Challenges and Opportunities for Sustainable Nitrogen Management in Dairy Systems

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored. See Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contributions

Chapter 2 of this thesis consists of a published paper (Lakhanpal et al., 2025) co-authored by Dr. Owen Fenton, Dr. Donal O'Brien, Dr. Nandita Basu, Dr. Brendan Horan, Dr. Karl Richards, and Dr. Cathal Buckley. I conducted the literature review that identified the knowledge gap and subsequently designed the experimental framework, which involved developing and simulating a range of nitrogen mitigation scenarios to evaluate strategies for more efficient farm-scale N management. I led the analysis and summary of results, wrote the first draft, and contributed extensively to manuscript editing and revisions. The study was primarily led by me, with support from Dr. Donal O'Brien on the simulation scenarios for the model. All co-authors contributed through discussion of results and revision of the manuscript.

Lakhanpal, G., Basu, N.B., O'Brien, D., Buckley, C., Horan, B., Richards, K.G. and Fenton, O., (2025). Simulation of management practices to reduce nitrogen losses to water and air on well-drained grass-based dairy farms in derogation. Journal of Dairy Science. https://doi.org/10.3168/jds.2024-25853

Chapter 3 of this thesis consists of a submitted paper (Lakhanpal et al., in preparation) co-authored by Dr. Owen Fenton, Dr. Nandita Basu, Dr. Brendan Horan, Dr. Karl Richards, Dr. Luis Lopez-Sangil, Gabriela Cardenas-Alvarez, and Tomas Condon. Following an extensive review of past studies, I developed the experimental field and laboratory design, along with the data analysis approach, to address the identified knowledge gap and quantify the legacy N pool in well-drained soils. I led the data collection, analysis, and interpretation of results, wrote the first draft, and contributed extensively to manuscript revisions. Soil and groundwater data collection was supported by Gabriela Cardenas-Alvarez, Dr. Lopez-Sangil, and Tomas Condon. Drs. Fenton, Basu, Horan, and Richards provided input on design, analyses and interpretation. All co-authors contributed to editing and revising the manuscript.

Lakhanpal, G., Fenton, O., Basu, N.B., Horan, B., Richards, K.G., Cardenas-Alvarez, G., Lopez-Sangil, L. and Condon, T., (In Review) Multi-Decadal Assessment of Relationships between Intensive Dairy Farm Nitrogen Surplus, Soil Organic Nitrogen Legacy Stores in Soil and Their Impact on Groundwater Nitrate Concentrations in South of Ireland.

Chapter 4 of this thesis will be prepared as a manuscript. I was primarily responsible for study design, data synthesis, data collection, and drafting of the manuscript. Drs. Fenton, Basu and Horan, contributed to study design and analysis. Dr. Lopez-Sangil and Ms. Gabriela Cardenas-

Alvarez assisted with technical expertise, field equipment, data collection and processing. Ms. Orla Mattimoe collated paddock-level clover content data for the Johnstown Castle site, and John Murphy and Tomas Condon contributed multi-decadal groundwater quality data for both sites. I wrote the first draft, and all co-authors contributed to manuscript revisions.

Abstract

Nitrogen (N) is central to agricultural productivity, yet its mismanagement drives water and air pollution across the world. Ireland's grass-based dairy systems are among the most N-intensive in the European Union (EU), with high inorganic and organic fertilizer sustaining productivity but creating persistent surpluses that threaten groundwater and surface water quality. Despite major policy efforts, Ireland continues to struggle to meet EU Water Framework Directive (WFD) chemical targets for good water status. However, Ireland is still seeking the renewal of its Nitrates Derogation, which allows exceptionally high stocking rates up to 220 kg N ha^{-1} yr^{-1}. This tension between economic needs and environmental compliance defines one of the country's greatest agri-environmental challenges. The EU is moving toward tighter nutrient limits and nature restoration objectives, making it essential to understand whether sustainable dairy production can coexist with future regulatory expectations.

One of the main obstacles to achieving water-quality goals is the temporal disconnect between management interventions and measurable improvements, which can erode stakeholder confidence and obscure the true impact of mitigation policies. In Ireland, the EU WFD program of measures (PoMs) is carried out under the Nitrates Directive, which include nutrient management, land management and farmyard management strategies to protect water quality. These lags are increasingly attributed to legacy N i.e., reactive N accumulated in soils and subsoils from past surpluses that continue to leach long after inputs decline. While groundwater legacy effects are recognized (i.e., the time it takes water to travel through the soil termed hydrologic time lag), few studies worldwide have directly quantified soil legacy N (i.e., biogeochemical time lags), and none had done so in Ireland prior to this research. Understanding the scale, distribution, and persistence of these soil pools is critical for designing realistic mitigation timelines and adaptive policies.

The overarching aim of my research was therefore to assess N dynamics and environmental outcomes in Irish dairy systems by evaluating mitigation scenarios and quantifying legacy soil N accumulation to understand how current and historical management, drainage class, and hydrogeological setting influence both near-term losses and the pace of environmental recovery. I combined process-based modelling, multi-decadal farm data, deep soil coring, and groundwater monitoring to connect farm management decisions with both

short- and long-term system responses. Together, these studies form the first integrated assessment of soil N legacies in Irish dairy systems.

In Chapter 2, I used the €riN-MDSM model to simulate N flows, surpluses, and losses in a well-drained dairy farm operating under derogation conditions. This model, developed to represent N cycling in Irish grass-based systems, quantifies losses of nitrate (NO_3^-), ammonia (NH_3), nitrous oxide (N_2O), and dinitrogen (N_2) from urine, dung, slurry, dairy soiled water, and fertilizer. I simulated a range of management scenarios, including reduced inorganic N rates (200–225 kg N ha^{-1}) and organic rates (170–430 kg N ha^{-1}), substitution of calcium ammonium nitrate (CAN) with protected urea, and restrictive grazing during vulnerable winter–spring months. Results showed that integrated approaches combining restrictive grazing, protected urea, and reduced fertilizer inputs lowered NO_3^- leaching by up to 44 % and NH_3 volatilization by 31 %, bringing water losses close to the 30 kg N ha^{-1} threshold for good groundwater quality. These findings demonstrated that substantial environmental gains are possible through system-level optimization, but that even under improved management, N surpluses remain high, implying persistent risks to water and air quality. This modelling work provided a critical benchmark for assessing what levels of mitigation might be achievable within the derogation framework and highlighted the need to understand how historical surpluses continue to affect recovery, setting the stage for the legacy N analyses that followed in the next phase of this research study.

In Chapter 3, I conducted a 24-year investigation (2001–2024) at Moorepark Teagasc Research Farm (known as Curtins Research Farm locally) in southern Ireland, a well-drained, karstic site with low denitrification potential. I reconstructed multi-decadal N budgets from detailed farm records and collected 75 soil cores across 15 paddocks, 1m deep profiles representing a gradient of historical management intensity. Annual N surpluses frequently exceeded 200 kg N ha^{-1} yr^{-1}, leading to cumulative soil N accumulation of 4,000–5,500 kg N ha^{-1} in the top 50 cm. Groundwater NO_3^- loads declined from over 70 kg N ha^{-1} in the early 2000s to under 30 kg N ha^{-1} by 2024, yet concentrations have plateaued rather than continuing to fall. This persistent signal alludes that subsoil N stored from past decades continues to mineralize and leach, sustaining groundwater nitrate levels despite reduced inputs. These findings provide the first direct quantification of legacy soil N in Irish dairy systems, showing that deep soil stores act as long-term sources of reactive N, constraining the pace of water quality recovery even when surface management improves.

In Chapter 4, I expanded the investigation to include the Johnstown Castle Teagasc Dairy Research Farm, a variably drained site in the southeast with finer-textured soils and higher denitrification potential. I analyzed 45 soil cores from 9 paddocks, 1 m deep profiles covering an 18-year management period and compared results to those from Curtins. Despite lower annual surpluses (~100–150 kg N ha^{-1} yr^{-1}), Johnstown Castle soils contained 4,000–11,000 kg N ha^{-1} in the upper 50 cm, substantially higher than the well-drained Curtins profiles. The difference reflected higher clay and silt content, which enhanced N retention through adsorption and organic-mineral associations, as well as shallower water tables and moderate denitrification that reduced nitrate transport to groundwater but trapped nitrogen in the soil profile. These results revealed a clear trade-off: well-drained systems potentially act as "fast transmitters," showing rapid leaching but quicker recovery when management improves, whereas variably drained systems are possibly "slow retainers," buffering groundwater in the short term but accumulating persistent legacy N stores that prolong recovery.

By linking long-term management records, soil data, and investigating groundwater trends across these contrasting systems, I demonstrated that N accumulation is governed by the interaction of soil texture, soil drainage class, hydrology, denitrification potential, and historical management intensity. Across both sites, total soil N accumulation exceeded 3,000–11,000 kg N ha^{-1}, far higher than values reported for most temperate cropland systems, confirming the exceptional capacity of Irish grassland soils to store reactive N from decades of intensive management.

This thesis makes several novel contributions. It provides the first empirical evidence of soil legacy N magnitudes in temperate dairy grasslands, quantifies their long-term influence on water quality and nitrate dynamics, and develops a conceptual framework concerning drainage, soil texture, and hydrology to N retention and release. It also demonstrates how soil legacy N can be reframed as both a risk and a resource—a potential nutrient reservoir that, if managed strategically, could offset fertilizer needs during the transition to lower-input systems.

These findings have direct implications for Ireland's compliance with the EU Nitrates and WFD. Current six-year reporting cycles are too short to capture recovery in legacy-affected catchments, creating the perception of policy failure. Integrating soil monitoring to 1 m depth alongside existing high-resolution catchment and groundwater networks would enable more accurate assessment of progress and support realistic, site-specific mitigation timelines. Legacy N must be explicitly incorporated into nutrient models, regulatory assessments, and PoMs to

ensure that both soil and water systems are managed as coupled components of the nitrogen cycle.

Ultimately, this research underscores that Ireland's path to sustainable dairy production requires addressing both current N surpluses and historical legacies. The methods and evidence developed here — combining modelling, deep soil sampling, and long-term monitoring offer a blueprint for future national assessments and international comparisons. As EU policy evolves toward stricter nutrient limits and nature restoration goals, understanding and managing legacy N will be fundamental to aligning agricultural productivity with environmental resilience.

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Dedication

This PhD is dedicated to the women who raised me: Raj Lakhanpal, Manju Verma, Rashmi Lakhanpal, and Gayatri Lakhanpal, and to the past, present, and future.

The work within these pages reflects on the actions of the past that shape our present and the choices of the present that shape our future. How can we, as human beings, strive for balance?

It is also dedicated to my ancestors, the farmers I knew and those I did not, whose hands once worked the same earth I now study. The impact of their actions flows through generations, through me, and onward into a future I may never see.

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Chapter 1

Introduction

"We call this sequence of effects the nitrogen cascade. … The same atom of reactive nitrogen can cause multiple effects in the atmosphere, in terrestrial ecosystems, in freshwater and marine systems, and on human health."
(James N. Galloway, 2003)

Human activities now generate substantially more reactive nitrogen (Nr) than natural terrestrial processes, disrupting the global nitrogen cycle and driving a cascade of environmental impacts across air, land, and water systems (Galloway et al., 2003; Sutton et al., 2011). This cascade demonstrates how a single atom of Nr can sequentially contribute to eutrophication, groundwater contamination, soil acidification, and greenhouse gas emissions. Since most newly created Nr is produced to support food production, agriculture is central to both the problem and the potential solutions. Recent research has highlighted the importance of quantifying legacy N and understanding the factors that regulate its mobilization, including the role of soil drainage class in shaping groundwater nitrate dynamics (Ascott et al., 2021; Fenton et al., 2009; Forrestal et al., 2025; Jahangir et al., 2012; Van Meter et al., 2016). This chapter sets the global context of N pollution, outlines the major N pathways in agricultural landscapes, and examines how dairy systems contribute to and can mitigate N losses. It then narrows to Ireland's pasture-based sector and policy context before identifying key research gaps that motivate this thesis: (i) limited representation of fine-scale farm management in N models, (ii) poor integration of legacy N accumulation and temporal lags, and (iii) lack of Ireland-specific frameworks linking long-term management, soils, and groundwater. The chapter concludes by presenting the objectives and hypotheses that guide the remainder of the thesis.

1.1 Global nitrogen pollution

Over the past century, human activities have led to a striking 14-fold increase in the levels of Nr in the environment (Carpenter et al., 1998). The surge is primarily due to anthropogenic activities, including industrialization, intensive agriculture relying heavily on mineral fertilizers, and fossil fuel combustion (Brown et al., 2005; Chen et al., 2018; de Klein

et al., 2017). Increased use of Nr has contributed to significant environmental pollution, including coastal eutrophication, groundwater contamination, soil acidification and greenhouse gas (GHG) emissions (Arsenault et al., 2009; O'Brien et al., 2012).

An increase in N pollution threatens both freshwater and coastal ecosystems (Erisman et al., 2018; Fowler et al., 2013; Galloway & Cowling, 2021). Elevated N concentrations promote algal growth, leading to accelerated eutrophication and the formation of hypoxic conditions, characterized by dissolved oxygen (DO) levels below 2 mg L^{-1} (Diaz et al., 2017; Harding et al., 2019; Van Meter et al., 2018). Both eutrophication and hypoxic zones have been recorded globally, from the freshwaters of Lake Erie and Lake Okeechobee to saline systems including the Baltic Sea and the Gulf of Mexico (Conley et al., 2011; Dubrovsky & Hamilton, 2008; Rabalais & Turner, 2019; Scavia et al., 2017).

The quality of groundwater is deteriorating worldwide, with contamination levels often surpassing acceptable thresholds. In the United States of America (USA), approximately 4.4 % of domestic wells and 2 % of public water wells have elevated N concentration primarily attributed to agricultural sources (Ransom et al., 2022). Over half of the aquifers in the USA have at least one groundwater well sampling point that exceeds the Maximum Admissible Concentration (MAC) limit of 10 mg nitrate-N (NO_3-N) L-1 (Environmental Protection Agency (EPA), 2002; Pennino et al., 2017). In the European Union (EU) 27 member states, nitrate (NO_3^-) is recognized as one of the major groundwater pollutants, impacting 18 % of the groundwater (European Environment Agency (EEA), 2018). Additionally, in countries like Denmark and China, N concentrations in over 40 % and 28 % of wells, respectively, exceed the World Health Organization (WHO) standards of 10 mg L-1 (Hansen et al., 2019). Notably, chronic human exposure to N, even at concentrations well below WHO limits, poses severe health risks, and increases the risk of cancer (Ward et al., 2018).

1.2 Nitrogen cycling within an agricultural context

The nitrogen (N) cycle has been considerably influenced by the agriculture industry. Agricultural landscapes contribute to the majority of N pollution, by supplying N to aqueous systems through surface and subsurface runoff (e.g., leaching to groundwater, interception by tile drainage with direct transfer to surface waters). Globally over half of the N used for crop fertilization ends up lost to the environment through multiple pathways (Lassaletta et al., 2014). Only 47 % of the applied N is successfully converted to harvested products. It is noteworthy

that global use of N fertilizers has escalated, increasing nine-fold from 1960 to 2012 (Lassaletta et al., 2014; Swaney et al., 2018).

Agricultural landscapes receive N from various sources, including synthetic fertilizers (inorganic), manure (organic), biological N fixation, and atmospheric deposition. Mineralization processes transform organic matter from manure and soil organic matter into ammonium (NH_4^+). This NH_4^+ is then oxidized to form nitrite (NO_2^-) and subsequently NO_3^- through nitrification. Conversely, denitrification processes convert NO_2^- and NO_3^- to di-nitrogen gas (N_2) and nitrous oxide (N_2O) emissions.

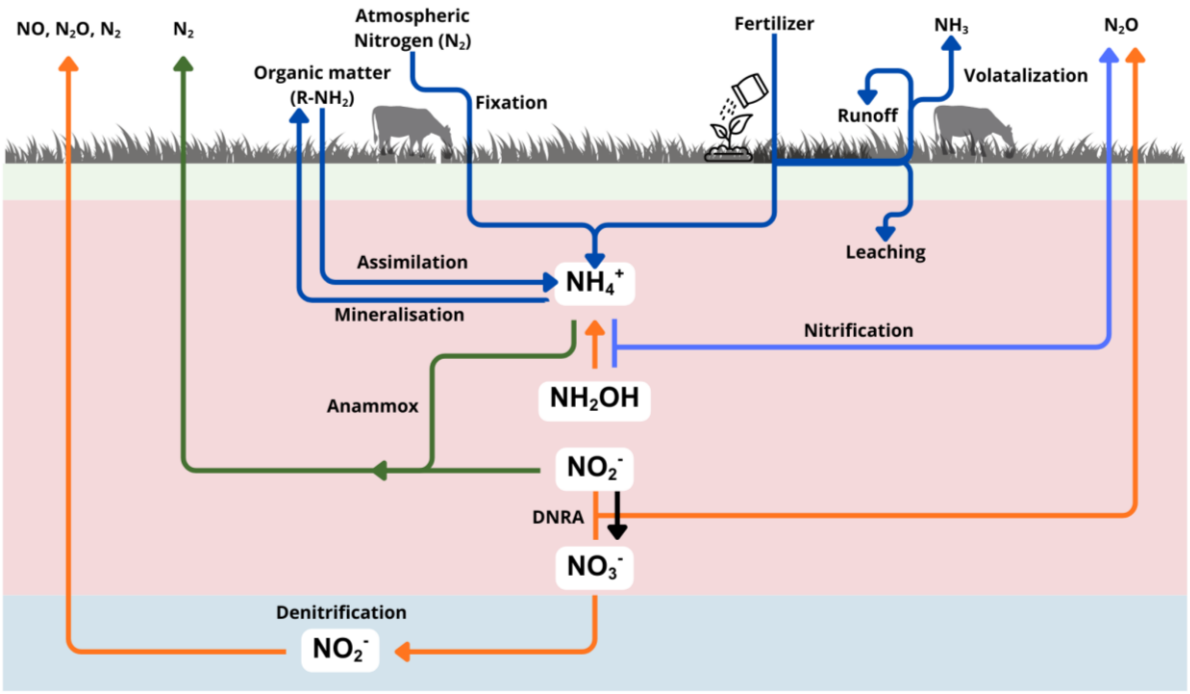


Figure 1- 1 Schematic of Nitrogen cycling in dairy systems (Adapted from Clagnan et al. (2017)).

