategies on the receiving river. The different sections of this chapter present a summary of the findings of this thesis work and an integrated discussion about their implications.

6.2 Quantification of N under intensive vegetable crop farming

This study identifies the quantity and timing of the majority of nitrate-N leaching losses under a range of vegetable crops in the Horowhenua region. The results suggested that mean nitrate-N amounts in the soil may range between 61 to 78 kg nitrate-N ha^{-1} during the growth period of potatoes in the 0-60 cm depth, and 39 kg nitrate-N ha^{-1} in fallow periods of green vegetable crops in the first 20 cm of soil depth. When using drainage volumes predicted by a soil water balance (from the APSIM model, Chapter 4) and nitrate-N concentrations measured in samples from suction cups at 60 cm depth, mean nitrate-N leaching losses under potato crops were 65 kg nitrate-N ha^{-1} during the potato fallow period and 30 kg nitrate-N ha^{-1} under the subsequent onion crop, adding up to 95 kg nitrate-N ha^{-1} for the period of May 2021 to February 2022. At a green vegetable site, an overall mean leaching loss of 201 kg nitrate-N ha^{-1} yr^{-1} was estimated using lysimeters at 60 cm depth. Nitrate-N leaching losses and concentrations were remarkably variable in the present study; however, similar variability has also been found in similar studies (Chapter 3). Spatial variability may not only be due to differences in factors such as crop growth and soil properties, but also the result of uneven distribution of N fertiliser application or residues from previous crops across the field (Francis et al., 2003).

Mean nitrate-N concentrations in drainage water monitored in suction cups and lysimeters at 60 cm were greater than the New Zealand Drinking Water standard (NZDW) (11.3 mg nitrate-N L^{-1}) in all treatments and years. Overall mean nitrate-N concentrations monitored in shallow (1 m) and deep (5 m) piezometers were also greater than the NZDW, including a shallow piezometer with a mean almost three times this limit (31.6 mg nitrate-N L^{-1}). This implies that neither shallow groundwater nor deep groundwater are suitable for drinking in these areas.

Data measured in this study indicated that nitrate-N distribution in the soil can be strongly influenced by weather. Under similar management conditions in the potato growing field, soil nitrate-N in the first year was mostly concentrated in the first layers of the soil, while in second year it was more homogeneous throughout the profile. This was attributed to intense rainfall events that occurred shortly after the potato sowing and fertiliser application in the second year. Nitrate-N leaching losses mostly occurred in the winter-spring drainage season, which in turn is related to the amounts of rainfall and evapotranspiration. As weather and soil properties are quite variable, it is often difficult to compare drainage and N leaching amounts between different studies of vegetable crops (Francis et al., 2003).

Results from these trials suggest that different rotations and crops can produce different amounts of N leaching. A key factor influencing this difference was attributed to the residual soil N concentration between crops. Crop rotations with larger nutrient losses often had approximately 34 ppm in the 0-60 cm soil depth, and rotations with smaller nutrient losses were more likely to have approximately 7.2 ppm in the 0-60 cm soil depth. In other words, findings from this study indicate that, when aiming to decrease N leaching under a crop, it is critically important to account for initial soil N concentrations in the N fertiliser programme.

The statistical analysis performed here demonstrated that there was no significant yield improvement when larger quantities of N fertiliser were applied; however, large N fertiliser rates significantly increased the nitrate-N concentrations found in drainage water at 60 cm depth. These results provide evidence that demonstrates to vegetable crop growers that excessive rates of N fertiliser are not necessary to achieve optimum crop yields, and efforts should be placed in meeting N crop demand at the right time instead. Furthermore, greater N application rates pose a risk to the quality of receiving waters.

N fertiliser application rates typically used by growers were studied in the experimental trials reported here and they also fall within the national recommended range for the corresponding crop (and in some cases were even lower) (Reid & Morton, 2019). Large nutrient losses were mostly associated with the N mineralisation of crop residues, as there were no N applications during the winter-spring seasons and vegetable crop residues tend to have small C:N ratios. Findings from this study demonstrated the importance of avoiding fallow periods in vegetable crop systems, especially during the drainage season, if the risk of nutrient losses is to be reduced.

Results reported here for the green vegetables crops site in the lysimeters could reflect one of the worst N leaching scenarios for growers of the area, i.e., where large amounts of N pre-planting were estimated, whole crops had to be reincorporated due to the COVID pandemic, which was followed by an extended fallow period leading to large amounts of N leaching. Despite the special circumstances associated with the COVID-19 pandemic, the incorporation of complete crops may not be that rare in a volatile market (NZIER, 2019), where the costs of product harvesting can sometimes surpass the sales profit, making it unviable to harvest, and incorporation of the crop is the least costly option. In addition, large yield variability in the field and high-quality standards from distributors and retailers often lead to large percentages of the crop being discarded, increasing crop residues in the field and/or surrounding areas. Efforts to reduce fallow periods in the region have been increased since 2019 (Bloomer et al., 2020), but the ability to grow catch crops during the drainage season is highly dependent on soil moisture content with fieldwork often being limited by waterlogging. The chance of all these risk factors coinciding is not yet well understood in the Arawhata catchment.

The impact of in-field mitigation techniques studied in this work was variable depending on the practice. Smaller nitrate-N concentrations in soil were found when a controlled release N fertiliser was applied at the same rate as a conventional N fertiliser, but yields under the CR fertiliser were also significantly lower. A possible explanation for this was the lack of available soil N in the initial stages of crop growth. Some studies have suggested that a small complementary application of conventional N fertiliser at planting could help overcome this shortfall of soil N (Reid & Morton, 2019; Wilson et al., 2010). Other in-field mitigation strategies investigated here did not reduce nitrate-N leaching losses significantly (such as chicken manure) or performed poorly in terms of yield (such as split liquid fertiliser applications). Thus, potential strategies to reduce N leaching must be carefully studied and planned before implementation, including a consideration of the cost/benefit ratio of crop production.

6.3 Use of APSIM New Generation to estimate field scale N losses

APSIM Next Generation simulates nutrient losses under crops in a much more frequent time-step and in more detail than other field scale models. The use of manager scripts allows flexibility to the model to simulate a range of scenarios and provide more reliable predictions. The amount of data provided to the model determines the accuracy and quality of APSIM's prediction, therefore, detailed predictions require more and more detailed input information. Many of the default and recommended parameters from APSIM are based on Australian conditions, therefore, some of these parameters need to be modified to better reflect New Zealand conditions. As such, this thesis work could serve as a guide to the parameterisation of APSIM Next Generation in, at least, the lower North Island. Furthermore, comparisons between measured and predicted data presented here helped to evaluate whether APSIM could be a suitable model to predict nutrient losses in vegetable crop fields. The Literature Review (Chapter 2) showed that only a handful of studies have been conducted in New Zealand to test APSIM's ability to predict nutrient losses in market gardens.

In Chapter 4, APSIM's predictions were compared with detailed measurements in two vegetable crop fields in the Horowhenua region. In terms of the water balance, APSIM had good to very good predictive power, with *NSE* coefficients for simulation of soil moisture ranging from 0.35 to 0.8, and an overall mean *NSE* of 0.6 when considering all N fertiliser treatments in all years and sites. Drainage volumes were only measured at the Green vegetables site from lysimeters at 60 cm depth, and comparison between this data and APSIM's predictions showed *NSE* coefficients ranging from 0.31 to 0.7, with an overall mean *NSE* coefficient of 0.51. Consequently, the field scale water balance simulated by APSIM Next Generation was considered reliable, and parameters that might be difficult to measure, such as drainage volumes, can be estimated relatively well by this model.