Nitrogen is a vital nutrient for all life forms, yet it is often inaccessible in its dominant atmospheric form (N_2), which is biologically inert. Ecosystem functioning relies instead on a range of Nr compounds, such as NH_4^+, NO_3^-, NO_2^-, N_2O, NO, and various organic N forms (Galloway et al., 2003; Erisman et al., 2011). These Nr species participate in a complex biogeochemical cycle driven by oxidation–reduction processes that govern their transformation, mobility, and fate in the environment. The N cycle is strongly regulated by

hydrological connectivity, particularly along soil–water interfaces, which influence the transport and transformation of N compounds across ecosystems (Cabello et al., 2004).

Among the reactive N forms (Figure 1-1), NO_3^- is the most prevalent contaminant in soils and groundwater due to its high solubility and mobility. Additional agriculturally derived N pollutants include NH_4^+, NO_2^-, and N_2O, the latter of which is also a potent greenhouse gas. These compounds frequently arise as intermediates or endpoints of key microbial processes such as nitrification and denitrification. Nitrification involves the aerobic oxidation of NH_4^+ to NO_3^-, while denitrification refers to the stepwise anaerobic reduction of NO_3^- to N_2 gas (Rivett et al., 2008). However, incomplete denitrification can result in the accumulation of NO_2^-, NO, and N_2O—compounds with significant environmental impacts.

Mitigating the environmental burden of Nr requires either reducing its generation or enhancing its removal, particularly by promoting complete reduction to N_2 through denitrification or alternative microbial pathways (Galloway et al., 2003). Other significant N transformation pathways include dissimilatory NO_3^- reduction to NH_4^+ (i.e., Dissimilatory nitrate reduction to ammonium -DNRA), anaerobic ammonium oxidation (anammox), and co-denitrification, each contributing to N cycling under specific environmental conditions.

The microbial reduction of NO_3^- to NH_4^+, known as DNRA is an anaerobic process that can occur under environmental conditions like those supporting denitrification (Kelso et al., 1997). Soil physicochemical properties, particularly permeability and associated hydrological conditions, play a critical role in regulating the prevalence of DNRA (Dzakpasu et al., 2014). The process is favored under strongly reducing conditions and in environments with elevated concentrations of dissolved organic carbon (DOC).

A key factor governing the relative dominance of DNRA versus denitrification is the ratio of available carbon to nitrate. High C:NO_3^- ratios—typically exceeding 12—combined with low redox potential promote DNRA, whereas lower C:NO_3^- ratios favor denitrification pathways (Yin et al., 1998; Rütting et al., 2011). In addition, DNRA is carried out by obligate anaerobic microorganisms, which occupy a narrower ecological niche than denitrifying bacteria, thereby limiting the environments in which DNRA can occur (Buss et al., 2005).

The rates and pathways of these transformations are strongly influenced by environmental variables, including soil physicochemical properties, substrate concentrations, vegetation cover, land management practices, and climatic factors (Saggar et al., 2013).

Understanding these regulatory controls is essential for predicting nitrogen dynamics and designing strategies to reduce nitrogen losses to the environment.

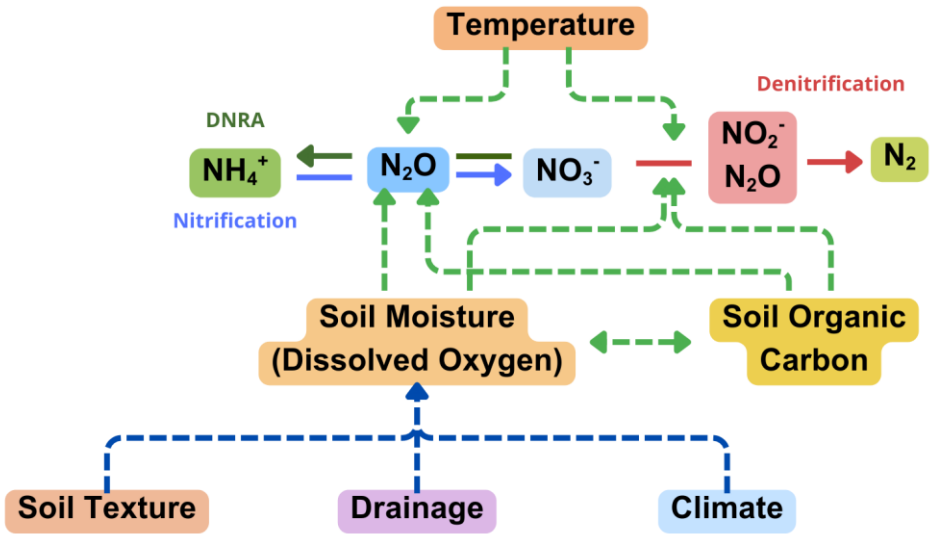


Figure 1- 2 Key soil and hydrological parameters influencing nitrogen (N) attenuation and speciation in the vadose zone and shallow groundwater. The figure highlights nitrogen species susceptible to transformation and potential loss, which may contribute to groundwater quality deterioration. (Adapted from Clagnan et al. (2017))

Figure 1-2 (adapted from Clagnan et al. (2017)) presents the key soil and hydrological controls on N attenuation and speciation, with a particular focus on the influence of soil texture and drainage class. These variables are especially relevant across the two study sites, which differ in their hydrogeological characteristics—one being uniformly well-drained, and the other exhibiting a mosaic of drainage conditions, ranging from well- to poorly-drained zones.

At the variably drained dairy farm, such heterogeneity results in spatially distinct N transformation processes. In well-drained areas, oxidising conditions support the conversion of NH_4^+ to NO_3^- via nitrification, with NO_3^- readily leaching to groundwater. Conversely, in poorly drained areas, where saturated, anaerobic conditions prevail, NO_3^- is more likely to undergo DNRA, resulting in the accumulation of NH_4^+. These reduced N species are then transported laterally via artificial tile drainage systems installed within the top 1 m of the soil, ultimately discharging into adjacent surface water bodies.

This spatial variability in drainage and redox conditions underscores the importance of landscape structure in regulating N loss pathways and the potential for both groundwater contamination and surface water enrichment across intensively managed agricultural systems.

1.3 Nitrogen management in dairy systems

In agricultural systems, intensive dairy farming stands out as a major contributor to N pollution. A notable reason for the intensified pollution is the industry's transition from localized, small-scale crop-livestock systems to global, demand-driven supply chains (Herrero et al., 2013; Steinfeld et al., 2006; Tilman & Clark, 2014). This results in a disconnection, where livestock feed is produced continents away from where the animals are raised; for instance, South American soybeans feed cows in the EU (Freeman & Thornton, 2006; Haan et al., 2010). Consequently, these changes have profoundly altered N biogeochemical cycles at the global scale.

In the EU context, livestock supply chains account for a large portion of pollution: 82 % of the total agricultural ammonia (NH_3) emissions and 73 % of all agricultural emissions to water bodies (Leip et al., 2015). Notably, the National Rivers Authority (NRA) of England and Wales highlighted that 55 % of all agricultural pollution incidents were attributed to dairy farming (NRA, 1997). Leading dairy exporters include the EU, New Zealand and India (Food and Agriculture Authority, (FAO), 2019; Uwizeye et al., 2020). Countries with intensive dairy farming, such as Republic of Ireland (henceforth mentioned as Ireland) and New Zealand, predominantly export the majority of the milk produced within the country as dairy products and milk powders for infant formula to global markets such as Africa and Asia (More, 2009). For perspective, this relatively small island nation of Ireland contributes only 0.94 % to global milk production, where it exports 85 % of its annual output and holds 10 % of the global infant formula market (Bord Bia, Irish Food Board, 2015; International Dairy Federation (IDF), 1988) The international dairy trade has been dominated by the EU (30 %) and Australasia, with New Zealand contributing 32 % and Australia 12 % (Dairy Australia, 2008).

The heavy emphasis on dairy product export means that while products reach global consumers, the environmental issues tied to biodiversity loss, water pollution, and air quality primarily impact local ecosystems, often undermining ecosystem services like drinking water provision from surface and groundwater (Galloway et al., 2003; Haan et al., 2010). The vast majority of dairy products may be exported beyond regional boundaries, but the environmental consequences linger in the production locales (Lassaletta et al., 2014; Oita et al., 2016). To

mitigate these environmental challenges, a combination of precision agriculture, an in-depth grasp of N transfers across diverse agricultural landscapes, and insights into various climatic influences are crucial (Sutton et al., 2013, 2011; Uwizeye et al., 2020).

N losses from intensive dairy systems are comparable across different parts of the world and come from both fertilizers and direct animal contributions like urine and dung. N losses from grazed pastures vary from 22-79 kg N ha^{-1} yr^{-1}, whereas N losses from grazed forage vary from 27-153 kg N ha^{-1} yr^{-1} (McDowell et al., 2017). These figures are in line with losses from semi-confined animal feeding operations in the USA and Europe (McDowell et al., 2016). Intensive dairy systems are particularly of interest, due to N losses from both organic and inorganic fertilizers (Del Prado et al., 2011; Gourley et al., 2012; Kohn et al., 1997) and animal inputs such as urine and dung deposition in the fields (Baily et al., 2011; Fenton et al., 2009; Huebsch et al., 2013; Misselbrook et al., 2005).

A typical dairy farm is divided into multiple farm units that include grassland and farm building structures that include animal housing, milking parlor, manure and silage storage and management machinery. Efficient N management is pivotal for reducing the environmental impact of dairy farms and enhancing their sustainability. Factors like fertilizer application for pasture, animal housing and grazing, manure management and storage influence how efficiently N is used in dairy systems (Powell et al., 2010). Nitrogen use efficiency (NUE) gauges this effectiveness, representing the cow's capability to convert feed N to milk and the ability of crops to utilize fertilizers for growth (Powell et al., 2010). NUE operates on the mass balance concept, which involves using data on N input and output for its computation (NUE = N output / N input). It's essential to understand NUE values in context with N surpluses (N input - N output) and associated productivity levels (N output, which is milk, slurry and/or harvested forage) (EU Nitrogen Expert Panel, 2015). Studies indicate an enhanced NUE, marked by a 13 % drop in fertilizer usage from 1990-2005 and a reduced N surplus—the gap between N inputs and outputs at the farm-gate level (Bouraoui & Grizzetti, 2011; Byrnes et al., 2020). Globally, only 5-15 % of fertilizer N ends up in our food, and very little animal excreted N gets recycled (Erisman et al., 2018; Janssen & Oenema, 2008). Average pasture-based livestock production systems have very low NUE (<30 %) resulting in extremely large surpluses that are susceptible to losses (Burchill et al., 2016). In Europe, high-intensity dairy systems tend to have the lowest NUE (< 30 %), resulting from N losses through volatilization and along runoff (Buckley et al., 2016; EPA, 2024; Mihailescu et al., 2014). Enhancing NUE can concurrently reduce environmental burdens and elevate farm productivity. Groot et al.

(2006) show the potential need for more effective N management of N imports from fertilizers and concentrate feeds, which can contribute to decreased N overall imports and increased rates of NUE.

Balancing the environmental and economic benefits of intensive dairy farming is complex (van Grinsven Hans & Grizzetti Bruna, 2011). Apart from N losses into the environment, the pressure to meet the high production or milk outputs from farms alongside the environmental concerns that follow is a difficult equilibrium (Alexandratos, 2011). Economically, managing N through chemical fertilizers, N inhibitors, and balancing feed imports and livestock, adds layers of complexity. In response, international policies aim to limit environmental N losses while supporting agricultural advancement. However, adhering to these policies, especially in intensive agriculture, poses on-going challenges (Sutton et al., 2013). In Europe, over application of nutrients in intensive dairy farming continues to ignite environmental concerns in soils, groundwater and surface water bodies (Volk et al., 2009). Specifically in North Atlantic Europe, which is characterized by a temperate climate, farmers can grow large quantities of grass from March to September (Creamer & O'Sullivan, 2018; Schulte et al., 2018). This extended growing season supports high stocking rates and frequent fertilizer applications, resulting in sustained N inputs and increased pressure on surrounding ecosystems.

1.4 Dairy systems, water quality and water quality policy in Ireland

Since 2007, the Irish dairy industry has undergone substantial expansion, largely driven by the abolition of EU milk quotas (from 2015) that had previously capped production. Between 2007 and 2019, total milk output rose by 68 %, from 4.93 to 8.29 billion liters, with approximately one-third of production destined for export (Dillon et al., 2019; National Milk Agency (NMA), 2019). Despite producing around 5 % of the EU's total milk supply, Ireland consumes only 6 % of its production domestically (National Milk Agency (NMA), 2017). As of 2021, Ireland ranks as the most competitive EU member in terms of both total and cash production costs per kilogram of milk solids (Dillon et al., 2021). The sector's productivity growth—up 24 % between 2010 and 2018—has been underpinned by improvements in grass utilization, spring calving, and herd genetics.

However, this rapid expansion has intensified environmental pressures. The EPA-Ireland and others have identified the removal of milk quotas as a key driver of water-quality decline, particularly in the south and southeast, where dairy density is greatest (Binfield et al.,

2008; Čechura et al., 2021; Läpple et al., 2022). Across northwestern Europe, surface waters in intensive agricultural regions—Belgium, northern Germany, and the Netherlands—have the poorest ecological status, with over 90 % rated "less than good." While Ireland's waters compare favorably at the European scale, they still fall short of EU Water Framework Directive (WFD) targets (Whytock, 2020). The most recent EPA Water Quality Report (2016–2021) shows that nearly half (46 %) of surface waters remain in unsatisfactory condition, with declines most pronounced in estuaries (–15.7 %) and coastal waters (–9.5 %) along the southern and south-eastern coasts, primarily due to agricultural runoff and leaching.

Ireland's mild climate and long growing season make grass-based dairy systems the most productive in Europe (Dillon et al., 1995; Humphreys et al., 2008). Optimal conditions occur in the south and southeast, where soils are moderately to well drained and rainfall ranges from 800–1,000 mm yr^{-1}. In contrast, farms on heavier, poorly drained soils in high-rainfall zones (> 1,200–2,500 mm yr^{-1}), such as the west and border regions, produce only ~30 % of national milk output and require careful management to prevent soil compaction and runoff (Tuohy et al., 2021). Artificial drainage networks, while improving trafficability, also increase hydrological connectivity and pathways for nutrient loss.

At the European level, water quality protection is governed by the WFD, 2000/60/EC, which mandates that Member States achieve good chemical and ecological status in all waters within defined reporting periods. To meet these objectives, Ireland implements a suite of Programs of Measures (PoMs), the most significant being the Nitrates Directive (91/676/EEC). This Directive underpins Ireland's agricultural water-quality policy by setting limits on N and phosphorus applications and prescribing good agricultural practices. It is reviewed and revised every three years through Ireland's National Action Program (NAP), under which farm advisors support farmers in adopting mitigation measures. The Department of Agriculture, Food and the Marine (DAFM) also conducts regular on-farm inspections to assess compliance with the Directive. In parallel, scientific research—especially outputs from the DAFM-funded Agricultural Catchments Program (ACP) and core and externally funded research within Teagasc—feeds directly into NAP reviews and PoM revisions, ensuring that policy is informed by the latest evidence on nutrient dynamics and water-quality trends.

Despite these robust frameworks, Ireland continues to struggle to meet EU WFD objectives, particularly in regions dominated by intensive dairy production (i.e., S and SE of the country). A major barrier to progress lies in the time lag between management

improvements and measurable water-quality recovery, which complicates policy evaluation and weakens farmer confidence. This biogeochemical lag reflects the legacy N accumulated in soils and groundwater from decades of surplus inputs—an issue that current policy timelines often fail to consider. Legacy N can continue to leach or mineralize long after inputs decline, delaying recovery and undermining progress toward WFD targets.

Addressing these challenges requires advanced modelling and scenario analysis tools capable of evaluating alternative mitigation strategies at both farm and national scales. Such models can test combinations of stocking-rate reductions, restricted grazing, or fertilizer substitutions to identify pathways that balance production with environmental protection. Critically, these models must also incorporate legacy N dynamics and site-specific soil and hydrological conditions to predict the true pace of water-quality recovery. By linking detailed farm management data with environmental outcomes, modelling can provide policymakers with evidence-based solutions and realistic timelines, ensuring that expectations for water-quality improvements align with the biophysical realities of legacy storage and release.

The EPA has already taken steps to target action more effectively through Pollutant Impact Potential (PIP) maps for N (Figure 1-3) and P (EPA Catchments Unit, 2016–2021), identifying high-risk catchments where nutrient losses are greatest. Complementary initiatives, such as the Local Authority Waters Program (LAWPRO) and the Agricultural Sustainability Support and Advisory Program (ASSAP), work directly with farmers to design locally tailored solutions. However, Ireland still lacks integrated tools that combine policy modelling, legacy accounting, and real-time monitoring. Without these, achieving meaningful and timely progress toward EU WFD goals, especially under increasing pressure to reduce fertilizer allowances will remain a significant challenge for both regulators and the dairy sector.

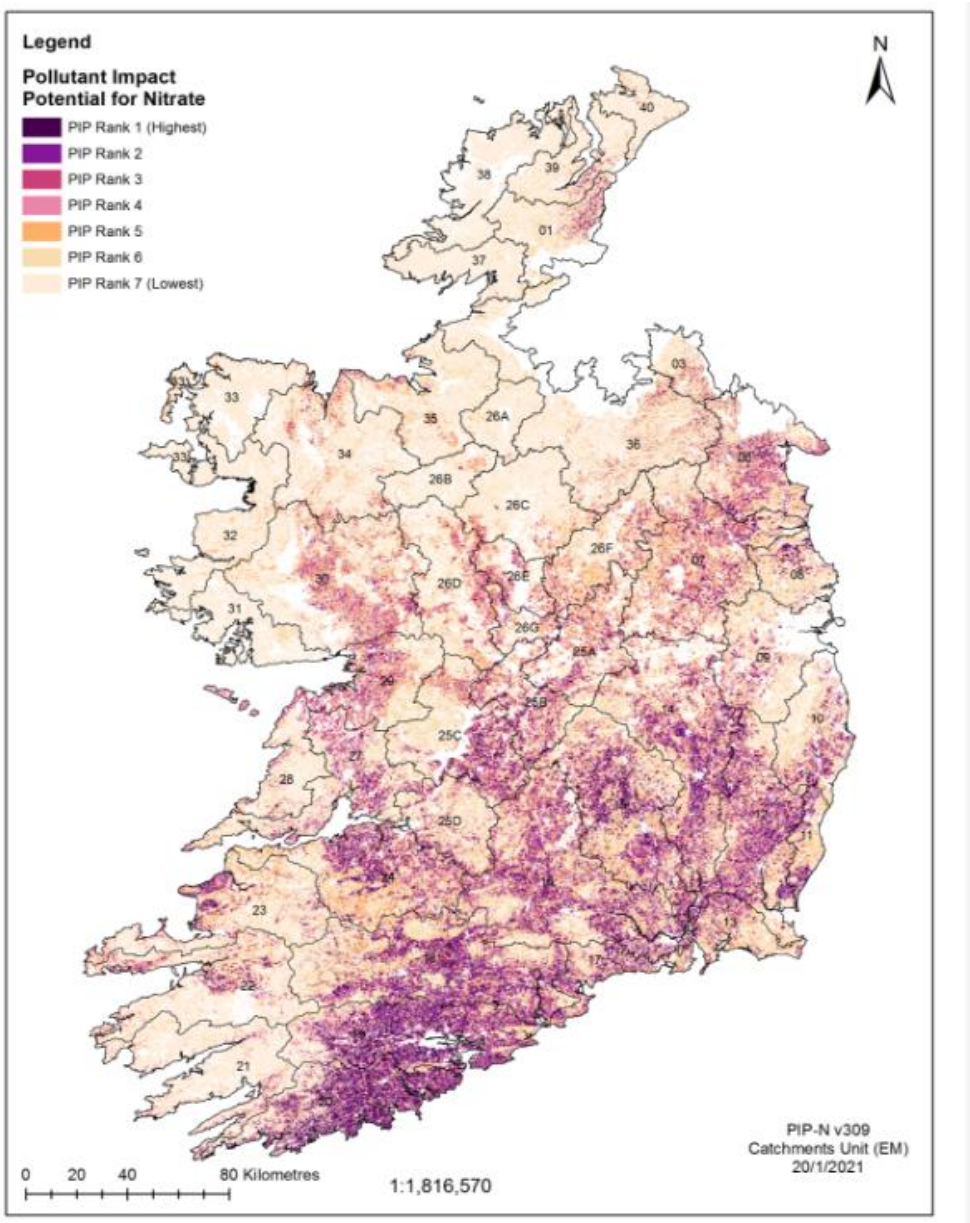


Figure 1- 3 Pollutant Impact Potential Map for Nitrate, Ireland. Source - EPA Catchments Unit using data from 2016-2021.

Persistent NO_3-N concentrations in groundwater, even after reductions in fertilizer inputs, are increasingly attributed to time lags within the soil–water continuum. These lags arise from two distinct but interlinked processes: hydrologic time lags, referring to the time required for recharge water and dissolved NO_3-N to travel through the unsaturated and saturated zones before reaching groundwater (Meals et al., 2010; Van Meter & Basu, 2015), and biogeochemical time lags, associated with the gradual mineralization, immobilization, and release of N stored in soil and subsoil pools (Van Meter & Basu, 2017; Weitzman et al., 2022). Together, these processes create a temporal disconnect between management change and water-quality response. Van Meter and Basu (2015, 2018) demonstrated that when

biogeochemical storage and release processes are considered alongside groundwater travel times, lag periods can extend from years to several decades, delaying visible improvements in water quality even after inputs are reduced. Likewise, Weitzman et al. (2022) showed that deep soil N storage in the vadose zone slows NO_3^- migration and prolongs subsurface retention, effectively creating a long-term legacy reservoir.

While hydrologic legacy effects are widely recognized, relatively few studies worldwide have directly quantified soil legacy N—the biogeochemical component of this lag—and none had done so in Ireland. Understanding and quantifying these dual lags is essential for designing realistic mitigation timelines, improving nutrient management efficiency, and informing credible policy evaluations under the EU ND and WFD.

1.5 Research Gaps

Nitrogen is central to agricultural productivity, yet its mismanagement has led to extensive environmental degradation through NO_3^- contamination of groundwater, eutrophication of surface waters, and gaseous losses to the atmosphere. In Ireland's grass-based dairy systems, the drive for productivity under high stocking rates has intensified N inputs, creating persistent surpluses and challenging compliance with EU environmental directives. Current frameworks for estimating N losses rely heavily on farm nutrient balances and NUE metrics. However, these approaches overlook the complex interactions between detailed farm management, soil processes, and hydrology that govern N dynamics and environmental outcomes.

- Limitations of current modelling frameworks

  Existing models, such as DeNitrification DeComposition (DNDC) (Li, 1996), simulate carbon (C) and N cycling under variable climates but lack resolution to represent the detailed management practices that characterize Irish dairy systems — including slurry application timing, grazing intensity, and fertilizer form. Moreover, these models rarely integrate long-term farm activity data or soil legacy effects, resulting in an incomplete understanding of N cycling under real-world management and climatic conditions. There remains a shortage of models that combine fine-scale farm data, such as land use, manure distribution, and silage cutting frequency with site-specific environmental drivers. This limits our capacity to predict how management changes influence N surpluses, leaching, and emissions.
- Lack of quantification of soil legacy N

Although groundwater NO_3-N contamination and lagged water-quality responses are well-documented, the accumulation, magnitude, and persistence of legacy N in agricultural soils remain poorly quantified. Existing studies focus largely on surface fluxes, neglecting subsurface N storage that can continue to release reactive Nr for years or decades after input reductions. This knowledge gap is critical because legacy soil N represents a substantial and long-lasting contributor to N dynamics, influencing both crop uptake and the delayed response of groundwater quality. Quantifying soil N legacies has never been undertaken at this scale in Irish dairy systems, despite evidence that intensive farming in the south and southeast has generated large surpluses since milk quota abolition in 2015.

- Insufficient understanding of how management, soil, and hydrogeology interact
  Most studies isolate management from biophysical controls, overlooking how historical and current practices interact with soil texture, drainage, and hydrogeological setting to determine N accumulation and release. The influence of denitrification potential, shallow water tables, and artificial drainage on N fate remains particularly underexplored. These interactions determine whether soils act as short-term transmitters or long-term retainers of surplus N, a distinction crucial for designing site-specific mitigation strategies.
- Limited empirical evidence from long-term, contrasting soil systems
  While high-resolution monitoring exists for surface water and groundwater in Ireland (e.g., Teagasc, ACP for 5 meso-catchments around the country), comparable soil datasets are lacking. Limited studies have measured soil N to depth across contrasting drainage and hydrogeological settings. There is an urgent need to pair long-term farm-scale N budgets with deep soil coring and groundwater monitoring to establish how decades of management have shaped subsurface N stores and their environmental expression.
- Policy and mitigation knowledge gaps
  Although mitigation techniques such as nitrification and urease inhibitors (Harty et al., 2017; Monaghan et al., 2009; Roche et al., 2016; Smith et al., 2018) and diverse sward systems (Beukes et al., 2014; Chapman et al., 2008; McCarthy et al., 2020) show promise, their long-term effects on soil legacy N remain untested. Current policy measures under the EU ND and WFD assess progress on six-year cycles that cannot capture the delayed recovery expected in legacy-affected systems. Moreover, uniform rules overlook regional contrasts in soil drainage, hydrogeology, and denitrification potential, limiting the effectiveness of mitigation programs and risking misinterpretation of policy success or failure.

In summary, this thesis addresses the critical gap in understanding how N accumulates, persists, and is mobilized in intensive dairy systems. It combines detailed farm-scale modelling with multi-decadal soil and groundwater datasets to link management practices with environmental outcomes. The overarching aim of this research is to assess N dynamics and environmental outcomes in Irish dairy systems by evaluating mitigation scenarios and quantifying legacy soil N accumulation to understand how current and historical management, drainage class, and hydrogeological setting influence both near-term losses and the pace of environmental recovery.

By integrating modelling, soil coring, and long-term monitoring across two contrasting sites — Teagasc Moorepark Dair Research Farm (known locally as Curtins) (well-drained, karstic soils) and Teagasc Johnstown Castle Dairy Research Farm (variably drained, fine-textured soils) — this work provides the first empirical quantification of legacy soil N in Ireland. It establishes a framework for understanding both the short-term effects of management and the long-term behavior of legacy N, offering actionable insights for scientists, policymakers, and the dairy industry to design effective, region-specific strategies for sustainable N management.

1.6 Research Objectives

The overarching aim of this thesis is to assess N dynamics and environmental outcomes in Irish dairy systems by evaluating mitigation scenarios and quantifying legacy soil N accumulation, to understand how current and historical management, drainage class, and hydrogeological setting influence both near-term losses and the pace of environmental recovery. To achieve this, the research integrates farm-scale modelling, long-term management data, and empirical soil and groundwater analyses across contrasting dairy soil systems. Together, these approaches bridge the gap between process-based understanding and real-world management, providing evidence to inform sustainable nutrient policy and practice.

This thesis is structured around three interlinked research objectives (ROs) outlined in the systems map in Figure 1-4:

- RO1: Analyze the impact of various N mitigation scenarios in grass-based intensive dairy farm systems on environmental benefits and farm production.
- RO2: Quantify the role of legacy N using current and decadal farm management inputs in dairy farming systems located on free-draining soils.

- RO3: Assess the effect of soil properties, farm activity, and hydrogeology on legacy N accumulation and stores across free-draining versus variably draining soils.

Figure 1-4 illustrates how agricultural intensification, soil and hydrological processes, and policy responses interact to shape N cycling and environmental outcomes. It links the underlying drivers ("Why"), system processes ("How"), and the research integration ("Research Objectives") that together frame this thesis. The map highlights how legacy N arises from long-term management and hydrogeological conditions, feeds back into policy and farm management, and informs strategies for improved NUE and realistic recovery timelines under the EU WFD and ND.

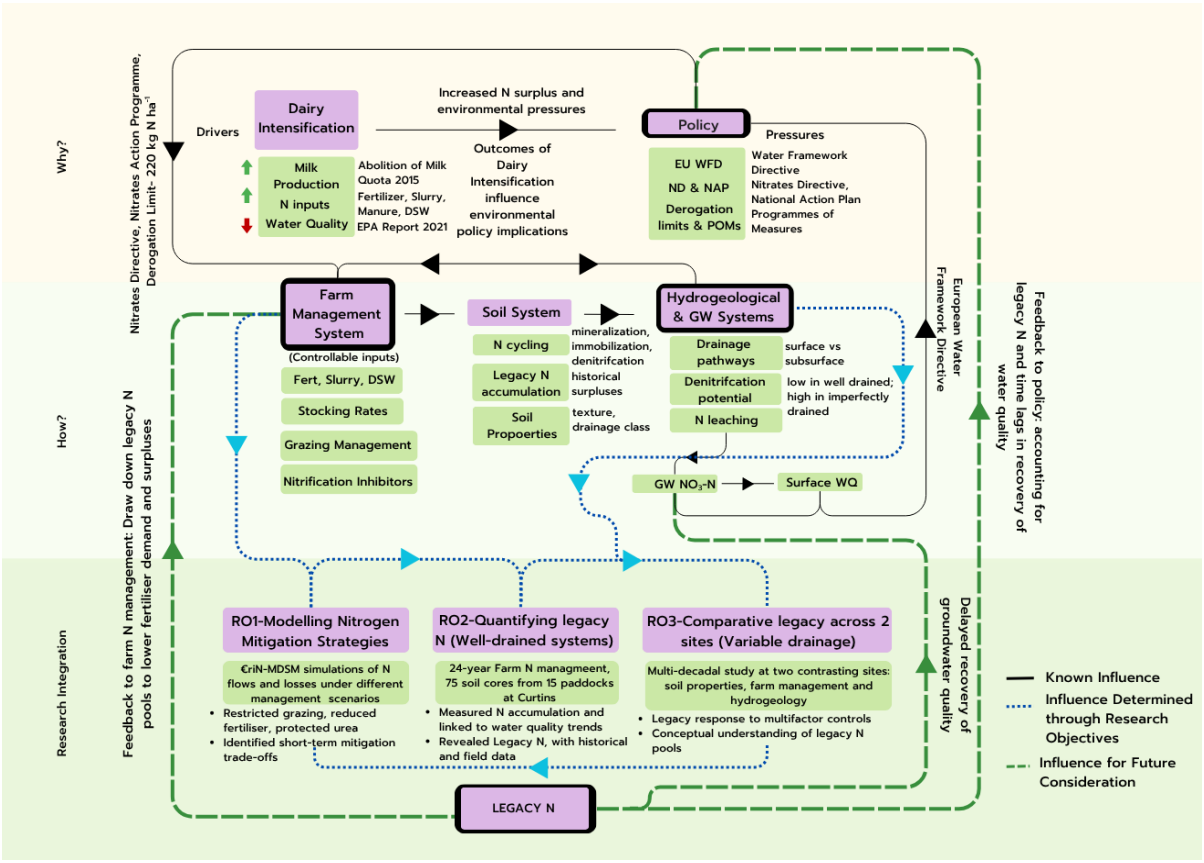


Figure 1- 4 Conceptual systems map of nitrogen (N) dynamics, legacy accumulation, and research integration across Irish grass-based dairy systems.

The central hypothesis of this study is adopting efficient agricultural practices, including manure management, restrictive grazing, reduced fertilizer application rates, the use of nitrification inhibitors, and mixed sward pastures can sustain or enhance crop yield and soil health while simultaneously reducing soil N accumulation and associated N losses to water in

temperate intensive dairy systems. We also hypothesize that decades of intensive farming have resulted in the accumulation of substantial legacy N in soils, which may act as a reservoir for plant uptake and strongly influence current N dynamics. These processes are expected to sustain or increase farm productivity while mitigating environmental N losses and improving our understanding of N fluxes in pasture-based dairy farming. These hypotheses will be tested through a comparative analysis of experimental farms implementing climate-smart practices and control farms managed conventionally. The evaluation will combine field and laboratory campaigns with an Irish grass-based, farm-scale N modelling framework.

To address RO1, I developed multiple scenarios and simulated them using the €riN-MDSM model to evaluate farm-scale N budgets alongside milk production for intensive dairy farms, using a baseline derogation farm in Ireland as a reference. The mitigation strategies tested included varying rates of inorganic and organic N application, the use of urea inhibitors, and restrictive grazing practices applied over extended periods.

To address RO2 and RO3, I designed an integrated field and laboratory program that involved soil coring to depths of up to 1 m to assess N dynamics within the soil profile. In parallel, I collated over 20 years of high-resolution, paddock-level farm management data to quantify and evaluate legacy N and its influence on groundwater NO_3–N concentrations. This combined dataset enabled the assessment of legacy N dynamics under intensive dairy farming across contrasting soil drainage classes in the south and southeast of Ireland.