APSIM Next Generation's estimations of nitrate-N concentrations in soil was considered acceptable. *NSE* performance coefficients for soil nitrate-N concentrations were highly variable and ranged from -6.85 to 0.84. Possible limitations in the representativeness of the observed data (i.e., only the soil under the ridge was sampled in the first year) could have compromised any potential agreement between measured and simulated values. The overall mean *NSE* increased to -0.45 when measured data from the second season was compared with predictions. Although both of these overall means were negative, they were greater than the limit of -1.0 considered in this study (and suggested by many others, as indicated in the Chapter 4) to separate a poor from an acceptable prediction of nutrient concentrations. APSIM's ability to estimate nitrate-N concentrations in the soil was better at the site with green vegetable crops than at the site with the potato-onion crops. This was attributed to the frequency of sampling and variability of the measured data; nitrate-N concentrations at the Green vegetables site were measured only before and after each crop cultivation, whilst in the potato-onion rotation site, concentrations were also measured during the crop growth period where N cycling would have been more dynamic and less homogeneous (i.e., due to recent fertiliser application).

APSIM was unable to accurately predict nitrate-N concentrations in drainage water, with *NSE* coefficients ranging from -0.1 to -7.33 and an overall mean of -2.28 in the field cultivated with potatoes. This seemingly poor performance probably had less to do with any shortcoming in APSIM but was mostly associated with the large within-treatment variability in each sampling date of the observed data.

Results from the Chapter 4 suggest that the APSIM model can be used to estimate cumulative annual nitrate-N leaching losses under crops and its performance was generally scored as satisfactory. The minimum NSE coefficient obtained in Chapter 4 was 0.15 and the maximum was 0.85, with an overall mean NSE coefficient of 0.53 across all treatments, years and sites. In the field trials reported here, a range of annual leaching losses was found, with values from 58 to 240 kg nitrate-N ha^{-1}, this range will reflect different periods of times, soil depths, crops and fertiliser treatments. Comparisons with other studies from the same area indicated that the predicted and observed values were within the normal range (Norris et al., 2017).

Although APSIM was generally able to predict a number of key variables, there were cases where APSIM had difficulties in matching the observed data. For instance, soil and drainage nitrate-N concentrations and leaching losses in plots with no fertiliser were usually underestimated by the model, which was attributed to an overestimation of potato yields for this treatment, and to previously reported limitations in APSIM's simulation of N mineralisation and residue decomposition (Sharp et al., 2011a; Sharp et al., 2011b; Vogeler et al., 2019; Vogeler et al., 2020). The model also tended to over-estimate nitrate-N leaching losses from experimental plots with excessive applications of N. This may have been a consequence of discrepancies in denitrification between measured data and simulations (Van Groenigen et al., 2010).

It was concluded that APSIM Next Generation was helpful in providing explanations for differences in a range of variables between treatments, sites and years i.e., it tracked changes in fertiliser rates, N crop uptakes, leaching losses and other inputs and outputs. APSIM can, therefore, be used to explore the relative effectiveness of mitigation measures over a range of years with varying climatic conditions.

Findings from this study underline the importance of having a sound knowledge of when drainage might occur as this informs the design and selection of mitigation practices. For example, according to APSIM simulations, treatments that targeted the form and timing of N delivery during the crop season (split fertiliser application, liquid fertiliser application and controlled release fertiliser) did not reduce nitrate-N leaching losses significantly because there were no drainage events during the crop growing period, and by the start of the drainage season, all fertilised treatments had similar amounts of N in the soil.

APSIM was used to investigate the influence of crop rotation on nitrate leaching at a depth of 100 cm. Nitrate-N leaching losses in the long-term (1991 to 2021) were simulated for a selection of different crop rotations which were considered to be representative of the area. Results indicated that mean annual nitrate-N leaching losses in the potato-onion rotations were approximately 47 kg ha^{-1}, and losses under different green vegetable crops ranged between 41 and 117 kg nitrate-N ha^{-1}, with an overall mean of 79 kg nitrate-N ha^{-1}. Data analysis from these long-term results indicated that the monitored years in this study were amongst the ones that had greater N leaching losses.

The depth at which nitrate-N leaching is said to be 'lost' as leachate is an important consideration. For instance, in the Chapter 4, APSIM Next Generation predicted different nitrate-N leaching losses depending on the depth to which they were estimated, which suggests that nitrate-N was still subject to a range of processes (such as crop uptake or denitrification) in the subsurface environment. The depth at which N leaching losses are estimated was also found to vary between international studies, as discussed in the Chapter 3. Therefore, policies seeking to establish maximum cumulative N leaching losses under vegetable production need to specify the rootzone depth and take into account the subsurface environment (factors such as maximum and minimum level of the shallow water table, redox conditions, permeability, among others).

6.3.1 Recommendations to the use of APSIM in simulating vegetable crops

APSIM requires extensive parameterisation. The use of APSIM in this study revealed some critical parameters that need careful consideration: the following list of learnings and recommendations are offered to help future users of APSIM to simulate leaching losses under vegetable crops.

- Initial nitrate-N concentrations in the soil were demonstrated to be critical for the following nitrate-N leaching losses from the field, and so comprehensive field sampling and analysis is recommended. However, these concentrations may not be representative of the long-term value within a rotation, and therefore it is recommended to set this parameter with the most 'informed opinion' possible.
- SCRUM-APSIM module, the main sub-model in APSIM Next Generation which simulates vegetable crop growth, has been developed from OVERSEER model estimations and a sigmoidal crop growth function. This model is highly general and needs detailed and realistic yields to project crop N demand and growth.
- The Potato module uses a highly sensitive parameter denominated as "stems per seed", which will impact on the resulting yield and N uptake. A correct estimation of this parameter is required, as different N application rates and timing of application might have an influence on this parameter.
- When evaluating the performance of APSIM Next Generation in predicting soil nitrate-N and nitrate-N concentrations in drainage water, it is recommended to use

various statistical coefficients to gain a better idea of performance, instead of basing criteria on only one statistical test. In this study, the use of *NSE* alone would likely have had limitations, as soil nitrate-N and water nitrate-N concentrations are, many times, more variable in space than in time.

- Spatial variability can also be highly problematic when comparing model's predictions and observed nitrate-N concentrations. As vegetable crops make use of beds and ridges, concentrations in the soil after N fertiliser application are highly uneven. Therefore, it is recommended to allow some time after the use of N fertiliser before soil sampling, which may vary according to the product and the weather conditions.

New and more detailed information for users about APSIM's parameters can also be found in Cichota et al. (2021).

6.4 Use of SWAT+ to upscale field scale estimations of N losses and simulate mitigation scenarios

The Soil and Water Assessment Tool Plus (SWAT+) model is a widely used eco-hydrologic catchment-scale model. Like APSIM, it requires numerous input parameters for reliable predictions. SWAT estimates nutrient losses from the rootzone with little scope to modify or intervene in soils processes inside the model. Given that SWAT was developed in the USA, and that it employs very simple algorithms for its predictions of N leaching from the soil profile, its estimates of N leaching in the Arawhata catchment are likely to be unreliable. Fortunately, the recently restructured version of SWAT+ allows the user to indirectly input estimated nutrient losses and simulate their impact at a catchment scale. In a novel approach developed in this study, more reliable simulations of field scale nitrate-N losses (APSIM and Overseer) have been inserted into the SWAT+ model to investigate the impact of land use and management practices in the Horowhenua region on water quality in the Arawhata stream.

Some previous water balances and modelling of the Arawhata catchment have broadly characterised water flow in the area (Thomas & Gibbs, 2014; Thomas & Garden, 2021). Nonetheless, no clear specific flow pathways were identified before this study. In Chapter 5, flow rate and N concentrations were estimated at several strategic points (culverts) of the Arawhata catchment. These results suggested that N contributions in surface runoff were minimal and the main source of recharge of this stream was through shallow groundwater and deep groundwater. These findings agree with previous studies that suggested that a Basement High, situated underneath the Arawhata stream, could be forcing groundwater to flow towards the surface, forming both the Arawhata stream and Lake Horowhenua. Results suggested that flow and nitrate-N contributions of these culverts may occur in very intense storm events that may last for only a few hours. High variability, as indicated by the large standard deviation, in nitrate-N concentrations was found in one of the sampling points located near dairy farms.

The water balance of SWAT+ was correctly calibrated by using average monthly observations of flow rate of the Arawhata stream. The performance parameters for the comparison of these two variables were 0.68 for *NSE* and 0.74 for R^2 for years 2017-2022, indicating that SWAT+ was able to predict flow rates in the Arawhata stream.

A major problem was encountered when attempts were made to simulate nitrate-N losses under vegetable crops across the catchment using APSIM. Growers are reluctant to share information about the crop rotations that they employ. Therefore, there is limited information about the area of specific crops or crop rotations in the Horowhenua. To address this lack of information, a farm consultant with working knowledge of crop rotations was interviewed. This person was able to identify representative or typical rotations employed in the Arawhata catchment. The most common crop is brassica. Two models were created to weigh different leaching losses according to the most produced crop (brassicas). In one model it was assumed that brassica crops accounted for 65% of nitrate-N leaching losses under vegetable crop rotations, and in the other model the brassica crops accounted for 80% of nitrate-N leaching losses from vegetable crops. These two models provided two different values of the ability of the catchment to attenuate nitrate-N in the groundwater, which were then used for simulating the effect on water quality of different agricultural practices.

Nitrate-N attenuation estimations from SWAT+ reported in the Chapter 5 (nitrate-N half-life of 800-1100 days) were found to be relatively low. As sand and gravel are the predominant rock materials in the subsurface environment (Chapter 3), low residence time and low N attenuation were expected in this catchment (Elwan, 2018). SWAT+ was deemed as helpful to an approximate assessment of the nitrate-N attenuation capacity of the Arawhata catchment.

Coupling field-scale models with SWAT+ was useful to predictions of nitrate-N leaching losses from soils and river loads. Comparisons of mean monthly nitrate-N loads (considering years 2017-2022) between predicted and observed data showed *NSE* values of 0.68 to 0.71, and R^2 values of 0.7-0.73. Mean annual observed and predicted nitrate-N loads in the Arawhata stream were also similar. Though vegetable crop areas only constituted 29% of the catchment, N contribution from these areas to the N loads of the stream were significant.

Simulations from SWAT+ could work as a tool to help achieve water quality objectives of the Lake Horowhenua and the Arawhata stream. The impact of selected in-field mitigation strategies were evaluated at catchment-scale in SWAT+, producing N reductions ranging from 1% to 21% from single practices. Even greater N reductions can be achieved when these practices are combined. Coupling SWAT+ and catchment scale models can therefore aid in the quantification of nutrient losses and the effectiveness of mitigation scenarios It can, therefore, work as an instrument to facilitate the development and implementation of policies. Some caution is needed as nutrient losses in long-term simulations in situations where land use is highly complex and dynamic is a major challenge for hydrologic models (Samarinas et al., 2020).

6.5 Applicability of APSIM Next Generation and SWAT+ in other locations of New Zealand

Although vegetable crops are only grown in a few areas of New Zealand, the impact of this land use on the water quality of lakes and rivers can be substantial. The other main areas that produce vegetables in New Zealand include Auckland (with areas such as Pukekohe and Matamata), Canterbury, and, to some extent, Bay of Plenty (Horticulture New Zealand, 2017). Currently, a number of institutions are parameterising and using APSIM to predict nutrient losses from vegetable farms in most of the aforementioned areas (except Bay of Plenty) under the programme 'Sustainable Vegetable Systems', including Manawatu-Whanganui and Hawkes Bay areas (Potatoes NZ, 2021).

Some limitations of APSIM could play a role in certain areas and should be considered when using this model in other locations in New Zealand. For instance, APSIM does not have the ability to predict ammonia volatilisation and might not be completely suitable for soils that have an alkaline pH. APSIM also lacks the ability to simulate nutrient losses in surface runoff, and therefore, may be of limited use for sloping fields or soils that are regularly waterlogged. In addition, some crops have not yet been incorporated into the crop modules, and on occasions, simulations have to be conducted using the most similar crop available.

The SWAT+ model was also found to be a flexible model that can be adapted to many situations and can be potentially used in other areas of New Zealand. Constrains in the use of SWAT+ are more related to the availability and quality of the large quantity of information required for parameterisation than to the model *per se*. Catchments that include highly artificial landscapes may also be represented wrongly by SWAT+ (i.e., several reservoirs, dams, artificial drainage and sewage, etc).

Literature suggests that APSIM model is useful to simulate crop and soil variables under different climate change scenarios (Tang et al., 2020; Teixeira, de Ruiter, et al., 2018). Therefore, these scenarios could theoretically be up-scaled to catchment systems of New Zealand by coupling both APSIM and SWAT+ models.

Several challenges may be encountered when parameterising and calibrating these models. However, once achieved, the combined use of APSIM-Overseer-SWAT+ becomes more straightforward.

6.6 Financial implications of using in-field mitigation strategies

The costs and benefits associated with in-field mitigation strategies can be quite variable and highly dependent on the conditions that they are used in. There are relatively few in-field mitigation strategies to reduce N leaching under vegetable crops and they are generally low-cost (McDowell et al., 2013). A rigorous analysis of the costs and benefits of using in-field mitigation measures to reduce nitrate-N leaching losses is beyond the scope of this thesis so what follows is a simple evaluation of some of the financial implications of the findings of this research.

As an example, in the case of the potato-onion rotation site of this study, the farm could store many tonnes of potatoes. However, when no shed is owned, the construction of one could become expensive and might not be affordable to all growers (a hypothetical 75 m^2 shed could cost approximately $25,000 without including detailed materials). The early harvest of potatoes and subsequent sowing of ryegrass (as cover crop) was predicted by APSIM to reduce nitrate-N leaching losses by approximately 47% (or 22 kg N ha^{-1}). The cost of ryegrass seeds (to use as a cover crop for incorporation) is approximately $77 ha^{-1} (Cridge seeds, 2022), and the cost of cultivation, sowing and spraying totals $345 ha^{-1} (Reynish, 2020). Assuming a reduction of 22 kg N ha^{-1}, this gives a cost of $19 kg N reduced ha^{-1}. If all the hypothetical N mitigated (22 kg N ha^{-1}) is assumed to be incorporated in the soil and made available for the next potato crop, and assuming a price of $2.94 kg N^{-1} as urea fertiliser (Ravensdown, 2022), the total cost of reducing 47% of N (or 22 kg N ha^{-1}) would be approximately $357 ha^{-1}. The price of 1 kg N ha^{-1} saved from N fertiliser application is highly relative to the type of N fertiliser applied and the effective amount of N available after the incorporation of ryegrass.