Chapter 2

Simulation of Management Practices to Reduce Nitrogen Losses to Water and Air on Well Drained Grass-based Dairy Farms in Derogation

This chapter is a mirror of the following published article

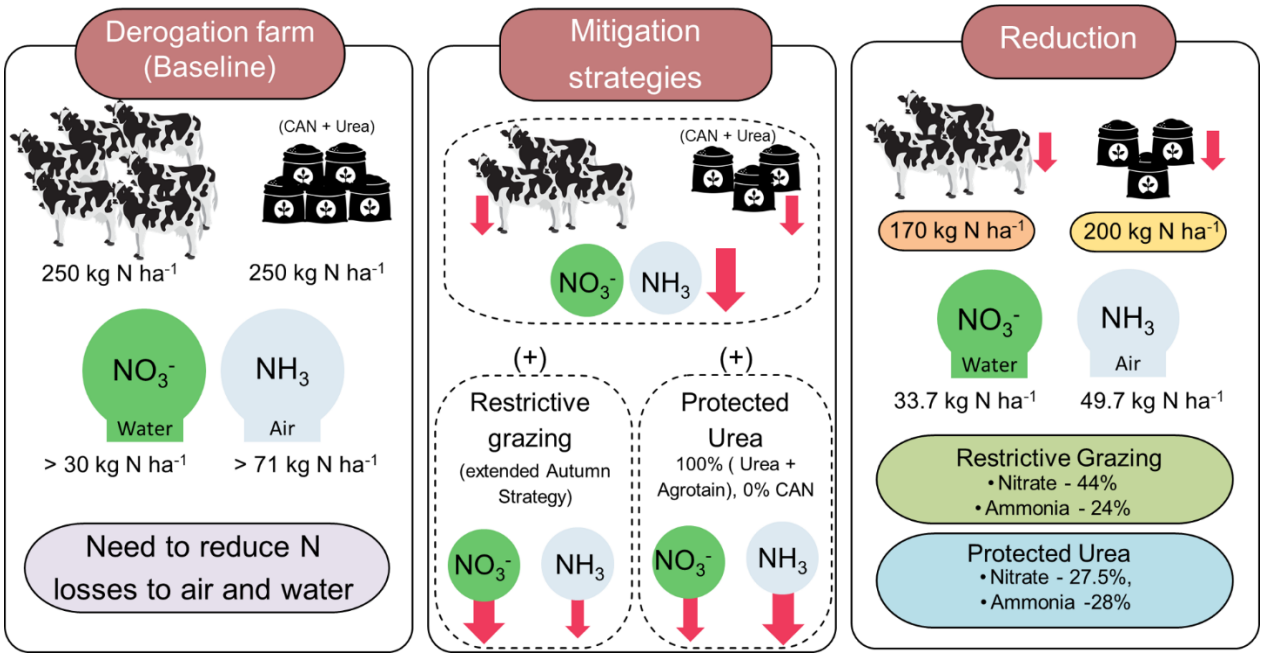
Lakhanpal, G., Basu, N. B., O'Brien, D., Buckley, C., Horan, B., Richards, K. G., & Fenton, O. (2025). Simulation of management practices to reduce nitrogen losses to water and air on well-drained grass-based dairy farms in derogation. *Journal of Dairy Science*. April 18, 2025. https://doi.org/10.3168/jds.2024-25853

2.1 Overview

A greater understanding of nitrogen (N) flows between productive use in dairy farms and environmental losses can inform regulation, support policy and manage expectations around delivery of good water and air quality. Here, we use the €riN-MDSM model that was built to simulate N flows and losses for Irish grass-based dairy farms to quantify N flows and losses for a well-drained dairy farm at derogation (i.e., stocked at the maximum allowable rate of 250 kg N ha^{-1}) in Ireland and then propose alternate scenarios that can minimize environmental losses. The model is capable of simulating nitrate (NO_3^-), ammonia (NH_3), nitrous oxide (N_2O) and dinitrogen (N_2) emissions from urine, dung, slurry, dairy soiled water and fertilizer N under a range of management practices. Specifically, we propose alternative systems around derogation by varying inorganic (200 or 225 kg N ha^{-1}) and organic (170, 230, 268, 340, 430 kg N ha^{-1}) fertilizer rates; substituting calcium-ammonium-nitrate (CAN) with protected urea (i.e., urea + Agrotain) and implementing restrictive grazing for vulnerable time periods when losses are the greatest (e.g. October to February or September to February). €riN-MDSM estimated that at derogation the farm N surplus was 241.3 kg N ha^{-1}, leached loads were 46.6 kg N ha^{-1} (target for good water quality outcomes is 30 kg N ha^{-1}) and gaseous emissions were 127.3 kg N ha^{-1}. Better environmental outcomes were observed by reducing stocking rates to 170 kg N ha^{-1} and 230 kg N ha^{-1} decreasing leaching losses by 21.5 % and 6.4 %, respectively. Further reductions in inorganic fertilizer to 200 kg N ha^{-1} decreased leaching by 27.5 % and ammonia emissions by 31 %. By substituting CAN fertilizer with protected urea, ammonia emissions decreased by 5.2 % from derogation baseline. Further reductions were possible by varying stocking rates and fertilizer rates together. Restrictive grazing suggestively decreased

NO_3^- leaching to groundwater, with reductions from derogation equivalent to 38.3 % and 28.8 % for 170 kg N ha^{-1} and 230 kg N ha^{-1} stocking rates, respectively. Further reductions in inorganic fertilizer to 200 kg N ha^{-1} resulted in a 44.3 % decrease in NO_3^- leaching to groundwater (~30 kg N ha^{-1} target) and a 24 % decrease in ammonia emissions to air. Future systems need to consider a combination of reduced fertilizer rates, restricted grazing, and use of protected urea to minimize N losses, especially during high rainfall periods. Although the results observed demonstrate opportunities to reduce N losses from grazing systems from a combination of reduced fertilizer rates, restricted grazing, and use of protected urea, the impacts of these mitigations must also be considered in terms of economic cost to farms. On that basis, future system trials should monitor flows and losses from the implementation of such mitigations in addition to the economic impacts to chart a way forward for better water and air outcomes while maintaining the profitability of Irish dairy farms.

Graphical Overview



2.2 Introduction

Over the past 100 years, human activities have severely contributed to a 14-fold increase in reactive nitrogen (Nr) in the environment (Carpenter et al., 1998). Anthropogenic activities such as industrialization, extensive use of mineral fertilizers for intensive agriculture and fossil fuel combustion have contributed to a dramatic increase in nitrogen (N) pollution (Brown et al., 2005; de Klein et al., 2017; Chen et al., 2018). This in turn contributes to various

environmental issues such as coastal eutrophication, groundwater contamination, soil acidification and greenhouse gas (GHG) emissions (Arsenault et al., 2009; O'Brien et al., 2012). In the European Union (EU), the livestock supply chain accounts for 82 % of agricultural ammonia (NH_3) emissions and 73 % of all agricultural emissions to water bodies (Leip et al., 2015). In the Republic of Ireland (henceforth called Ireland), according to latest Environmental Protection Agency (EPA) reports, agriculture has been identified as the most prevalent pressure for water quality, impacting in excess of 1000 water bodies (i.e., 60 % of all water bodies) "at risk" of not achieving their environmental objective under the EU Water Framework Directive (WFD). At the national scale, the proportion of the N load in waters from agriculture (pasture and arable) is 84 %, with the remainder from urban and domestic wastewater, industry and other sources (EPA, 2024).

Intensive dairy systems attempt to optimize milk production through better management of animal nutrition, breeding and health (Gourley et al., 2011; Akert et al., 2020; Wang et al., 2014). About 25-30 % of N consumed by lactating cows is converted to milk while the remainder is returned to soil via excrement through feces and urine (Wilkerson et al., 1997). When applied as organic fertilizer to pasture, manure provides a great source of N for crop growth. The primary sources of Nr pollution on intensive dairy farms occur from inorganic and organic fertilizer application (Fang et al., 2020; Ledgard et al., 2019). Manure produced by dairy cows is nutrient rich and if it is stored improperly or excessively applied and at wrong times, can lead to major groundwater and soil contamination (Di and Cameron 2002; Brito and Silva 2020) especially in terms of nitrate (NO_3^-) leaching to groundwater and NH_3 losses to air.

In order for NO_3^- to leach to groundwater, NO_3^- needs to be present in the soil when percolating water (i.e., drainage) migrates through the soil profile (Fenton et al., 2009; Hoekstra et al., 2020). Several factors influence NO_3^- concentrations in soil on dairy farms including soil type, precipitation, inorganic and organic fertilizer application rates, inorganic fertilizer type, fertilizer application timing and amount of time spent by animals grazing which accounts for urine and dung deposits in paddocks. At a policy level, the Commission of the EU have targeted a reduction of nutrient losses to water by at least 50 %, while ensuring that there is no deterioration in soil fertility. This will require a reduction in the use of inorganic fertilizers by at least 20 % by 2030 (European Commission 2019). Mitigation strategies to reduce N losses to the environment include changes to the type and rates of fertilizers being applied to grassland. For example, calcium-ammonium-nitrate (CAN) is the standard inorganic fertilizer used on dairy farms in Ireland (containing ~27 % N), consisting of half ammonium-N and half

NO_3^- and is therefore riskier in terms of NO_3^- leaching losses to groundwater. In contrast, newly developed protected urea fertilizer contains a urease inhibitor within or on the coating of the fertilizer granule, enabling full conversion of N to the NH_3 form. This means when applying protected urea there is less NO_3^- available to be lost in the leached pathway. Protected urea encompasses diverse formulations, among which Agrotain serves as a stabilizer to mitigate volatilization, thereby enhancing N availability for plant utilization. The key active component, N-(n-Butyl) thiophosphoric triamide (NBPT), operates as a urease inhibitor within Agrotain, effectively regulating urea hydrolysis and subsequent N release. Protected urea has proven benefits over CAN; it is cheaper on an elemental N basis, and, while it may appear slightly more expensive than unprotected urea, it will give the same "effective N" for the plant as urea, at a 12 % lower spreading rate (Rahman et al., 2021; Ray et al., 2021). The use of protected urea reduced average NH_3 emissions by 78.5 % of applied N (Forrestal et al., 2016). It also produced consistent yield and N uptake efficiency similar to CAN (Harty et al., 2017) while effectively lowering N_2O emissions (Harty et al., 2016).

Other mitigation strategies aim to reduce large N loads at specific times during the grazing season. For example, urine patches are associated with high N loss to groundwater e.g. research shows that N load in a single patch can be up to 1,000 kg ha^{-1}, with research on dairy farms in Ireland showing a range from 300-500 kg ha^{-1} (Dennis et al., 2011; Richards et al., 2005; Hoekstra et al., 2018). The coverage of urine patches in any paddock can be as large as between 14-20 % (Dennis et al., 2011; Maire et al., 2018; Jolly et al., 2021; O'Neill et al., 2021). In terms of timing, urinations in autumn in temperate regions (September-November) increase the risk of high N load losses in leachate to groundwater. With regard to mitigation strategies, reducing the concentration of N and the numbers of urine patches can be achieved by restricting grazing access during these months. Similar approaches using restricted grazing (or on-off grazing) have been investigated in spring or summer with field trials in Denmark (Hansen et al., 2012) and New Zealand (Christensen et al., 2012) in the past, with mixed outcomes. In the New Zealand study, NO_3^- leaching from the duration controlled (4 hour) plots was 65 % lower compared with the standard duration (7 hour) plots, (i.e., 2.8 kg N ha^{-1} and 8.0 kg N ha^{-1} for the duration controlled grazing and standard grazing plots, respectively).

In the context of dairy farms in Ireland, the average grazing season lasts 235 days per year (Teagasc, 2021). Therefore, implementing restricted grazing access, where animals are held off pasture for 14 hours per day between grazing bouts, has the potential to maintain overall grazing season length and reliance on grazed pasture. This approach could capture a

significant portion of urine production, particularly during periods of high rainfall and waterlogged soils in autumn, thereby reducing NO_3^- leaching. A longer-term strategy for restricted grazing, especially during autumn and transitioning to winter, may offer additional benefits. In Ireland, there has been a 7 % increase in annual rainfall over the last 30 years (1991-2020 as compared to 1961-1990) with an average annual rainfall of 1288 mm (Met Éireann 2024) and 39.2 % falling between September and December. Given recent years' wet conditions in Ireland impacting grass growth, sediment loss, and deposition of urine and dung in pastures, further investigation into restrictive grazing is warranted covering a part of or the entire wet period. New Zealand, with an environment and climate very similar to Ireland, has established a threshold of 30 kg N ha^{-1} for N leaching losses to meet water quality targets (Beukes et al., 2012). Values exceeding this threshold pose environmental risks and may compromise these targets. Therefore, it is crucial to reduce leaching losses below this limit to ensure environmental standards are met (Beukes et al., 2012; Norris et al., 2019; McDowell et al. 2021).

Many studies continue to use the N footprint approach with models such as DairyMod (Harrison et al., 2017), DayCent (Del Grosso et al., 2000; Parton et al., 1998), EPIC (Izaurralde et al., 2012), and DeNitrification-DeComposition (DNDC) (Li, 1996, Li et al., 1997, 2006). These models provide a holistic assessment of N implications across the production chain but may obscure the impact of individual mitigation measures on farm production systems. The €riN model (Hoekstra et al., 2020) offers a more detailed simulation of farm management effects at the individual farm level. It dynamically integrates the Moorepark Dairy System Model (MDSM) (Shalloo et al., 2004) with a grass growth module and N loss modules, operating on a monthly time step to generate yearly outputs. This integration presents several advantages over the N footprint approach, enabling the simulation of various farm management practices and their effects on nitrogen use efficiency (NUE), losses, and economic outcomes within an Irish grass-based dairy system.

The objectives of the study is to analyze the effect of various mitigation strategies on N losses from a dairy farm using the €riN-MDSM model. The derogation baseline scenario is for a derogation dairy farm on well drained soils representing farms in south and south-east Ireland, with inorganic fertilizer application rates of 250 kg N ha-1, and an organic fertilizer rate of 250 kg N ha-1. This acts as the study baseline, replicating a dairy farm on well-drained soil in the south or south-east of the country, stocked at the maximum allowable rate under the Nitrates Derogation in 2021. The organic rate of 250 kg N ha^{-1} is based on a maximum stocking

rate of 107 cows per hectare, which is the maximum allowable rate. The first set of scenarios (Scenario I) involved altering the amounts of inorganic and organic fertilizers, with inorganic fertilizer application rates of 200 kg N ha^{-1} (Low; L), 225 kg N ha^{-1} (Medium; M), and 250 kg N ha^{-1} (High; H), and organic fertilizer application rates of 170 kg N ha^{-1} (Very Low Stocking; VLS), 230 kg N ha^{-1} (Low Stocking; LS), 250 kg N ha^{-1} derogation (Baseline; B), 268 kg N ha^{-1} (High Stocking; HS) , 340 kg N ha^{-1} (Very High Stocking; VHS), and 430 kg N ha^{-1} (Extremely High Stocking; EHS). The second set of scenarios (Scenario II) focused on exploring the effect of change of type of inorganic fertilizer application from derogation baseline. Scenario II was implemented in addition to Scenario I with transitioning the inorganic fertilizer application solely (100 %) to protected urea (urea + Agrotain) and 0 % Urea and CAN, departing from a urea and CAN combination. The third set of scenarios focused on exploring the effect of restrictive grazing (Scenario III), implemented in addition to derogation baseline where restrictive grazing is applied from December to February with imposing duration-restricted grazing practices during the autumn months (October-February) and an extended autumn strategy for a longer period (September-February).

2.3 Materials and methods

2.3.1 €riN-MDSM model

The €riN model (Hoekstra et al., 2020) is a computer model that simulates monthly N flows on an Irish grass-based dairy farm. The model dynamically links the Moorepark Dairy System Model (MDSM) (Shalloo et al., 2004), to provide a grass growth module and a number of N loss modules in order to simulate the effect of farm management on NUE, N losses and economic performance. Hoekstra et al. (2020) modelled N losses using €riN for an average representative dairy farm in Ireland and provides a comprehensive account of the modules in €riN and the MDSM and how they are linked.

Briefly, the main input parameters for the €riN-MDSM model are location, grassland area, soil drainage class (well, moderate or poorly drained), inorganic fertilizer levels, manure application rates and pattern, reseeding rates, grazing and silage dates. The model splits a dairy farm into three nominal areas: 1) grazing only; 2) grazing with one-cut of silage and 3) grazing with two-cuts of silage. The model computes actual grass supply according to the seasonal pattern of growth and uses fertilizer, manure and reseeding rates to modify the maximum level of grass production; 17.5 t DM ha^{-1} (O'Donovan et al., 2011). Additional calculations in the MDSM simulate the amount of grass produced in the grazing and silage areas of the farm.

Demand for grazed grass and grass silage determines the proportion of grass utilized on farm. €riN assigns monthly utilization rates to grazed grass and applies a fixed proportion (e.g., 75 %) to grass silage. The MDSM quantifies grass demand based on the herd's dietary requirements and the nutritional value of forages and supplements. The nutritional module of this model requires data on milk yields, body weights, pregnancy rates, calving pattern, replacement rates, housing periods and concentrate feeding levels. The net energy (NE) system of O'Mara (1997) is applied to determine the energy demand of the herd in lactation feed units (UFL). This data along with records of concentrate supplementation are used to calculate the amount of NE from forage, which is related to housing and grazing periods to determine the NE supplied by grass silage and grazed grass. The NE supplied by herbage is converted to DM intake based on the energy density of forages and added to concentrate feed to estimate the total DM demand. The MDSM multiplies feed demand by the N content of the components of the diet to calculate an animal's intake of N. N outputs are calculated from milk and cattle sales based on the N content of these products. N excretion was computed as the difference between N intake and N outputs.

€riN takes N inputs, such as slurry, silage, and purchased animals, as well as N outputs, including milk production and animals sold or deceased, from the MDSM module to estimate N excretion and associated N losses. The model accounts for N excreted on passageways and on collecting yards i.e., dairy soiled water (DSW), and partitions N excretion between dung and urine based on the research of Kebreab et al. (2001). €riN-MDSM calculates NO_3^-, NH_3, nitrous oxide (N_2O) and dinitrogen (N_2) emission loads from urine, dung, slurry, DSW and fertilizer N by multiplying the N load (kg N ha^{-1}) with emission factors from empirical Irish research. All losses such as NO_3^- leaching and NH_3, N_2O and N_2 emissions are expressed in kg N ha^{-1}. NO_3^- emission factors vary by soil drainage class and by season for nutrient applications in the form of urine and DSW. NH_3 emission factors for urine, dung and slurry are sensitive to timing and storage facilities, and N_2O loss factors depend on soil drainage class. N_2 emissions are estimated using N_2 to N_2O loss ratios. Excess or surplus N not vulnerable to leaching, volatilization or denitrification is returned to the soil N pool. €riN-MDSM uses a mass flow approach to calculate the soil N balance and computes farm N surplus as the difference between N imports and N exports.

Comprehensive validation of the €riN model is constrained by limited datasets covering all input and output parameters (Hoekstra et al., 2020). However, its sub-models have undergone prior evaluation and are published in peer-reviewed literature. The model was

assessed against N balance data from Burchill et al. (2016) for an intensively managed Irish grass-based dairy system, demonstrating agreement in N output, environmental losses, and NUE. Variations in N_2 and NO_3^- losses were attributed to differences in soil types. NH_3 emissions aligned with published dairy system estimates, while NO_3^- leaching rates were consistent with values reported in Ireland and New Zealand. Despite inherent uncertainties, N_2 emissions fell within the limited range of existing studies, suggesting the model provides realistic N flow estimates for Irish dairy systems. A preliminary sensitivity analysis identified minimal differences in environmental losses compared to the scenario analysis, with model outputs remaining within published ranges, reinforcing its reliability. Given this redundancy, scenario analysis was deemed sufficient for evaluating model responses.

2.3.2 Derogation dairy farm simulation

The EU Nitrates Directive (ND) (OJEC, 1991) is a regulation designed to restrict the use of agricultural fertilizers to agronomic optimal levels, with the goal of reducing excess N and phosphorus (P) losses into aquatic ecosystems. In every member state, the ND is incorporated into a set of Programs of Measures (PoMs) under the WFD (OJEC, 2000). Applying population weights from the Central Statistics Office (CSO) based on farm size and system, the Teagasc National Farm Survey (NFS) represents over 15,000 specialist dairy farms, as classified under the EU Farm Accountancy Data Network (FADN) typology (Dillon et al., 2023). The 2020 Census of Agriculture for the Republic of Ireland (CSO, 2022) reported 15,319 specialist dairy farms, indicating that the population-weighted NFS sample provides a strong representation of the national dairy sector. Between 2021 and 2023, NFS data show that 44–49 % of dairy farms operated under a Nitrates Derogation. Environmental losses from these farms are significantly higher compared to non-derogation farms in Ireland, highlighting the need for targeted mitigation strategies.

A Baseline stylized dairy farm has been established under the National Action Program (NAP) (OJEC, 2000) in Ireland and was previously set up in €riN and published by Hoekstra et al. (2020) (Table 2-1). The present study established a likely baseline for derogation dairy farms in Ireland (i.e. represents 44-49 % of dairy farms in the above context) and examines strategies to reduce N losses. Derogation represents grass-based dairy farms with a nitrates derogation operating at the maximum allowable stocking rate, specifically in the south and south-eastern part of Ireland. The farm is located on a well-drained soil and is notionally 40 ha with an 11 % reseeding rate. It accommodates an average of 107 cows, resulting in a stocking

rate of 2.75 livestock units ha-1 (equivalent to one cow per ha) as per Dillon et al. (2018). The farm received 250 kg inorganic N ha^{-1} (urea and CAN) and had an organic load of 250 kg N ha^{-1} (from stocking rate of the animals on farm), respectively. Soil P and potassium (K) levels are set to represent optimum conditions in Index 3 (i.e., optimum) and are within the recommended pH level (6 to 6.5). In terms of fertilizer, 106 kg N ha^{-1} is applied on grassland for 1st cut silage and 100 kg N ha^{-1} for 2nd cut silage. The slurry application pattern is set at 70 % spring and 30 % summer. Milk yield averaged 5444 L cow^{-1} with milk fat at 4.2 % and milk protein at 3.5 %.

Table 2- 1 Derogation Baseline Dairy Farm as inputted into €riN-MDSM.

Factor	***Parameter***	***Units***	***Derogation Baseline Farm***
Grassland	Farm Size	ha	40
	Dominant Soil drainage class		Well drained
	Soil P	mg L^{-1}	7 (equivalent to Index 3, optimum)
	Soil K	mg L^{-1}	120 (equivalent to Index 3, optimum)
	Soil pH		6 (optimum)
Herbage	Species		Perennial ryegrass
	Reseeding rate	% grassland	11
	Fertilizer N	kg N ha^{-1}	250
	CAN	% fertilizer N	88%
	Urea	% fertilizer N	12%
	Fertilizer P	kg P ha^{-1}	16
	Fertilizer K	kg K ha^{-1}	23
	Organic N load	kg N ha^{-1}	250
	First cut silage	ha	18.7
	Second cut silage	ha	11.2
	Grass yield	t Dry Matter ha^{-1}	13.8
Herd	Dairy breed		Friesian
	Cow Number (mean)		107.5
	0-1 year (mean)		8.9
	Stocking rate	Livestock Units ha^{-1}	2.75
	Contract rearing	% replacements	100
	Replacement rate	%	21%
	Age at first calving	months	24
	Mean calving date		15th February
	Grazing season	days	275
	Pasture not grazed		December- February
	Concentrate	kg Dry Matter cow^{-1}	932
	Silage purchases	kg Dry Matter cow^{-1}	8
Milk	Fat	%	4.2
	Protein	%	3.5
	Milk yield	kg cow^{-1}	5,607
	Milk solids yield	kg cow^{-1}	432
Slurry	Application method		Broadcast using a splash plate
	Application rate/Distribution	m^3 ha^{-1}/% Spring	16.3/70

2.3.3 Scenario I: Reducing Inorganic and Varying Organic Fertilizers

In Ireland, chemical fertilizers typically comprise urea and CAN (with higher proportions of CAN as compared to urea). The present study explored three chemical fertilizer rates: 200, 225, and 250 kg N ha^{-1}, representing low fertilizer (LF), medium fertilizer (MF), and high fertilizer (HF) rates, respectively. The upper limit here corresponds with derogation baseline (defined previously) (Table 2-2). The quantity of organic fertilizer is directly linked

to the farm stocking rate (S), reflecting the number of animals per unit area over a specified period. Coefficients related to the number of livestock units and adjustments for age-related weight were considered. Literature-based coefficients to estimate the amount of manure produced by farm animals which are integrated within €riN-MDSM were used.

Table 2- 2 Eighteen simulations with varying inorganic fertilizer application rates and stocking rates (which sets the organic fertilizer rate).

Scenario	***Fertilizer[a]***	***Stocking Rate[b]***
	Inorganic Simulation	**Organic Simulation**
	kg N ha^{-1}	**kg org N ha^{-1}**
VLSHF	250	170
VLSMF	225	170
VLSLF	200	170
LSHF	250	230
LSMF	225	230
LSLF	200	230
B	250	250
BMF	225	250
BLF	200	250
HSHF	250	268
HSMF	225	268
HSLF	200	268
VHSHF	250	340
VHSMF	225	340
VHSLF	200	340
ESHF	250	430
ESMF	225	430
ESLF	200	430

Abbreviation used: an Inorganic fertilizer only as a proportion of Urea and CAN, b Organic fertilizer defined by stocking rates on the farm, E=extreme, VH=very high, VL=very low, S=stocking rate, LF=low fertilizer rate; MF-medium fertilizer rate, HF= high fertilizer rate, B=Baseline (Derogation)

Stocking rates were evaluated below and above derogation baseline (B, 250 kg N ha^{-1}): very low stocking (VLS) at 170 kg N ha^{1}, low stocking (LS) at 230 kg N ha^{-1}, high stocking (HS) at 268 kg N ha^{-1}, very high stocking (VHS) at 340 kg N ha^{-1}, and extremely high stocking (EHS) at 430 kg N ha^{-1} (Table 2-2). The different scenarios e.g. very low stocking rate paired with high fertilizer (VLSHF) equates to 170 kg organic N ha^{-1} with 250 kg N ha^{-1} inorganic fertilizer. Similarly, extremely high stocking rate with a low fertilizer application rate (ESLF) equates to 430 kg organic N ha^{-1} with 200 kg N ha^{-1} inorganic fertilizer. The stocking rate of 268 kg N ha^{-1} reflects the actual organic N load observed in 2021 prior to legislative changes

in N excretion rates from 85 to 95 kg N cow^{-1}. Stocking rates of 340 and 430 kg N ha^{-1} were used to simulate the N levels typically found on the milking platforms of dairy farms, with lower N rates applied to other farm areas not accessible by cows. The scenarios were created within input files, which were subsequently imported and executed as macros in €riN-MDSM. Each of the 18 scenarios outlined above were simulated individually within €riN-MDSM. The model utilized emission factors to estimate various components of the farm N balance. In the farm N balance, silage and slurry imports or exports are represented by positive or negative values, respectively.

2.3.4 Scenario II: Protected Urea and Organic Fertilizer

Derogation baseline involves the application of a combination of urea and CAN to stimulate grass growth. Scenario II, proposes protected urea (100 %) as the sole chemical fertilizer N. Protected urea (urea + Agrotain) is known for its effectiveness as an inhibitor, considerably reducing NH_3 emissions and leaching losses compared to conventional urea and N_2O emissions compared to CAN fertilizer. To implement this scenario, we adjusted all CAN proportions per month to zero, redirecting all inorganic fertilizer applications to protected urea (Table 2-3).

Table 2- 3 Nitrogen fertilizer application strategy for Derogation Baseline in €rin-MDSM

Grassland management	***Month***	***Application rate***
		kg N ha^{-1}
Grazing	February	25
	March	38
	April	47
	May	32
	June	32
	July	30
	August	20
	September	25
Grass silage		
1st cut	April	106
2nd cut	June	100

2.3.5 Scenario III: Duration Restrictive Grazing

Restricted grazing in a derogation baseline farm scenario is limited in the autumn months from December to February with 275 grazing days. In this study two restricted grazing

durations were simulated in addition to the baseline. The first period was from October-February, with 215 grazing days and the second was extended further from September-February, with 185 grazing days with the assumption that cows would take about 33 % of their usual time to achieve the same dry matter intake. Consequently, we modified the factor for daily grazing time (time spent by animals feeding outdoors) to 0.33, allowing animals to graze for 8 hours in the fields and spend the remaining time indoors in housing. This adjustment reduced N deposited as urine and dung on pasture, directing this N towards slurry collection and storage during autumn. The shift of urine and dung to stored slurry leads to the model to internally optimize fertilizer application. With larger quantities of slurry available in the restrictive grazing practice, the model preferentially shifts towards utilizing organic slurry as fertilizer over inorganic fertilizers to meet the required grass growth and stores slurry for spreading in the following spring season. Moreover, the N content in slurry is lower when restrictive grazing practices are used, and hence losses from housing increase simultaneously.

2.4 Results

2.4.1 N Balance

For the representative derogation dairy farm, the average farm-gate level N surplus was 241.3 kg N ha^{-1}, with total N inputs of at 340.5 kg N ha^{-1}, and N outputs of 99.2 kg N ha^{-1} (Table 2-4). N inputs comprises of concentrate feedstock, forage feed, fertilizer, atmospheric deposition, replacement animals and slurry import, while N outputs include milk and meat sales (Figure 2-1) The total N losses, including NO_3^- leaching losses, NH_3 emissions, N_2, and N_2O emissions, were observed at 172.5 kg N ha^{-1}.

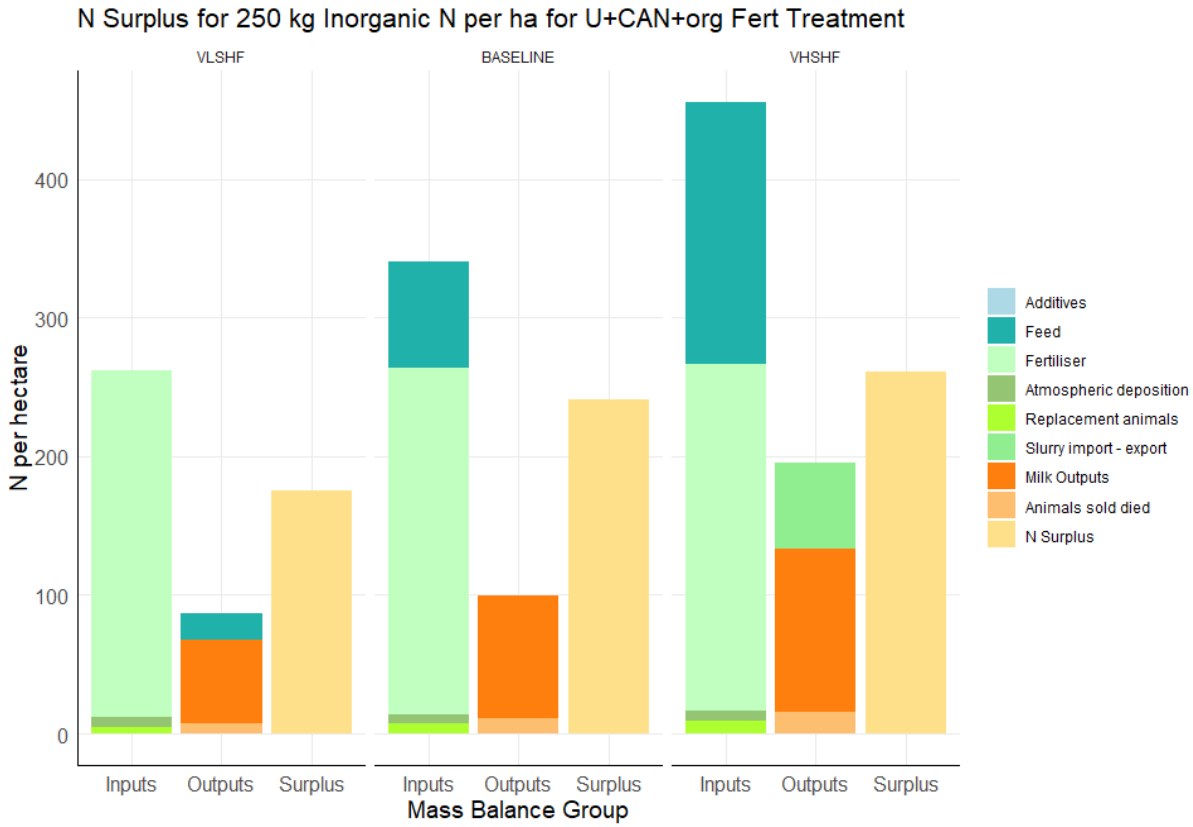


Figure 2- 1 Comparison of nitrogen surplus, inputs and outputs for different stocking rates (SR) at high fertilizer (HF) application rate of 250 kg N ha^{-1}: very low stocking (VLS) rate, Baseline (Derogation), and very high stocking (VHS) rate.

Abbreviations used: N: Nitrogen, SR: Stocking Rate, ha: hectares, Fert: fertilizer, Org: organic.

Table 2- 4 Nitrogen surplus, outputs, losses and milk output for grass-based dairy system across various scenarios of organic and inorganic fertilizer application rates with three different treatments: 1) Urea + CAN + Organic Fertilizer, 2) Protected Urea+ Organic and 3) Restrictive grazing (Sep-Dec).

Scenario	***Inorg Fert***	***Org Fert***	***Milk Output***			***Total Output***			***N surplus***			***Total N losses***		
			1	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3***
VLSHF	250	170	59.6	59.6	59.4	67.3	67.3	67.1	175.6	175.6	171	129	128.6	117
VLSMF	225	170	59.6	59.6	59.4	67.4	67.4	67.1	163.9	163.9	159.3	126.1	125.7	114
VLSLF	200	170	59.6	59.6	59.3	67.3	67.3	67	139	139	134.5	123	122.7	110.9
LSHF	250	230	79.7	79.7	79.3	90	90	89.6	219.5	219.5	213.3	159.5	159	143.5
LSMF	225	230	79.7	79.7	79.3	90	90	89.6	203.7	203.7	197.5	156.8	156.4	140.7
LSLF	200	230	79.7	79.7	79.2	90	90	89.4	185.1	185.1	178.9	154	153.4	137.8
B	250	250	87.8	87.8	87.4	99.2	99.2	98.7	241.3	241.3	234.5	172.5	171.7	154.8
BMF	225	250	87.8	87.8	87.4	99.2	99.2	98.7	224.1	224.1	217.2	169.6	168.8	151.8
BLF	200	250	87.8	87.8	87.2	99.2	99.2	98.5	207.6	207.6	200.7	166.5	165.9	148.5
HSHF	250	268	93.7	93.8	93.2	105.8	105.8	105.3	258.1	258.1	250.8	182	181	163
HSMF	225	268	93.8	93.8	93.3	105.9	105.9	105.4	241.7	241.7	234.3	179.3	178.4	160.2
HSLF	200	268	93.9	93.9	93.4	106.1	106.1	105.5	226.6	226.6	219.3	176.5	175.6	157.4
VHSHF	250	340	118.1	118.1	117.5	133.3	133.3	132.5	260.7	260.7	253.8	194.1	193	172.4
VHSMF	225	340	118.6	118.6	117.8	133.8	133.8	132.9	243.9	243.9	237.2	191.3	190.2	169.6
VHSLF	200	340	118.8	118.8	118.0	133.9	133.9	133	229.4	229.4	223.4	187.4	186.4	166.4
ESHF	250	430	148.7	148.7	147.7	167.7	167.7	166.5	274	274	259.9	217.2	216	186.7
ESMF	225	430	149.2	149.2	148.1	168.1	168.1	166.9	253	253	240.2	213.9	212.9	184.3
ESLF	200	430	149.6	149.6	148.4	168.5	168.5	167.2	229	229	218	210.2	209.2	181.6

*€riN-MDSM model optimization accounts for more slurry distribution over inorganic fertilizer application to support grass growth and efficient N management in restrictive grazing. Abbreviation used in Table: N: Nitrogen, ha: hectares, CAN: Calcium Ammonium Nitrate, Fert: fertilizer, Inorg: Inorganic, Org: organic, E=extreme, VH=very high, VL=very low, S=stocking rate, LF=low fertilizer rate; MF- medium fertilizer rate, HF= high fertilizer rate, B=Baseline (Derogation).

Considerable changes were observed in N inputs, outputs, and surplus values with variations in inorganic fertilizer application rates and stocking rates. Under Scenario I (Urea + CAN + Organic Fertilizer system), N inputs ranged from 206 kg N ha^{-1} to 441 kg N ha^{-1} for VLSLF to VHSHF scenarios, while N outputs ranged from 67.3 kg N ha^{-1} to 168.5 kg N ha^{-1} (Table 2-4). A reduction in inorganic fertilizer application rates to 225 kg N ha^{-1} and 200 kg N ha^{-1} from the derogation baseline resulted in a 5 % and 10 % reduction in N inputs, respectively (Table 2-4), while having no impact on N outputs in meat and milk. However, reductions in stocking rates to 230 kg N ha^{-1} and 170 kg N ha^{-1} led to a decrease in N inputs by 9 % and 28.6 %, as well a decrease in N outputs by 9.2 % to 32 %, respectively. Increasing stocking rates had a positive impact on both N inputs and outputs, with an increase of organic fertilizer application rate to 268, 340 and 430 kg N ha^{-1} resulting in 6.6 %, 34.3 % and 68.9 % more N outputs, respectively (Figure 2-2).