If in the example above, a catch crops such as beetroot is selected (which may use similar harvesting machinery as potatoes and onions), APSIM estimated a reduction of 7% in N leaching losses. Beetroot is normally cultivated in 1-m beds of 8 rows; with 0.3 m of spacing between beds and 0.2 m between plants. A population of 310,000 beetroot plants ha^{-1} will cost $2,900 for beetroot seeds. Spraying, cultivation and sowing costs were assumed to be similar to fodder beet and could round \$999 ha^{-1} (Reynish, 2020). For an estimated market price of \$0.67 kg^{-1} for beetroot (Horticulture New Zealand & Plant and Food Research, 2020) and a conservative marketable yield of 36 t ha^{-1} (Reid and Morton, 2019), estimated profits could approximate \$20,220 ha^{-1}.

In the case of the Green vegetables site, the reduction in N fertiliser application in the crop rotation is likely to result in reductions in costs without compromising yields. For instance, soil N in the 0-20 cm depth in the Green vegetables site were nearly 100 kg N ha^{-1}, and N fertiliser recommendations for this amount of N in the soil are about 60 kg N ha^{-1}. The difference between the amount applied by the grower and the amount recommended is 57 kg N ha^{-1}, which would be estimated in a cost reduction of approximately \$330 ha^{-1}. Therefore, the measurement of residual N before crop sowing could help reduce both costs and nitrate-N leaching losses.

6.7 Future research considerations

- *Increase in soil data base and physical parameters*

The models used here need many soil parameters to estimate water and nutrient balances. Some of the data used to parameterise the models date from the 1980s. These values may be less representative of current soil properties given the management and land use changes that have occurred since these measurements were made, which might have affected soil properties. Furthermore, the soil map used in the current study was also from the same date. A more recent and detailed soil database is essential to more reliable modelling.

- *Weather predictions and climate change scenarios in vegetable crop production*

APSIM Next Generation has the potential to estimate the change in crop yields and environmental outcomes resulting from changes in climate. Though there is currently some work being done on the likely impact of climate change, further detailed research is urgently required to identify the necessary actions and to develop policies. SWAT+ also has the ability to estimate flow rate changes and nutrient concentrations in rivers under different climate scenarios.

- *Validation of predictions pre- and post- use of mitigation strategies*

A proper validation of the estimations of nutrient leaching losses before and after the implementation of in-field mitigation strategies has not been yet done in APSIM Next Generation or SWAT+.

In using APSIM Next Generation

- *Incorporation of soil and landscape changes into the model*

Fertiliser management and similar practices can be modelled, but there is a range of other in-field mitigation technologies such as tillage management that cannot currently be simulated in APSIM. Similarly, as mentioned earlier, APSIM is not able to simulate nutrient losses in runoff, which depending on the situation might become important.

- *Incorporation of dynamic cultivation areas and plant synergistic effects*

In-field mitigation managements that focus on interrow cropping and that encourage synergistic effects between plants cannot be modelled in APSIM: future development of the model to predict these effects could be valuable.

- *Further research into analysis of mitigation measurements*

Due to time constrains, other in-field mitigation measurements, such as leaving vegetable crops residues to resprout during the drainage season, the incorporation of straw and carbon-rich materials to remove mineral N and slow down mineralisation, and other similar practices could not be evaluated in APSIM in the present study. A detailed analysis of the impact of the implementation and costs of using different mitigation strategies to reduce nitrate-N leaching losses in vegetable crop farms in New Zealand is necessary. This could help growers and stakeholders to make more informed decisions.

- *Development of the SurfaceOM model*

Results of this and previous studies (Sharp et al., 2011a; Sharp et al., 2011b; Vogeler et al., 2019; Vogeler et al., 2020) have indicated that APSIM may misrepresent the mineralisation rates in cool climates. Furthermore, the physical form of residues, whether chopped or whole, left on the surface or incorporated, is not considered and the decomposition rates are essentially the same for all forms.

In using SWAT+

- *Account for temporal changes in land use*

As discussed previously in this chapter, it is not possible to evaluate changes in land use in SWAT+ (i.e., it is a fixed input in the model), which can become an important source of variation in the model. Integration of such changes into the model could considerably improve the model's utility.

- *Increase frequency in the measurement of flow and nutrient concentrations in rivers*

A lack of river flow and water quality data was an important limitation in the use of SWAT+ during this study. Nitrate-N concentrations in the Arawhata stream were measured only once a month for five years. This is likely to be the case for many other streams and rivers in New Zealand. An increase in the sample size of flow and nutrient concentrations in water bodies can substantially improve the calibration and validation of the model.

- *Monitoring of subsurface redox conditions and the link to the attenuation factor of SWAT+*

The measurement of redox conditions in boreholes, and the analysis of rock types of the catchment can help identify the attenuation capacities of a catchment. This can potentially be incorporated into the SWAT+ model as the spatially variable 'nitrate-N half-life' for each aquifer identified by the model. This might produce more accurate predictions of nitrate-N loads in lakes and rivers.

- *Better integration of groundwater models such as MODFLOW and RT3D-MS into SWAT+*

Previous versions of SWAT+ had an interface to MODFLOW and RT3D to simulate nutrient transport in the subsurface environment. More accurate nutrient losses predictions from SWAT+ can result from linking these two models along with the use of field-scale models such as APSIM and Overseer.

6.8 Conclusions

Important findings from the current study, which are in line with the main hypothesis, include:

- Nitrate-N concentrations in soil and losses in water were quantified under different vegetable crop rotations and soils at two field sites in the Arawhata catchment. Large concentrations were found in the topsoil (0-30 cm) (9.3 to 18.3 ppm), subsoil (30-60 cm) (7.3 to 9.6 ppm), and soil water (16.9 to 61.9 ppm). Nitrate-N leaching losses were also large, with values ranging from 95 to 225 kg nitrate-N ha^{-1}.

- APSIM Next Generation model was successfully calibrated and validated using the results measured in the fields. This model was able to predict soil moisture content (*NSE* values often >0.35), drainage depth (*NSE* from 0.31 to 0.7), nitrate-N concentration in the soil (*NSE* from -6.85 to 0.84), nitrate-N concentration in the soil water (*NSE* from -7.33 and -0.10), and, importantly, cumulative nitrate-N leaching (*NSE* from 0.15 and 0.85).

- APSIM Next Generation was deemed to be a suitable tool to identify and assess the likely long-term (over 30 years). N leaching rate at the filed sites, and to explore the effect of soil type on N leaching under these rotations. APSIM was also able to quantify the ability of a range of in-field mitigation strategies (change in soil type, potato harvest at maturity followed by a ryegrass crop during the drainage season, potato harvest at maturity followed by a winter catch crop, potential of drainage management, other alternative cover crops and catch crops instead of ryegrass, “in-season” versus “out-of-season” production, and alternative crop rotations) to reduce N leaching from intensive vegetable farming in the Arawhata catchment. The reduction from these scenarios ranged from 6 to 52% compared to the normal practice of growers of the area.