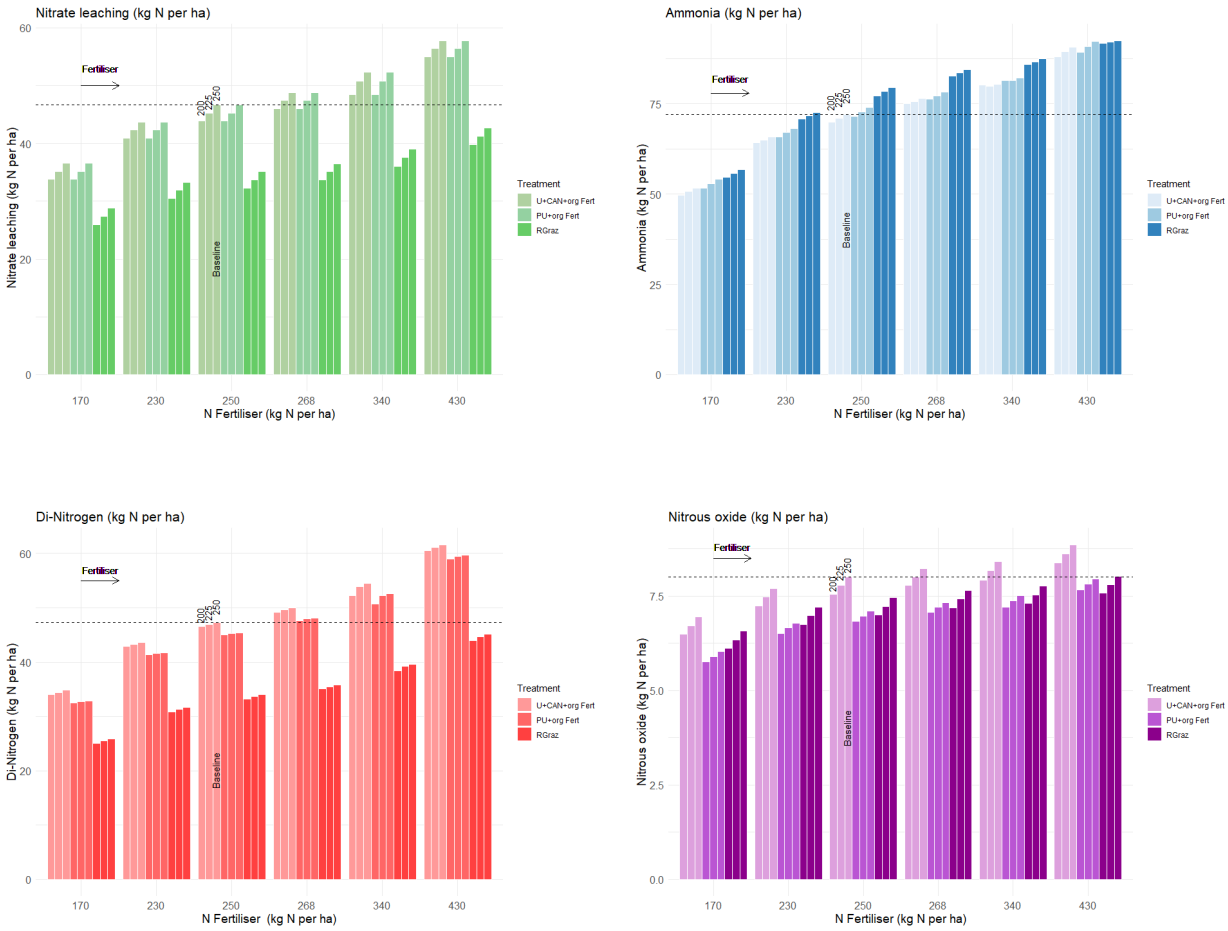


Figure 2- 2 Nitrogen (N) losses (kg N per ha) as nitrate leaching to groundwater, and ammonia, dinitrogen and nitrous oxide losses to air for various scenarios of organic and inorganic fertilizer application rates (kg N ha^{-1}) with three different treatments: 1) Urea + Calcium Ammonia Nitrate (CAN) + Organic Fertilizer; 2) Protected Urea+ Organic Fertilizer and 3) Restrictive grazing (RGraz) for longer time frame.

Abbreviations used in figure: N: Nitrogen, SR: Stocking Rate, ha: hectares, CAN: Calcium Ammonium Nitrate, Fert: fertilizer, Org: organic, RGraz- Restrictive grazing (longer time period), PU- protected Urea.

The N surplus ranged from 139 kg N ha^{-1} to 274 kg N ha^{-1} for Scenario I. Relative to derogation baseline, reducing inorganic fertilizer application rates to 225 kg N ha^{-1} and 200 kg N ha^{-1} decreased N surplus by 7 % and 14 %, respectively. The N surplus decreased with a reduction in stocking rates with organic N loadings of 230 kg N ha^{-1} and 170 kg N ha^{-1} reducing the N surplus by 9 % and 27 %, respectively. Conversely, increasing organic fertilizer application rates to 268, 340 and 430 kg N ha^{-1} resulted in an increase in N surplus by 7 %, 8 %, and 13.5 %, respectively. At VLS rates of 170 kg N ha^{-1}, milk production decreases resulting

in reduced milk N output (Figure 2-1). However, surplus feed is exported from the farm, enhancing the farm's overall N output. Conversely, at VHS rates of 340 kg N ha^{-1}, additional feed is needed within the system, which served as an extra N input. Meanwhile, slurry functions as N output, reflecting its removal from the farming system under these conditions.

Scenario II (shift from Urea + CAN + Organic Fertilizer to using protected urea (urea + Agrotain)) had a minimal to negligible impact on N inputs, N outputs, and N surplus (Table 2-4). However, Scenario III, restrictive grazing, especially for an extended period (September-December) had a noticeable impact on the N budget. For example, compared to derogation baseline, N inputs reduced by 2.15 % with restrictive grazing (longer time frame), while N outputs were barely impacted (Table 2-4). N surplus saw a 2.8 % decrease with restrictive grazing. A reduction in inorganic fertilizer to 225 kg N ha^{-1} and 200 kg N ha^{-1} reduced the N surplus of the baseline by 10 % and 16.8 %, respectively. Similarly, reductions in stocking rates to 230 kg N ha^{-1} and 170 kg N ha^{-1} decreased N surplus by 11.6 % and 29 %, respectively. Conversely, an increase in stocking rate to 268, 340 and 430 kg N ha^{-1} resulted in an increase in N surplus by 4 %, 5.2 %, and 7.7 %, respectively. Restrictive grazing for the shorter time frame showed similar trends to restrictive grazing for longer periods.

Milk yield stayed stable at about 14,914 L ha^{-1} across the three scenarios under different mitigation strategies at the baseline stocking rate of 250 kg N ha^{-1} and an inorganic fertilizer rate of 250 kg N ha^{-1}, showing minimal variation between Scenario I, Scenario II (Protected Urea), and Scenario III (Restrictive Grazing). However, in Scenario I, reducing the stocking rates to 170 kg N ha^{-1} and 230 kg N ha^{-1} decreased milk production by 32.1 % and 9.2 %, respectively. Moreover, reducing the inorganic fertilizer application from 250 kg N ha^{-1} to 200 kg N ha^{-1} and 220 kg N ha^{-1} at baseline stocking rates shows a minor reduction in milk yields. Under restrictive grazing (Scenario III), slurry output increases by 61.7 % at the baseline farm with 250 kg N ha^{-1} inorganic and organic fertilizer application rates, while there is minimal change in slurry output between Scenario I and Scenario II. For the derogation baseline farm, dairy farm revenue (milk + meat) shows minimal changes when applying additional mitigation strategies, such as Scenario II (Protected Urea) and a minimal decline by 0.5 % in Scenario III (Restrictive Grazing). Total revenue for the derogation baseline farm stands at €6116.9 ha^{-1}. Reducing stocking rates from 250 kg N ha^{-1} to 170 kg N ha^{-1} and 230 kg N ha^{-1} leads to a decrease in dairy farm revenue by 32 % and 9.3 %, respectively. In contrast, reducing inorganic fertilizer application rates from 250 kg N ha^{-1} to 220 kg N ha^{-1} and 200 kg N ha^{-1}, at the same stocking rate of 250 kg N ha^{-1}, results in a minor decline in farm revenue (Table 2-5).

Table 2- 5 Milk yield, slurry generated and dairy farm revenue across various scenarios of organic and inorganic fertilizer application rates with three different treatments: 1) Urea + CAN + Organic Fertilizer, 2) Protected Urea+ Organic and 3) Restrictive grazing (longer time period).

Scenario	***Inorg Fert (kg N ha^{-1})***	***Org Fert (kg N ha^{-1})***	***Milk Yield (L ha^{-1})***			***Slurry (m^3)***			***Dairy Farm Revenue (€ ha^{-1})***		
			1	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**
VLSHF	250	170	10119	10119	10083	298	298	482	4151.6	4151.6	4136.9
VLSMF	225	170	10125	10125	10085	298	298	482	4153.9	4153.9	4137.8
VLSLF	200	170	10118	10118	10068	298	298	482	4151.1	4151.1	4130.6
LSHF	250	230	13528	13528	13465	398	398	644	5550.1	5550.1	5524.4
LSMF	225	230	13528	13528	13459	398	398	644	5550.2	5550.2	5521.7
LSLF	200	230	13528	13528	13442	398	398	643	5550.2	5550.2	5514.8
B	250	250	14914	14914	14842	438	438	709	6116.9	6116.9	6087.2
BMF	225	250	14914	14914	14835	438	438	708	6117.1	6117.1	6084.7
BLF	200	250	14914	14914	14811	438	438	707	6117.0	6117.0	6074.5
HSHF	250	268	15905	15905	15825	468	468	758	6525.8	6525.8	6493.2
HSMF	225	268	15922	15922	15836	468	468	757	6532.0	6532.0	6496.6
HSLF	200	268	15952	15952	15858	476	476	760	6542.9	6542.9	6504.2
VHSHF	250	340	20061	20061	19944	725	725	1015	8218.0	8218.0	8170.2
VHSMF	225	340	20134	20134	20008	733	733	1018	8243.9	8243.9	8192.2
VHSLF	200	340	20166	20166	20030	765	765	1028	8254.8	8254.8	8199.3
ESHF	250	430	25253	25253	25080	1075	1075	1381	10332.7	10332.7	10261.9
ESMF	225	430	25328	25328	25140	1091	1091	1397	10360.1	10360.1	10283.3
ESLF	200	430	25396	25396	25190	1113	1113	1417	10383.6	10383.6	10299.6

Abbreviation used in Table: N: Nitrogen, ha: hectares, CAN: Calcium Ammonium Nitrate, Fert: fertilizer, Inorg: Inorganic, Org: organic, E=extreme, VH=very high, VL=very low, S=stocking rate, LF=low fertilizer rate; MF- medium fertilizer rate, HF= high fertilizer rate, B=Baseline (Derogation).

2.4.2 Nitrate Leaching to Groundwater

The average farm-gate level NO_3^- leaching losses for the derogation baseline were 46.6 kg N ha^{-1}, which is greater than the target of 30 kg N ha^{-1} for good water quality outcomes as set out in New Zealand (Table 2-6). Reduction in inorganic and organic fertilizer application rates, along with additional treatments such as restrictive grazing and use of 100 % protected urea, had a substantial impact on leaching losses. Under Scenario I, major changes were observed by varying inorganic and organic fertilizer application rates, resulting in NO_3^- losses ranging from 33.8 kg N ha^{-1} to 57.7 kg N ha^{-1} from VLSLF to ESHF (Table 2-5). Reducing inorganic fertilizer application rates to 225 kg N ha^{-1} and 200 kg N ha^{-1} from derogation baseline decreased NO_3^- leaching losses by 3 % and 6 %, respectively (Figure 2). Furthermore, varying stocking rates also influenced leaching losses. By reducing organic fertilizer application rates to 230 kg N ha^{-1} and 170 kg N ha^{-1}, leaching losses decreased by 6.4 % and 21.5 %, respectively. Under Scenario I, the consequences of increasing organic fertilizer application rates above derogation baseline to 268, 340 and 430 kg N ha^{-1} are clear. This led to substantial increases in leaching losses by 4.6 %, 12 %, and 24 %, respectively.

Table 2- 6 Nitrogen losses across various scenarios of organic and inorganic fertilizer application rates with three different treatments: 1) Urea + CAN + Organic Fertilizer, 2) Protected Urea+ Organic and 3) Restrictive grazing (longer time frame).

Scenario	***Inorg Fert***	***Org Fert***	***NO_3^-***			***N_2O***			***NH_3***		
	kg N ha^{-1}										
			1	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**
VLSHF	250	170	36.6	36.6	28.8	7.0	6.0	6.6	51.7	54.2	56.7
VLSMF	225	170	35.2	35.2	27.4	6.7	5.9	6.4	50.8	52.9	55.8
VLSLF	200	170	33.8	33.8	26.0	6.5	5.8	6.1	49.7	51.7	54.7
LSHF	250	230	43.7	43.7	33.2	7.7	6.8	7.2	65.9	68.2	72.7
LSMF	225	230	42.4	42.4	31.9	7.5	6.7	7.0	65.0	67.1	71.8
LSLF	200	230	41.0	41.0	30.5	7.2	6.5	6.7	64.2	65.8	70.9
B	250	250	46.6	46.6	35.1	8.0	7.1	7.5	72.0	74.0	79.5
BMF	225	250	45.3	45.3	33.7	7.8	7.0	7.2	71.0	72.8	78.4
BLF	200	250	43.9	43.9	32.3	7.6	6.8	7.0	69.9	71.5	77.2
HSHF	250	268	48.8	48.8	36.5	8.2	7.3	7.6	76.5	78.3	84.5
HSMF	225	268	47.5	47.5	35.1	8.0	7.2	7.4	75.6	77.1	83.5
HSLF	200	268	46.0	46.0	33.7	7.8	7.1	7.2	75.0	76.4	82.7
VHSHF	250	340	52.3	52.3	39.0	8.4	7.5	7.8	80.5	82.2	87.4
VHSMF	225	340	50.8	50.8	37.6	8.2	7.4	7.5	80.0	81.4	86.6
VHSLF	200	340	48.5	48.5	36.0	7.9	7.2	7.3	80.3	81.6	86.0
ESHF	250	430	57.7	57.7	42.6	8.9	8.0	8.0	90.8	92.4	92.5
ESMF	225	430	56.4	56.4	41.3	8.6	7.8	7.8	89.5	90.9	92.1
ESLF	200	430	55.0	55.0	39.8	8.4	7.7	7.6	88.0	89.3	91.8

Abbreviation used in Table: N: Nitrogen, ha: hectares, CAN: Calcium Ammonium Nitrate, Fert: fertilizer, Inorg: Inorganic, Org: organic, E=extreme, VH=very high, VL=very low, S=stocking rate, LF=low fertilizer rate; MF-medium fertilizer rate, HF= high fertilizer rate, B=Baseline (Derogation)

Restricting grazing (Scenario III) showed a major reduction in leaching losses of 24.7 % compared to the derogation baseline farm for the longer time frame (6 months) and 14.5 % reduction with the shorter time frame (5 months). Within the extended restrictive grazing scenario, reducing inorganic fertilizer application rates by 225 kg N ha^{-1} and 200 kg N ha^{-1} resulted in further decrease in NO_3^- losses by 27.7 % and 30.7 %, respectively. Reducing stocking rates or organic fertilizer application rates to 230 kg N ha^{-1} and 170 kg N ha^{-1} showed a 28.8 % and 38.2 % reduction in NO_3^- losses, respectively (Table 2-6). For very low stocking rates at 170 kg N ha^{-1} and low inorganic fertilizer application rates at 200 kg N ha^{-1}, restricted grazing decreased leaching losses by 44.3 %.

Increasing stocking rates or organic fertilizer application rates within Scenario III above derogation baseline to 268, 340 and 430 kg N ha^{-1} resulted in an overall decline in leaching

losses by 22 %, 16.3 % and 8.6 %, respectively. Shorter duration of restrictive grazing yielded comparable trends, however with lesser magnitudes of impact. For instance, reducing inorganic fertilizer rates to 225 kg N ha^{-1} and 200 kg N ha^{-1} decreased NO_3^- losses by 17.5 % and 20.5 %, respectively. Similarly, lowering stocking rates to 230 kg N ha^{-1} and 170 kg N ha^{-1} reduced leaching losses by 19.5 % and 31.4 %, respectively. Increasing stocking rates within the shorter time period of restrictive grazing, exhibited a similar reduction in leaching as the longer period but with a smaller magnitude.

2.4.3 Gaseous Emissions

The derogation baseline average gaseous emissions, including NH_3, N_2O, and N_2 emissions, were 127.3 kg N ha^{-1}. Specifically, NH_3, N_2O, and N_2 emissions were 72, 8, and 47.3 kg N ha^{-1}, respectively (Table 2-6). Under Scenario I (Urea + CAN + Organic Fertilizer), substantial changes were observed with varying inorganic and organic fertilizer application rates, resulting in NH_3 emissions ranging between 49.7 kg N ha^{-1} to 90.7 kg N ha^{-1}, N_2O emissions varied from 6.5 kg N ha^{-1} to 8.9 kg N ha^{-1}, and N_2 emissions ranging between 34 kg N ha^{-1} to 61.5 kg N ha^{-1} (Table 2-6). Reducing inorganic fertilizer application rates for a derogation baseline farm to 225 kg N ha^{-1} and 200 kg N ha^{-1} decreased NH_3 losses by 1.4 % and 3 %, N_2O losses by 2.8 % to 5.6 %, and N_2 losses by 0.8 % to 1.7 %, respectively. Furthermore, decreasing organic fertilizer application rates (Figure 2-3) to 230 kg N ha^{-1} and 170 kg N ha^{-1} decreased NH_3 losses by 8.5 % and 28 %, N_2O emissions by 3.8 % and 13 %, and N_2 losses by 7.8 % and 26.4 %, respectively. Conversely, increasing organic fertilizer application rates to 268, 340 and 430 kg N ha^{-1} increased NH_3 losses by 6.3 %, 11.8 %, and 26 %, N_2O losses by 2.8 %, 5 %, and 10.6 %, and N_2 losses by 5.6 %, 15 %, and 30 %, respectively.