- A catchment-level model was developed to simulate the impact of vegetable farms in the catchment on the N load in the Arawhata stream. This model coupled together APSIM/Overseer with SWAT+. The combined model was successfully calibrated and validated using observed values from the Arawhata stream. Predictions of flow rate of the stream from the model had a satisfactory performance (*NSE* of 0.68 and R^2 of 0.74). Additionally, the model was able to predict monthly nitrate-N loads (*NSE* values between 0.68 and 0.71, and R^2 values of 0.7 and 0.73). These results revealed the predominant hydrology and the fate of nutrient in the Arawhata catchment.

- Mitigation scenarios (ryegrass cover crop is grown in fallow periods of the potato-onion rotations; brassica rotations were replaced by green vegetable crops rotations with small nitrate-N leaching rates; a combination of the previous two scenarios; a decrease of 10% in the quantity of N fertiliser applied to all crops, and a decrease of 20% in the quantity of N fertiliser applied to all crops) simulated in the coupled APSIM/Overseer and SWAT+ models suggested that reductions between 17 – 21% of mean annual nitrate-N load in Arawhata stream can be achieved. Adopting rotations with fewer brassicas crops was the single most effective practice among those evaluated in this study. The coupled model is an effective tool for investigating the effect of mitigation measures on N loads to the Arawhata stream.

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Appendix A: supporting material for Chapter 2

Table A1. APSIM studies in New Zealand.

#	Year	Study	Agricultural system	Species	Area	Leaching	Mitigation technique(s)
1	2023	Vogeler et al.		RG-WC	Canterbury	yes	n.a.
2	2022	Vogeler et al.		Pasture	New Zealand	n.a.	n.a.
3	2022	Watt et al.		Forage brassicas	Canterbury, Hawkes Bay	n.a.	n.a.
4	2020	Bryant et al.		Forage spcs.	Waikato and Canterbury	yes	Use of alternative forage species
5	2020	Cichota et al.		Forage chicory	Waikato and Canterbury	n.a.	n.a.
6	2019a	Vogeler et al.		RG-WC	Canterbury and Otage	yes	Irrigation reduction
7	2019b	Vogeler et al.		RG-WC	Otago	n.a.	n.a.
8	2019	Trolove et al		Forage rape-RG	Canterbury	yes	n.a.
9	2018	Vogeler and Cichota		RG-WC	Waikato, Canterbury and Otago	n.a.	n.a.
10	2018	Teixeira et al.		Lucerne-RG	Canterbury	n.a.	n.a.
11	2018a	Cichota et al.		RG	Waikato	yes	n.a.
12	2018b	Cichota et al.		RG-WC	Canterbury	yes	Fertilisation management
13	2018	Cichota and Snow		Chicory, plantain and WC	Waikato	n.a.	n.a.
14	2017	Romera et al.		n.a.	Waikato	yes	Restricted grazing+nitrification inhibitor
15	2017	Khaembah et al.		Fodder Beet	Canterbury	n.a.	n.a.
16	2017	Vogeler and Cichota		RG	Canterbury	n.a.	n.a.
17	2017	Vogeler et al.		Rape and RG	Canterbury	yes	n.a.
18	2017	van der Weerden		RG-WC	Waikato, Manawatu, Canterbury and Southland	yes	Controlled grazing
19	2017	Vogeler et al.		RG-WC-Paspalum	Canterbury	yes	Use of diversified pasture
20	2016a	Vogeler et al.		RG-WC	Hawkes Bay	n.a.	n.a.
21	2016b	Vogeler et al.		RG-WC	Waikato, Canterbury and Southland	n.a.	n.a.
22	2015	Vibart et al.		RG-WC	Manawatu	yes	n.a.
23	2015	Vogeler et al.		RG-WC	Waikato	yes	n.a.
24	2015	Moot et al.		Lucerne	Canterbury	n.a.	n.a.
25	2015	Vogeler and Cichota		RG	Canterbury	yes	Optimal fertilisation rate
26	2015	Moot et al.		Lucerne	Canterbury	n.a.	n.a.
27	2015	Giltrap et al.		RG-WC	Waikato and Otago	yes	n.a.
28	2014	Cichota et al.		RG-WC	Waikato, Canterbury and Southland	n.a.	n.a.
29	2014	Newton et al.	Pasture & forage crops	RG-WC	Canterbury	n.a.	n.a.
30	2014	Shepherd and Snow		n.a.	Waikato	yes	N fertilisation avoidance
31	2014	Vogeler et al.		RG-WC	Waikato	yes	Fertilisation management
32	2014	Li et al.		RG-WC	Manawatu	n.a.	n.a.
33	2013	Lucci et al.		RG-WC	Waikato	yes	n.a.
34	2013a	Vogeler et al.		RG-WC	Waikato and Canterbury	n.a.	n.a.
35	2013b	Vogeler et al.		RG-WC	Waikato	yes	Several
36	2013c	Vogeler et al.		n.a.	Waikato	yes	N leaching forecasting
37	2013	Cichota et al.		RG-WC	Waikato and Canterbury	yes	n.a.
38	2013	Vogeler et al.		n.a.	Waikato	n.a.	n.a.
39	2013	Snow et al		RG-WC	Canterbury	yes	Use of diversified pasture
40	2013	Snow and White		RG-WC	Northland, Waikato, Manawatu and Canterbury	yes	Use of diversified pasture
41	2013	Cichota et al.		RG-WC	Waikato, Manawatu, Canterbury and Otago	yes	Nitrification inhibitor
42	2013	Cichota et al.		RG-WC	Canterbury	n.a.	n.a.
43	2012	Romera et al.		RG	Waikato	yes	n.a.
44	2012	Cichota et al.		RG-WC	New Zealand	yes	n.a.
45	2012	Vibart et al.		RG-WC	Southland	yes	n.a.
46	2011a	Vogeler et al.		n.a.	Waikato and Canterbury	n.a.	n.a.
47	2011b	Vogeler et al.		RG-WC	Waikato	yes	Leaching forecasting
48	2011	Romera et al.		RG	Waikato	yes	N leaching forecasting
49	2011	Li et al.		RG-WC	New Zealand	n.a.	n.a.
50	2011	Cichota and Snow		n.a.	Canterbury	yes	n.a.
51	2011	Beukes et al.		n.a.	Canterbury	yes	Leaching forecasting
52	2010	Teixeira et al.		Lucerne	Canterbury	n.a.	n.a.
53	2010	Cichota and Snow		n.a.	New Zealand	yes	n.a.
54	2010	Cichota et al.		n.a.	Waikato	yes	Nitrification inhibitor
55	2010	Brown et al.		RG	Canterbury	n.a.	n.a.
56	2007	Snow et al		n.a.	Manawatu	n.a.	n.a.
57	2006	Brown et al.		Lucerne	Canterbury	n.a.	n.a.
58	2020	Teixeira et al.		Maize	New Zealand	n.a.	n.a.
59	2018	Khaembah and Horrocks		Wheat, oats, ryegrass	Canterbury	yes	n.a.
60	2017	Palmer et al.		Wheat	Canterbury	n.a.	n.a.
61	2017	Teixeira et al.		Maize	New Zealand	n.a.	n.a.
62	2016	Zyskowski et al.		Maize and forage rotation	Waikato	yes	Cover crops (triticale)
63	2016	Teixeira et al.	Cereals	Maize and wheat	Canterbury	yes	Cover crops (wheat)
64	2015a	Teixeira et al.		Wheat, kale and maize	Canterbury	n.a.	n.a.
65	2015b	Teixeira et al.		Maize	Canterbury	yes	Cover crops (wheat)
66	2013	Trolove et al.		Sorghum, wheat, lucerne	Several	n.a.	n.a.
67	2012	Brown et al.		Wheat	Canterbury	n.a.	n.a.
68	2010	Teixeira et al.		Several	Canterbury	n.a.	n.a.
69	2021	Ojeda et al.		Potatoes	New Zealand	n.a.	n.a.
70	2015	Khaembah et al.		Potatoes-fallow-peas-potatoes, potatoes-wheat-potatoes	Canterbury	yes	n.a.
71	2011a	Sharp et al.	Vegetable crops	Potatoes-italian ryegrass	Canterbury	yes	n.a.
72	2011b	Sharp et al.		Potatoes-peas	Canterbury	yes	n.a.
73	2011	Brown et al.		Potatoes-peas	Canterbury	n.a.	n.a.
74	2021	Zhu et al.	Fruit crops	Vitis vinifera	New Zealand	n.a.	n.a.

Appendix B: supporting material for Chapter 3

Table B1. Hills Laboratory soil summary of tests and methods

Sample Type: SOIL Potato		
Test	**Method Description**	**Default Detection Limit**
Individual Tests		
Sample Registration*	Samples were registered according to instructions received.	-
Soil Prep (Dry & Grind)*	Air dried at 35 - 40°C overnight (residual moisture typically 4%) and crushed to pass through a 2mm screen.	-
Soil Sample Depth*		-
Basic Soil		
pH	1:2 (v/v) soil:water slurry followed by potentiometric determination of pH. In-house.	0.1 pH Units
Olsen Phosphorus	Olsen extraction followed by Molybdenum Blue colorimetry. In-house method.	1 mg/L
Potassium (MAF)	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	1 MAF units
Calcium (MAF)	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	1 MAF units
Magnesium (MAF)	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	1 MAF units
Sodium (MAF)	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	2 MAF units
Potassium	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	0.01 me/100g
Calcium	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	0.5 me/100g
Magnesium	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	0.04 me/100g
Sodium	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	0.05 me/100g
Potassium (Sat)	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	0.1 %BS
Calcium (Sat)	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	1 %BS
Magnesium (Sat)	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	0.2 %BS
Sodium (Sat)	1M Neutral ammonium acetate extraction followed by ICP-OES. In-house.	0.1 %BS
CEC	Summation of extractable cations (K, Ca, Mg, Na) and extractable acidity. May be overestimated if soil contains high levels of soluble salts or carbonates. In-house.	2 me/100g
Total Base Saturation	Calculated from Extractable Cations and Cation Exchange Capacity.	5 %
Volume Weight	The weight/volume ratio of dried, ground soil. In-house.	0.01 g/mL
Mineral Nitrogen Profile Hamilton		
Ammonium-N*	Analysed on an 'as received' fraction but reported on a dry weight basis. 0.1M KCl extraction followed by Berthelot colorimetry.	1 mg/kg
Nitrate-N*	Analysed on an 'as received' fraction but reported on a dry weight basis. 0.1M KCl extraction followed by Cd reduction and NED colorimetry.	1 mg/kg
Mineral N (sum)*	Sum of Nitrate-N and Ammonium-N, calculated on a dry weight basis.	2 mg/kg
Dry Matter*	Weight loss on drying at 105°C for 24 hours.	0.5 %
Moisture*	Moisture is calculated from the Dry Matter.	0.5 %
Sample Temperature on Arrival*	Temperature of soil on arrival.	-10 °C
Sample Type: SOIL Potato		
Test	**Method Description**	**Default Detection Limit**
Organic Soil Profile		
Potentially Available Nitrogen*	Determined either by NIRS or by conventional wet chemistry method of 7-day Anaerobic incubation followed by extraction using 2M KCl followed by Berthelot colorimetry. (Calculation for kgN/ha based on 15cm depth sample). Note that any Mineral N present is included in the AN/AMN result reported.	10 kg/ha
Anaerobically Mineralisable N/Total N Ratio*		0.5 %
Organic Matter*	Organic Matter is 1.72 x Total Carbon.	0.2 %
C/N Ratio*		0.5

Appendix C: supporting material for Chapter 4

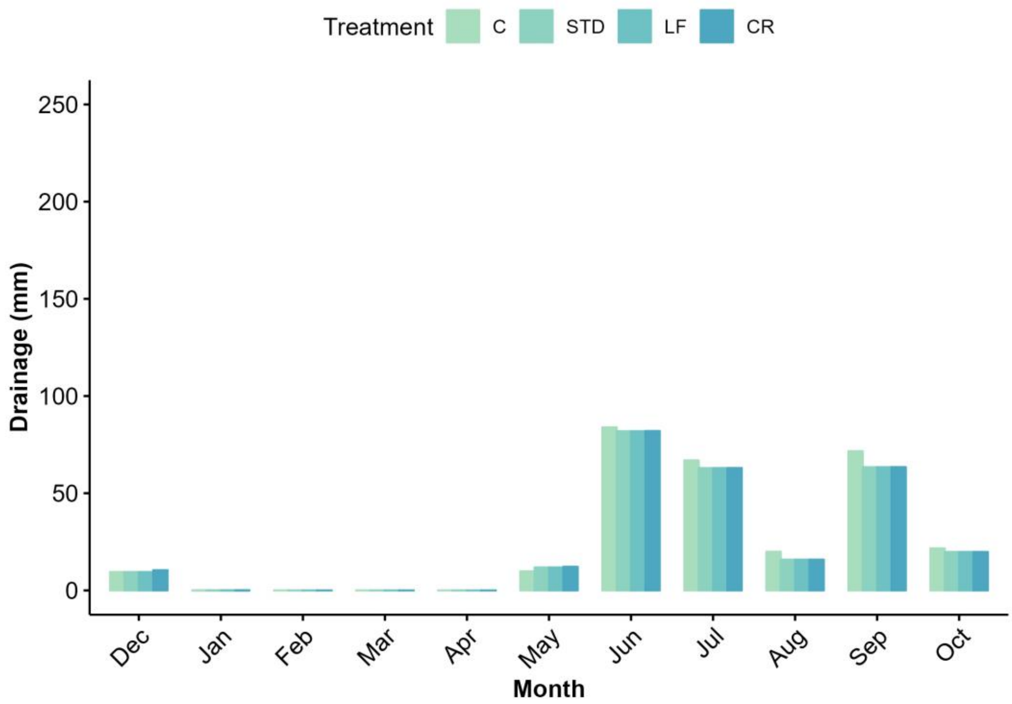

Figure C1. Simulated monthly drainage volumes at 60 cm for the first crop growing season from each treatment in the Potatoes site (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser).

Table C3. Cumulative simulated drainage volumes for the first potato growing season in the Potatoes site.