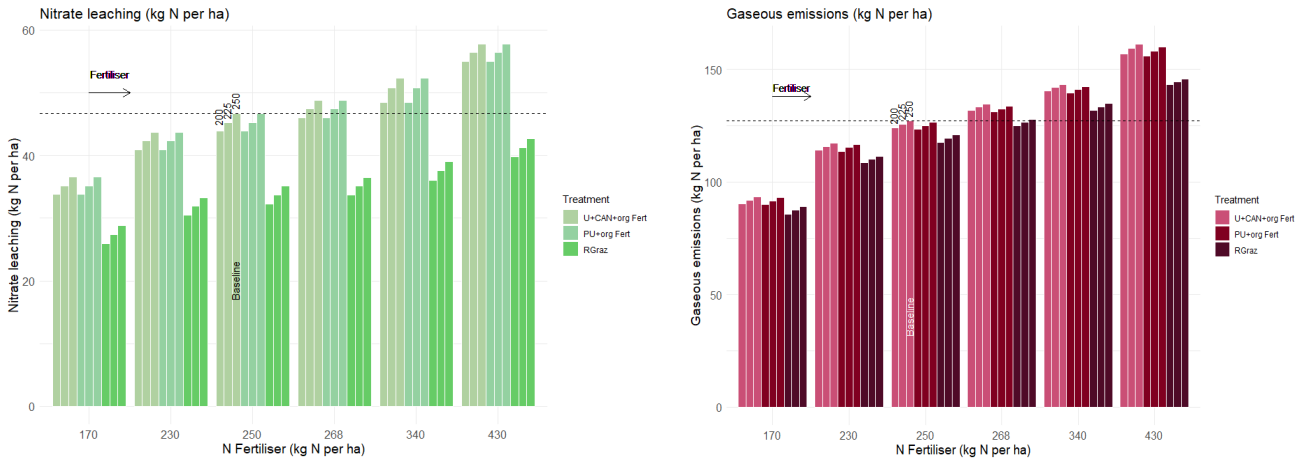


Figure 2- 3 N leaching and Gaseous emission losses across various scenarios of organic and inorganic fertilizer application rates with three different treatments of 1) Urea + CAN + Organic Fertilizer, 2) Protected Urea+ Organic fertilizer, 3) Restrictive grazing (RGraz) for longer time period. Abbreviations used: N: Nitrogen, SR: Stocking Rate, ha: hectares, CAN: Calcium Ammonium Nitrate, Fert: fertilizer, Org: organic, RGraz- Restrictive grazing, PU- Protected Urea.

For Scenario II, the use of protected urea had a negative impact on the derogation baseline increasing NH_3 emissions by 2.8 %. However, N_2O and N_2 emissions saw declines of 11.3 % and 4 %, respectively in this scenario (Table 2-5). Switching to protected urea and reducing inorganic fertilizer to 225 kg N ha^{-1} for the derogation baseline increased NH_3 loss by 1 %, with a further decrease to 200 kg N ha^{-1} resulting in NH_3 losses declining marginally by 0.6 %. N_2O losses and N_2 losses also showed declines from the baseline scenario by 13 % and 14.7 %, and 4.4 % and 4.8 % at inorganic fertilizer application rates of 225 kg N ha^{-1} and 200 kg N ha^{-1}, respectively. Coupling protected urea with lower organic fertilizer application rates (230 kg N ha^{-1} and 170 kg N ha^{-1}) reduced NH_3, N_2O, and N_2 emissions by 5 % to 25 %, 15 % to 24.6 %, and 11.8 % to 30.4 %, respectively. However, as organic fertilizer application rates increased, NH_3 and N_2 emissions increased, while N_2O emissions decreased slightly.

There were trade-offs caused by restrictive grazing for both long and short time frames. For example, long and short time frames considerably increased NH_3 emissions by 10.4 % and 10.8 %, respectively, compared to the derogation baseline. Restrictive grazing for the longer time period reduced N_2O emissions by 6.8 % (Tables 2-6, Table 2-7), and N_2 emissions by 28 %, whereas restrictive grazing for shorter time frames showed a comparatively smaller decline in emissions. Pairing restrictive grazing for the longer time period with reduced inorganic fertilizer application rates of 225 kg N ha^{-1} and 200 kg N ha^{-1}increased NH_3 emissions relative

to derogation baseline by 9 % and 7.2 %, respectively (Figure 2-3). Conversely, N_2O and N_2 losses exhibited a tendency to decrease by 9.7 % and 12.5 %, and 28.8 % and 29.7 %, respectively, for 225 kg N ha^{-1} and 200 kg N ha^{-1} inorganic fertilizer rates compared to baseline. The impact of reducing stocking rates on emissions varied: an 8 % reduction in organic fertilizer application rate (230 kg N ha^{-1}) showed a marginal increase in NH_3 emissions with respect to derogation baseline (Scenario I), while a 32 % reduction (170 kg N ha^{-1}) resulted in a 21 % decrease in NH_3 emissions. Conversely, increasing organic fertilizer application rate led to an upward trend in NH_3 losses. Both N_2O and N_2 emissions generally decreased with decreasing stocking rates. For VLS rate at 170 kg N ha-1 and LF rates at 200 kg N ha^{-1}, restricted grazing decreased NH_3 emissions by 24 %, N_2 emissions by 47 % and N_2O emissions by 23.6 % relative to baseline. Reduction in organic fertilizer application rates to 230 kg N ha^{-1} and 170 kg N ha^{-1} led to a reduction in N_2 emissions by 33 % and 45 %, respectively. Similarly, increasing stocking rates while applying restrictive grazing in comparison to derogation baseline, resulted in a reduction in N_2 by 24 %, 16 %, and 4.5 % for organic fertilizer application rates of 268, 340 and 430 kg N ha^{-1}, respectively.

Table 2- 7: Nitrate leaching and Gaseous emissions across various scenarios of organic and inorganic fertilizer application rates with three different treatments: 1) Urea + CAN + Organic Fertilizer, 2) Protected Urea+ Organic and 3) Restrictive grazing (longer time period).

Scenario	*Inorg Fert*	*Org Fert*	*Nitrate*			*Gaseous emissions (NH_3, N_2O, N_2)*		
	kg N ha^{-1}							
			1	**2**	**3**	**1**	**2**	**3**
VLSHF	250	170	36.6	36.6	28.8	93.5	93.1	89.2
VLSMF	225	170	35.2	35.2	27.4	91.9	91.5	87.6
VLSLF	200	170	33.8	33.8	26.0	90.2	89.9	85.8
LSHF	250	230	43.7	43.7	33.2	117.2	116.7	111.5
LSMF	225	230	42.4	42.4	31.9	115.8	115.3	110.1
LSLF	200	230	41.0	41.0	30.5	114.3	113.7	108.5
B	250	250	46.6	46.6	35.1	127.3	126.5	121.0
BMF	225	250	45.3	45.3	33.7	125.7	125.0	119.4
BLF	200	250	43.9	43.8	32.3	124.0	123.4	117.5
HSHF	250	268	48.8	48.8	36.5	134.7	133.7	128.0
HSMF	225	268	47.5	47.5	35.1	133.3	132.4	126.5
HSLF	200	268	46.0	46.0	33.7	131.9	131.1	125.0
VHSHF	250	340	52.3	52.3	39.0	143.4	142.4	134.9
VHSMF	225	340	50.8	50.8	37.6	142.1	141.0	133.4
VHSLF	200	340	48.5	48.5	36.0	140.5	139.5	131.8
ESHF	250	430	57.7	57.7	42.6	161.2	160.1	145.6
ESMF	225	430	56.4	56.4	41.3	159.3	158.2	144.5
ESLF	200	430	55.0	55.0	39.8	156.9	156.0	143.3

Abbreviation used in Table: N: Nitrogen, ha: hectares, CAN: Calcium Ammonium Nitrate, Fert: fertilizer, Inorg: Inorganic, Org: organic, E=extreme, VH=very high, VL=very low, S=stocking rate, LF=low fertilizer rate; MF- medium fertilizer rate, HF= high fertilizer rate, B=Baseline (Derogation).

2.5 Discussions

2.5.1 N Surplus

For derogation baseline, N inputs are predominantly high, and include feed, artificial fertilizer, atmospheric deposition, replacement animals and slurry import. Conversely, N outputs, primarily milk and meat, yield positive results but contribute to an increase in N surplus. The derogation baseline farm exhibits substantial N input requirements, with stocking rates at 250 kg N ha^{-1} and inorganic fertilizer application rates at 250 kg N ha^{-1}. Despite this, animals still require additional concentrate feeds to meet dietary demands. Concentrate feeds are imported onto the farm and are primarily driven by fertilizer inputs. Reducing N surplus is important to improve the efficiency of the farming system and to reduce emissions to both water and air (Quemada et al., 2020). Farm N surplus can be decreased through reducing inorganic fertilizer application rates and stocking rates (Nielsen & Kristensen 2005; Nevens et al., 2006). Reducing stocking rates, shifts feed production to become an output of the system, thereby decreasing N inputs. Conversely, increasing stocking rates intensifies feed demand, necessitating feed imports alongside increased slurry production (McCarthy et al., 2015; Børsting et al., 2003; Fenger et al., 2023). Thus, reducing stocking rates proves advantageous in reducing N surplus while fostering surplus grass growth suitable for export, however it does reduce milk output and reduces overall profitability. Moreover, lowering inorganic fertilizer application rates within the derogation baseline farm effectively reduces N inputs without compromising milk and meat outputs, thereby maintaining lower N surplus levels.

Transitioning from CAN to protected urea does not alter the N dynamics within the system, as evidenced by consistent trends observed in N inputs, outputs, and surplus. Our findings indicate that the shift to protected urea, while a different N source, does not considerably affect the farm's N balance compared to Scenario I. Specifically, there are no discernible differences in the impact of changing inorganic and organic fertilizer application rates on the overall N balance of the farm with the application of protected urea. These results highlight the stability and consistency of the overall farm's N management practices despite changes in fertilizer types.

Applying restrictive grazing impacted the N surplus primarily due to a small decline in N inputs as the €riN-MDSM model optimized the shift from applying inorganic fertilizers to slurry collected from housed animals. N outputs, however, did not change across many of the scenarios. The optimization within €riN-MDSM reduced the quantity of inorganic fertilizer

input, but the N content entering the system remained unaffected because the N content in slurry during restrictive grazing was higher than in other scenarios. The major impact on N surplus within restrictive grazing was due to the reduction in stocking rate and inorganic fertilizer application. Reducing the number of animals on the farm severely contributed to a reduction in N surplus and farm productivity but this reduction was not further enhanced by applying restrictive grazing. Even though restrictive grazing for the autumn months (shorter time frame) showed similar trends as the extended autumn period (6 months), there are substantial advantages to the longer time period for restrictive grazing as discussed below in terms of NO_3^- losses. Doole (2015), highlights that reducing leaching without investing in restrictive grazing measures is costly even though it helps reach water quality standards. But it requires broad farm de-intensification, including cuts in N fertilizer, stocking rates, and milk production due to the link between milk production and N leaching.

2.5.2 N losses to water and air

In the derogation baseline scenario, characterized by high leaching losses and NH_3 emissions due to the application of 250 kg N ha^{-1} of both inorganic and organic fertilizers, reducing these fertilizer application rates and stocking rates proves advantageous to reduce N losses. Following the abolition of the milk quota, the dairy industry experienced a substantial expansion, leading to the Teagasc report on N management strategies within grass-based dairy systems (Teagasc, 2021). This report established a derogation baseline farm with a high stocking rate of 271 kg N ha^{-1} and an inorganic fertilizer application rate of 250 kg N ha^{-1}, resulting in high leaching losses ranging from 45 to 51 kg N ha^{-1} yr^{-1} for varying inorganic fertilizer application rates. These values exceed the median NO_3^- leaching losses of 30 kg N ha-1 reported for similar environments (Norris et al., 2019). This target can help meet water quality targets in the region with achievable and realistic good practice objective for NO_3^- leaching losses below 30 kg N ha^{-1} yr^{-1} (Beukes et al., 2012). Necessitating and even further reduction in our derogation baseline farm which shows leaching losses well above the 30 kg N ha^{-1} threshold for an average dairy farm.

Stocking rates play a crucial role in leaching losses, with lower stocking rates correlating to less leaching losses (Roche et al., 2016). Our research supported this finding. For example, €riN-MDSM showed reducing stocking rate for the derogation baseline by 32 % decreased urine and dung N deposits on pastureland which reduced leaching losses by more than 20 %. The N load originating from urine patches (i.e., N per unit area affected by urine

deposition) served as the primary contributor to N loss into water bodies (Di & Cameron 2002; Hutchings et al., 2006)). Similarly, reducing inorganic fertilizer rates for this dairy system by 20 % reduced leaching losses by 5 % to 43.8 kg N ha^{-1}, which exceeds the 30 kg N ha^{-1} threshold for water quality targets. Beukes et al. (2012) found that farm-gate N surplus and N leaching increased with higher stocking rates and external N input. The higher N leaching in the study conducted by Beukes et al. (2017) was attributed to a greater influx and flow of N through the system, rather than solely to the higher stocking rate itself.

The reduction in stocking rate from 250 kg N ha^{-1} to 170 kg N ha^{-1} lead to a major decrease in NH_3 emissions from 71 kg N ha^{-1} to 51.7 kg N ha^{-1} for inorganic fertilizer application rate of 250 kg N ha^{-1}. As the number of livestock units decreases, the production of urine and dung declines, resulting in reduced NH_3 emissions into the air. Conversely, increasing stocking rates has pronounced impacts not only on NH_3 emissions but also on overall gaseous emissions, including N_2O and N_2 emissions. Additionally, while the impact of inorganic fertilizer application on reducing gaseous emissions is relatively small, it still contributes positively to emission reduction efforts.

The transition from CAN to protected urea as an inorganic fertilizer had no noticeable impact on leaching losses. Protected urea (urea + Agrotain), which stabilizes N by converting it to the more stable NH_3 form in the soil, mitigates leaching losses, particularly during periods of heavy or prolonged rainfall post-fertilizer application. This underscores the importance of protected urea in reducing leaching losses under adverse weather conditions. Protected urea buffers against leaching losses during high rainfall events (Di & Cameron 2002). However, the €riN-MDSM model efficiently applies this fertilizer during these vulnerable periods, which does not reflect reality. Consequently, the potential reduction in leaching losses is not quantified in this model but does exist and is causing the model to underestimate leaching losses in some years, while overestimating them in others. However, low or high inorganic fertilizer application rates with the use of protected urea do not show any change in leaching losses compared to Scenario I. This lack of difference is due to the nature of the €riN-MDSM setup, which does not account for weather conditions that are crucial for accurate leaching loss estimates. Conversely, the use of protected urea resulted in increased NH_3 emissions at high stocking rates. At lower stocking rates and lower inorganic fertilizer rates, a reduction in NH_3 losses was observed, although at a lower magnitude compared to Scenario I. This highlights the importance of focusing on farm scenarios below the derogation baseline farm.

While protected urea provides readily available N in the form of NH_3 for plant uptake and efficient grass growth, it can lead to higher NH_3 emissions when it replaces CAN (Forrestal et al. 2016). In Scenario II, the use of protected urea combined with reduced organic and inorganic fertilizer applications from the derogation baseline farm showed positive environmental implications in terms of both reduced leaching and gaseous emissions to the derogation baseline. This aligns well with studies conducted by Forrestal et al. (2016), where NH_3 losses from protected urea were reduced but still higher than with CAN alone.

Nitrous oxide emissions also follow a similar pattern, with Scenario II (urea + Agrotain) showing lower N_2O emissions compared to CAN + urea in Scenario I, which aligns with the findings of Forrestal et al. (2016). Regarding leaching losses, protected urea is known to reduce leaching during vulnerable periods of high rainfall. In our case, €riN-MDSM efficiently distributes fertilizer during non-vulnerable periods, resulting in leaching losses similar to those in the CAN + urea (Scenario I) scenario.

Restrictive grazing (Scenario III) effectively reduces leaching losses by decreasing the time animals spend outdoors on pastures, thereby reducing urine and dung deposition and resulting in a major decline in NO_3^- losses through leaching meeting < 30 kg N ha^{-1} threshold for water quality targets. Applying restrictive grazing over a six-month period, combined with a reduction in stocking rate and inorganic fertilizer application, successfully resulted in achieving median NO_3^- leaching losses of under 30 kg N ha^{-1}, meeting water quality goals. The manure collected from practices like restrictive grazing can be substituted for inorganic fertilizer which will increase NUE while reducing N surplus (Tong et al., 2022; Webb et al., 2013). However, with animals spending the majority of their time indoors, urine and dung patches on the soil redirected to indoor collection for slurry. This accumulation of slurry led to trade-offs between NO_3^- leaching and NH_3 volatilization. NH_3 emissions on the other hand still declined with the use of restricted grazing while simultaneously dropping stocking rates and fertilizer application. But the general trend of NH_3 emissions increased with the application of restrictive grazing at the same level of inorganic and organic fertilizer application as larger quantities of slurry were accumulated. Although this may lead to increased NH_3 losses through volatilization during slurry handling and storage, proper infrastructure and management practices can mitigate these emissions (Owusu-Twum et al., 2024).

The primary consequence of reduced grazing duration is increased manure accumulation, leading to higher NH_3 and methane (CH_4) emissions (with NH_3 quantified)

(Herzog et al., 2018). Reduced field time may also raise animal welfare concerns and increase labor demands for feeding and manure management (DeVries et al., 2015). However, since farmers already bring animals in for morning and afternoon milking, keeping them in the shed afterward does not significantly alter workload. Additional slurry storage may be required, depending on regulatory constraints. Finally, DeVries et al., (2015) and Herzog et al., (2018) highlighted that extended time on concrete surfaces might also increase the risk of animal injuries. Combining restricted grazing with strategic manure management, better housing space, appropriate flooring and feed adjustments can enhance overall sustainability without negatively impacting productivity.

A noteworthy observation from this scenario involving restrictive grazing is the substantial contribution of the extended autumn strategy, spanning from September to February, towards both reducing N losses and enhancing N management efficiency. While a shorter timeframe during the autumn months (October-February) also displayed a positive impact on leaching losses, the extended autumn strategy over six months (September-February) proved to be particularly effective. This approach, when combined with reductions in stocking rates and inorganic fertilizer application rates, demonstrated notable benefits in terms of N management and mitigation of losses. Restrictive grazing practices accounted for 60 % of the leaching reduction and decreasing N inputs accounted for the remaining 40 % reduction in N leaching losses (Beukes et al., 2017). This implies that farmers encountering restrictions on nutrient loss have access to technologies to address these limitations.