Treatment	**Cumulative drainage at 60 cm soil depth (mm)**
C	283.3
STD	265.1
LF	265.1
CR	266.6

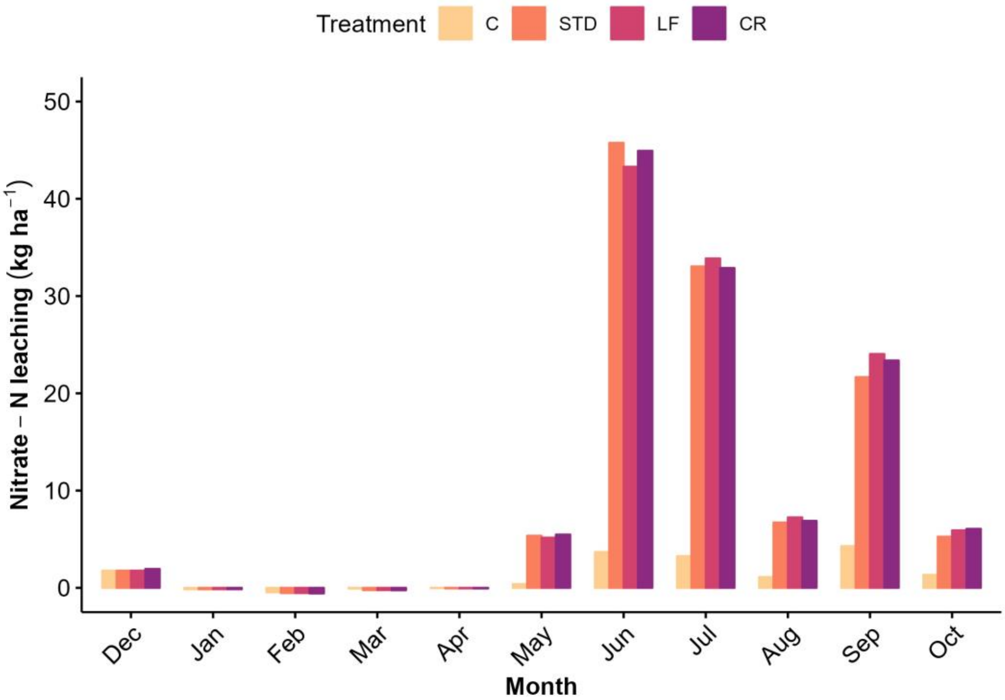

Figure C2. Simulated monthly nitrate-N leaching losses at 60 cm for the first crop growing season from each treatment in the Potatoes site (C for control; STD for standard practice; LF for split liquid fertiliser and CR for controlled release fertiliser).

Table C4. Cumulative simulated nitrate-N leaching losses for the first potato growing season in the Potatoes site.

Treatment	**Cumulative nitrate-N leaching at 60 cm soil depth (kg ha^{-1})**
C	15.2
STD	118.6
LF	120.3
CR	120.6

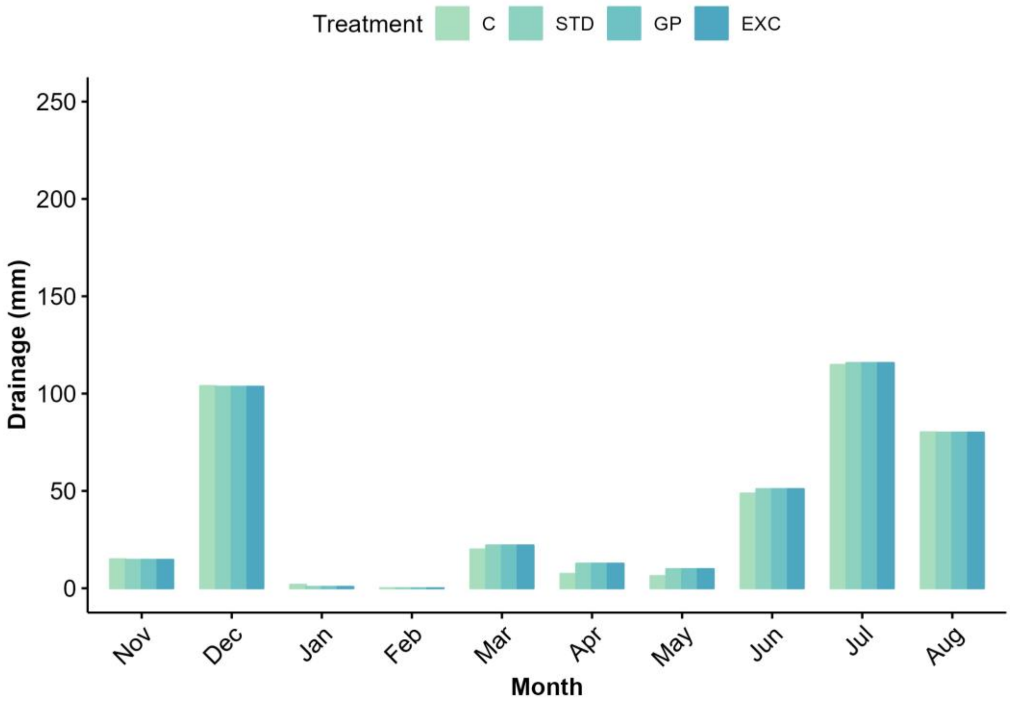

Figure C3. Simulated monthly drainage volumes at 60 cm for the second crop growing season from each treatment in the Potatoes site (C for control; STD for standard practice; GP for good practice and EXC for excess).

Table C5. Cumulative simulated drainage volumes for the second potato growing season in the Potatoes site.

Treatment	**Cumulative drainage at 60 cm soil depth (mm)**
C	397.3
STD	409.8
GP	409.8
EXC	409.8

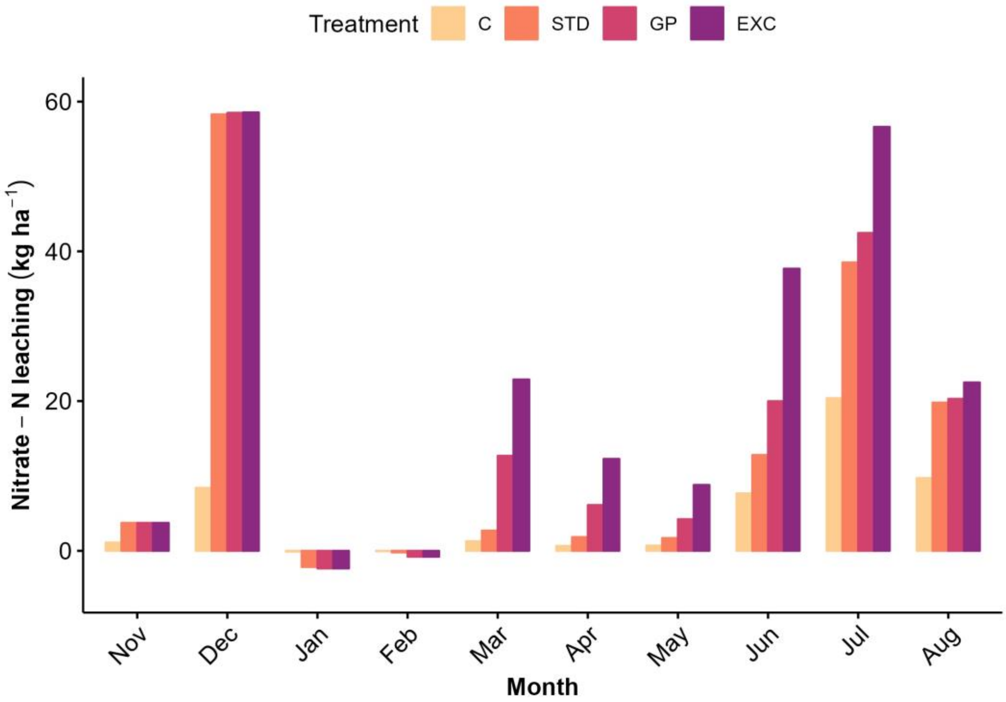

Figure C4. Simulated monthly nitrate-N leaching losses at 60 cm for the second crop growing season from each treatment in the Potatoes site (C for control; STD for standard practice; GP for good practice and EXC for excess).

Table C6. Cumulative simulated nitrate-N leaching losses for the second potato growing season.

Treatment	**Cumulative nitrate-N leaching at 60 cm soil depth (kg ha^{-1})**
C	49.8
STD	137.0
GP	164.9
EXC	219.9

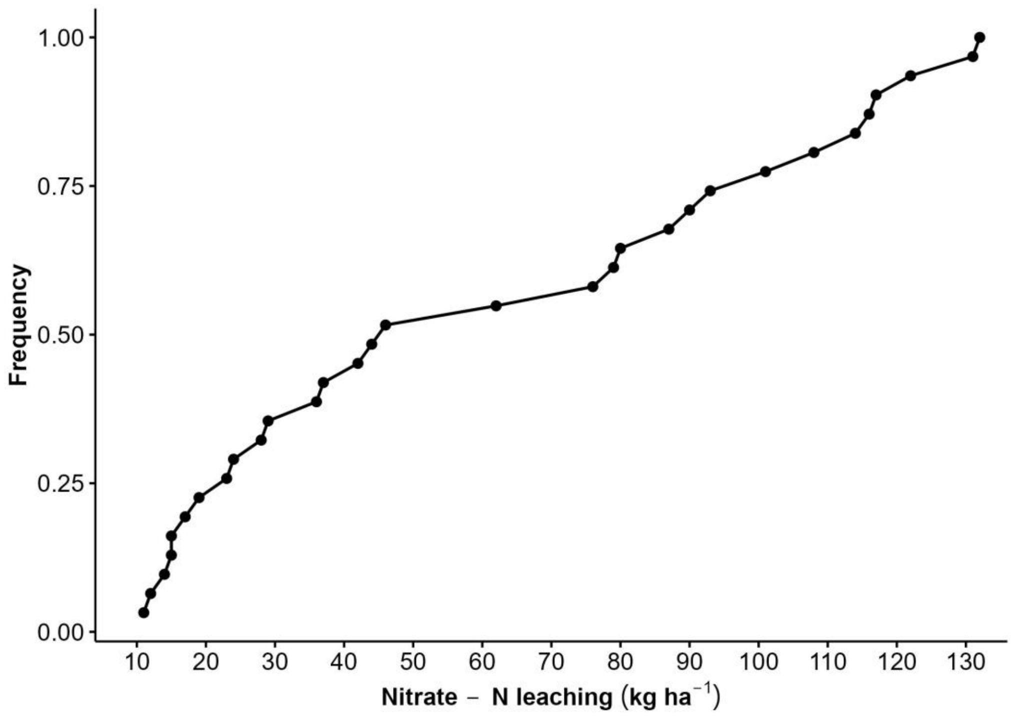

Figure C5. Frequency distribution of simulated nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 60 cm soil depth for years 1991 to 2021 in the Potatoes site.

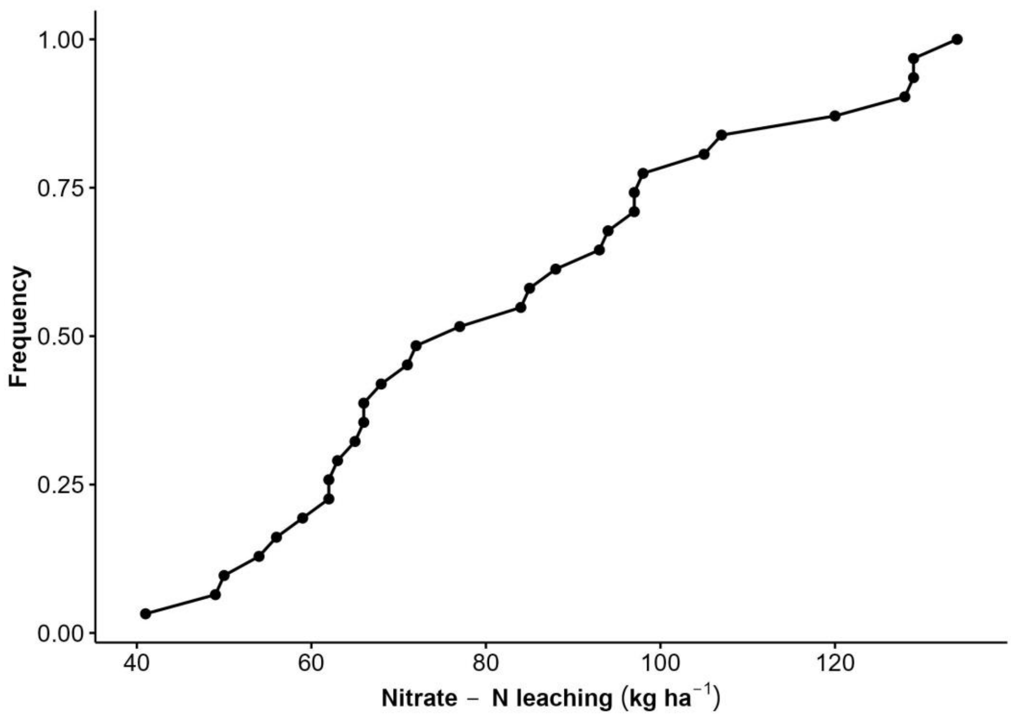

Figure C6. Frequency distribution of simulated nitrate-N leaching losses (kg nitrate-N ha^{-1}) at 60 cm soil depth for years 1991 to 2021 in the Green vegetables site.

Table C7. List of crops, expected yields, fertiliser rates and times used in simulated rotations in the Green vegetables site.

Crop	Expected yield	Fertiliser rates		YaraMila Complex	CAN
	t ha	Days from sowing	kg N ha^{-1}	kg ha^{-1}	kg ha^{-1}
Beetroot	130	0	36	300	
		30	81		300
Broccoli (summer)	8	0	20	100	30
		40	15		56
		50	15		56
Broccoli (rest of the year)	12	0	35	150	63
		40	20		74
		50	20		74
		65	20		74
Cauliflower (summer)	22	0	20	100	30
		40	38		141
		50	38		141
		65	38		141
Cauliflower (rest of the year)	30	0	50	210	92
		40	53		197
		50	53		197
		65	53		197
Cabbage (summer)	50	0	20	100	30
		40	53		197
		50	53		197
		65	53		197
Cabbage (rest of the year)	70	0	35	150	63
		40	37		136
		50	37		136
		65	37		136
Forage oats	8	0	0		
Lettuce (summer)	50	0	20	161	
		35	23		86
		45	23		86
		55	23		86
Lettuce (rest of the year)	50	0	9	71	
		40	9		32
		50	9		32
		65	9		32
Maize	18	0	35	292	
Pak Choi	10	0	50		180
		50	45		167
Squash	40	0	90	250	222

		15	90		333
Spinach (summer)	20	0	20	100	30
		40	23		83
		50	23		83
Spinach (rest of the year)	20	0	35	150	63
		40	15		56
		50	15		56
Spring Onion	30	0	20	100	30
		35	37		136
		45	37		136
		55	37		136