Ultimately it will be the climate alongside the cost of additional feed and slurry storage infrastructure needed (Hooda et al., 2000) that decides if restricted grazing becomes commonplace on dairy farms in Ireland. Besides climate three other factors will need to be considered: dietary composition, length of standoff, and duration of standoff. Both Dalley et al. (2020) and Waghorn et al. (2018) documented decreases in urinary N excretion when dairy cows were provided diets comprising 23-60 % fodder beet during late lactation. In Ireland, several aspects of restricted duration grazing need to be investigated on dairy farms including:

- Allowing two 4-hour grazing periods after each milking, with a diet mainly consisting of grass, and a 16-hour period of restrictive grazing.
- Providing 8 hours of daytime grazing access followed by 4 hours in the evening, with a primary focus on grass in the diet, and a 12-hour period of restrictive grazing.

- Offering only 8 hours of daytime grazing access, resulting in a diet comprising half pasture and half silage, supplemented with additional concentrates, and a 12-hour period of restrictive grazing.

The effectiveness of these options may vary depending on the duration in November, October, and September. Comprehensive analysis of these factors will provide valuable insights into the feasibility and effectiveness of implementing restricted grazing practices in Ireland. When combined with additional measures, this approach can help achieve the target of below 30 kg N ha^{-1} leaching losses, meeting water quality standards.

Another very important farm management practice for reducing leaching losses apart from focusing on low N diet for dairy cows (Robertson et al., 2023) is using alternate plant species Beukes et al. (2017; 2018). Previous studies have shown that mixed swards are shown to be highly favorable as well in terms of reducing leaching losses while still maintaining farm productivity (Beukes et al., 2017; Beukes et al., 2018; Pinxterhuis & Edwards 2018). Adopting diverse pastures, including plantain as part of the sward composition, could potentially reduce N leaching by 40 % while maintaining milk production, compared to pastures composed of perennial ryegrass and white clover (Romera et al., 2017, Forrestal et al., 2025). In Ireland, a five-soil type study over two years showed that ribwort plantain inclusion reduced NO_3^- leaching from grass-clover swards (Egan et al., 2025). The inclusion of plantain at 30 % reduced NO_3^- load in free-draining soils (as in this study) by 7–59 kg NO_3-N ha^{-1} (or by 32–74 %, mean 56 %) in year one and by 12–54 kg NO_3-N ha^{-1} (or by 93–99 %, mean 96 %) in year two.

Studies by Beukes et al. (2024), showed that implementing a combination of highest-rated N mitigation strategies considerably reducing N leaching at the farm level. The selection of stacked options of these N mitigation strategies such as: 1) decreasing N fertilizer application , 2) installing infrastructure off the paddock, 3) reusing N by cultivating maize silage in a specified area on the farm and utilizing effluent as a fertilizer source, followed by planting a catch-crop, 4) supplementing dietary salt to reduce urinary N concentration, and 5) employing a nitrification inhibitor vary based on the desired leaching reduction goal and the extent to which farmers are willing to compromise profitability to ensure the future resilience of their businesses. The animal-based mitigation methods that proved most effective in reducing leaching losses were those that aimed to decrease N excretion during grazing. This was

achieved by either reducing the number of animals per hectare, with or without a decrease in herd size, or by reducing the duration of grazing.

Outcomes from the modelled simulations in this study (derogation baseline and others) align with internationally published findings. For example, Christensen et al. (2019b) demonstrated that restrictive grazing with reduced fertilizer application (approximately 212 kg N ha^{-1} yr^{-1}) in New Zealand reduced N leaching losses by 52 % compared to standard grazing, similar to De Klein et al. (2006), who reported a 41 % reduction with restrictive grazing. Additionally, Christensen et al. (2019a) identified autumn grazing as a critical period for NO_3^- leaching, emphasizing that minimizing urine deposition (with losses to shallow groundwater) during this time yielded the most substantial reductions—consistent with the present study's focus on restricting grazing during the wettest months when grass growth has stopped and soil drainage occurs at highest rates. Similarly, the Canterbury Land and Water Regional Plan mandated a 30 % reduction in N losses from baseline levels by 2022 (Pinxterhuis & Edwards, 2017). To achieve this, dairy farms in the Selwyn catchment implemented strategies such as reducing stocking rates, optimizing fertilizer and slurry applications, and incorporating low-N supplements to limit N deposition. Comparable reductions in N losses were observed in this study, further supporting the effectiveness of targeted management strategies in mitigating N losses while maintaining farm profitability.

In the present study, we observed that the current derogation baseline for intensive dairy farming is set at high N values, necessitating an urgent shift towards lowering N inputs. This should be coupled with mitigation strategies such reduced stocking rates and restrictive grazing to effectively reduce leaching losses. This finding is consistent with research conducted in Wales and England by Cardenas et al. (2011), where it was determined that the most effective measures for reducing leaching losses in dairy systems generally included reductions in stocking rates, grazing duration, improvements in the timing of fertilizer application, and consideration of manure N. In light of our study findings, further investigation into a stacked mitigation approach involving restrictive grazing over longer time periods, combined with the use of N inhibitors such as protected urea, would be highly beneficial. Future agricultural systems should incorporate a combination of these practices and technologies, particularly during periods of high rainfall. Additionally, future system trials should monitor nutrient flows and losses while including economic analysis to guide progress toward improved water and air quality outcomes.

2.5.3 Limitations of the €riN-MDSM model approach and future work

Some shortcomings of the €riN-MDSM include its insensitivity to the timing of N application, whether in the form of fertilizer or slurry. Additionally, the model's estimation of N leaching from urine deposition is based on a single experiment by Stark et al. (2007). It could be improved through the incorporation of varying urine N content and more refinement of urine and fertilizer timing on leaching, NH_3, N_2O and N_2 emissions (Selbie et al., 2015). Furthermore, N losses are not directly influenced by weather factors in the model. The model calculates N losses independently of weather conditions, which is particularly evident with leaching losses when using protected urea; especially crucial during vulnerable periods of high rainfall. The switch from CAN to protected urea had no considerable effect on leaching losses. Protected urea stabilizes N and reduces leaching, especially after heavy rainfall. However, the €riN-MDSM model does not fully capture this impact, leading to inaccuracies in estimating leaching losses in some years. Under derogation regulations, a baseline farm located in the south and east of Ireland is required to maintain 16-18 weeks of slurry storage, whereas farms located in the west and north have a storage period of about 20+ weeks. However, under a restrictive grazing scenario, the additional slurry generated would require mitigation measures to reduce NH_3 emissions.

Future research efforts should focus on establishing long-term systems trials on dairy farms with varying rainfall, grass species mixtures (e.g. incorporating clover) and rainfall to evaluate the effected of multiple mitigation measures to reduce N losses. Additionally, outcomes from the €riN-MDSM should be compared with mechanistic models and further developed to incorporate clover (and N fixation), along with other grass species mixtures, (Cummins et al. 2021). The role of legacy and accumulated N and its release in soil drainage in time and space should also be considered (Ascott et al., 2021). Furthermore, the outputs of the €riN-MDSM model should be integrated with groundwater models that consider natural attenuation (e.g. Fenton et al., 2009; Jahangir et al., 2012) to assess water quality outcomes in associated water bodies, aligning with the reporting requirements of the EU WFD. Additionally, it would be highly valuable to run these scenarios in the €riN-MDSM model using data from commercial dairy farms, such as those in the Teagasc National Farm Survey, to understand the implications of adopting different N mitigation strategies.

This study demonstrates that restrictive grazing, when paired with reduced stocking rates and lower inorganic fertilizer application, can maintain farm productivity while majorly

reducing N losses through leaching and emissions. Although restrictive grazing may lead to increased NH_3 losses and costs due to larger slurry quantities, the former can be effectively managed through appropriate mitigation strategies and improved infrastructure. By integrating targeted mitigation approaches with efficient N management practices, it is possible to achieve water quality targets without compromising production levels. On that basis, future system trials should monitor flows and losses from the implementation of such mitigations in addition to the economic impacts to chart a way forward for better water and air outcomes while maintaining the profitability of Irish dairy farms.

2.6 Conclusions

Nutrient pollution from agriculture necessitates targeted farm management strategies. This study highlights restrictive grazing as an effective measure to reduce NO_3^- leaching, though it increases NH_3 and CH_4 emissions, manure accumulation, and potential animal welfare concerns. These challenges can be mitigated through strategic manure management, improved housing, and optimized feeding. A shift from CAN + urea to urea with Agrotain further reduces N losses, while protected urea proves most effective at lower stocking and fertilizer application rates. Managing the trade-off between NH_3 volatilization and NO_3^- leaching is critical, as increased slurry storage is required. The €riN-MDSM model's insensitivity to N application timing underscores the need for refinement. Stocking rates severely impact N losses, raising sustainability concerns under current derogation policies. Future research should integrate restrictive grazing, protected urea, and economic assessments to ensure environmentally and economically viable mitigation strategies for Irish dairy farms.

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Chapter 3

Multi-decadal Relationships between Nitrogen Surpluses, Soil Legacy Nitrogen Accumulation, and Groundwater Nitrate in Intensive Dairy Systems of Southern Ireland

This chapter is a mirror of the following article in preparation.

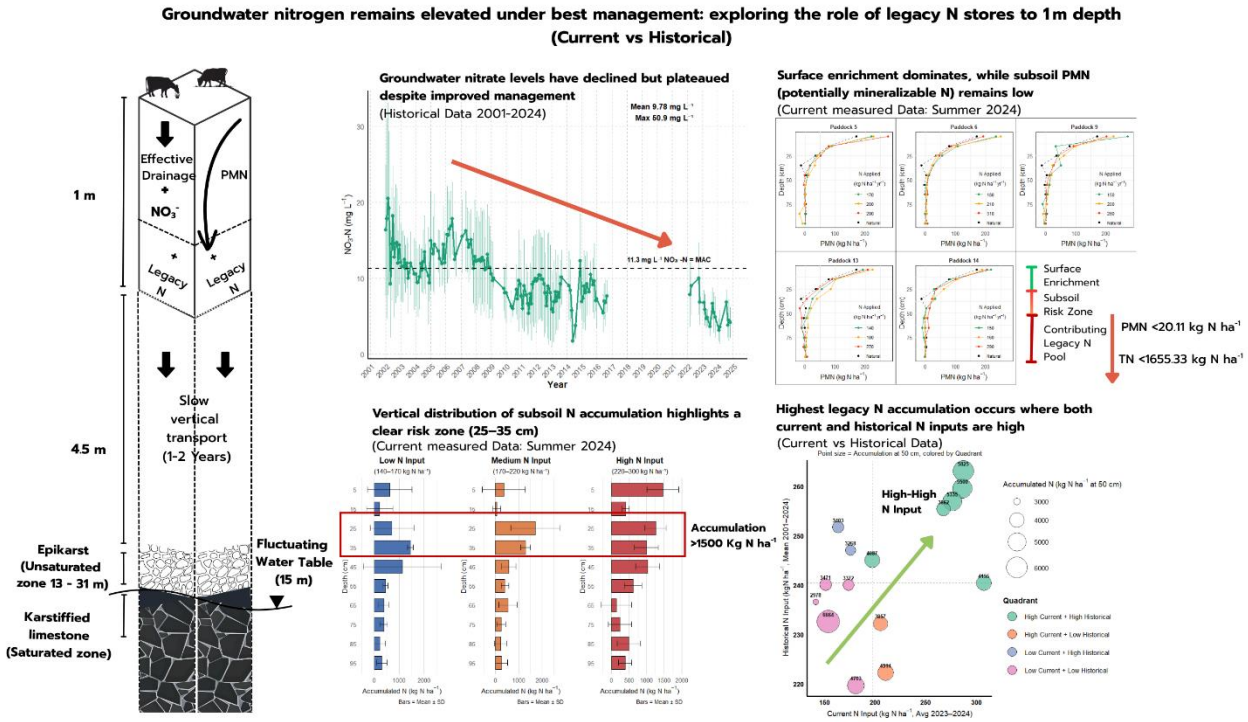
Lakhanpal, G., Fenton, O., Basu, N.B., Horan, B., Richards, K.G., Cardenas-Alvarez, G., Lopez-Sangil, L. and Condon, T., Multi-Decadal Relationships between Nitrogen Surpluses, Soil Legacy Accumulation, and Groundwater Nitrate in Intensive Dairy Systems in Southern Ireland.

3.1 Overview

On intensive dairy farms with free-draining soils, high nitrogen (N) surpluses and limited denitrification commonly lead to elevated nitrate (NO_3^-) losses to groundwater. This 24-year study on a farm in southern Ireland examined the long-term impacts of N management on subsurface N accumulation and groundwater quality. Persistent groundwater NO_3^- concentrations, despite recent input reductions, prompted an integrated analysis of historical (2001–2024) farm activity data and 2024 soil profile measurements of total N (TN) and potentially mineralizable N (PMN) to 1 m depth. While NO_3^- flux declined from over 70 to <30 kg N ha^{-1} by 2024 and now remain below the EU maximum admissible concentration of 11.3 mg L^{-1}, they have plateaued in recent years, suggesting a continued influence of legacy N. Soil cores revealed strong N stratification, with peak TN (5,223 kg N ha^{-1}) and PMN (>160 kg N ha^{-1}) in surface layers. Subsoil accumulation was most pronounced between 25–35 cm in high-input paddocks, where accumulation N to 50 cm reached >5000 kg N ha^{-1}. Further analysis showed the highest legacy N accumulation in areas with both high historical and recent inputs. These findings demonstrate that substantial N accumulation persists in subsoil layers, posing an ongoing risk to groundwater quality. While improved N use efficiency has reduced leaching loads, these findings are consistent with current understanding of subsurface N dynamics, whereby groundwater NO_3-N concentrations continue to decline but at a reduced rate. This trend reflects the persistence of legacy N stores, indicating that extended

timescales are required for their depletion before sustained improvements in water quality can be achieved. Future strategies should target deep soil monitoring and mineralization dynamics to improve nutrient management and accelerate recovery of groundwater quality.

Graphical Overview



3.2 Introduction

Over the past century, human activities have driven a 14-fold increase in reactive nitrogen (Nr) levels, primarily due to industrialization, intensive agriculture, and fossil fuel combustion (Brown et al., 2005; Carpenter et al., 1998; Chen et al., 2018; de Klein et al., 2017). This surge has led to widespread environmental issues, including coastal eutrophication, groundwater contamination, soil acidification, and greenhouse gas emissions (Arsenault et al., 2009; O'Brien et al., 2012). Elevated nitrogen (N) concentrations threaten freshwater and coastal ecosystems by promoting algal growth, accelerating eutrophication, and creating hypoxic zones, as observed in Lake Erie, the Baltic Sea, and the Gulf of Mexico (Diaz et al., 2017; Harding et al., 2019; Van Meter et al., 2018). Groundwater contamination is also a growing concern, with N levels exceeding safe drinking water limits in regions such as the United State of America (USA), the European Union (EU), and China, posing significant health risks, including increased cancer incidence (EEA, 2018; Hansen et al., 2019; Ward et al., 2018).

Intensive dairy farming is a major contributor to N pollution due to the industry's shift from localized crop-livestock systems to global supply chains, where feed production and livestock rearing occur in separate regions. For example, South American soybeans are used to feed dairy cows in the EU, altering N biogeochemical cycles and increasing pollution (Freeman & Thornton, 2006; Haan et al., 2010). In the EU, livestock supply chains contribute 82 % of agricultural ammonia (NH_3) emissions and 73 % of all agricultural emissions to water bodies (Leip et al., 2015). Dairy-exporting countries such as Ireland and New Zealand produce milk primarily for international markets, with Ireland exporting 85 % of its output and holding 10 % of the global infant formula market (More, 2009; Bord Bia, 2015). While dairy products are consumed globally, the environmental impacts including biodiversity loss, water pollution, and air quality deterioration remain concentrated in production regions (Galloway et al., 2003; Haan et al., 2010). To mitigate these issues, the further development of precision agricultural methods together with a deeper understanding of N transfers to groundwater, and tailored climate-specific management strategies are essential (Sutton et al., 2011; Uwizeye et al., 2020).

Nitrogen losses from intensive dairy systems arise from both fertilizer applications and direct animal inputs such as urine and dung, with losses ranging from 22-153 kg N ha^{-1}, similar to those in confined feeding operations in the USA and EU (McDowell et al., 2016, 2017). Effective N management is crucial for sustainability, influencing factors such as pasture fertilization, grazing intensity, and manure handling (Powell et al., 2010). Nitrogen Use Efficiency (NUE), which measures how efficiently N inputs are converted into productive outputs, are typically low in pasture-based dairy systems (<30 %), leading to high N surpluses which are susceptible to losses (Burchill et al., 2016). Improving NUE can simultaneously enhance productivity and reduce environmental damage by optimizing fertilizer use and balancing feed imports (Groot et al., 2006). However, balancing economic demands with environmental sustainability remains a challenge, as nutrient over-application continues to impact soil and water quality, particularly in intensive dairy regions of North Atlantic Europe (Creamer & O'Sullivan, 2018; R. Schulte et al., 2018; Volk et al., 2009). International policies aim to regulate N losses while supporting agricultural productivity, but compliance remains complex in high-intensity dairy systems (Sutton et al., 2013).

Previous farm scale research in Ireland using static and dynamic factors collated over a decade, showed how management changes on intensive dairy systems can positively influence NUE (Huebsch et al., 2013) and alter water quality trajectories for the better. Research showed that during a 11-year study period, where N fertilizer inputs decreased from 294 kg N ha^{-1} to

249 kg N ha^{-1} and stocking rates and productive outputs increased from 75 kg N ha^{-1} to 98 kg N ha^{-1}, the NUE increased from 22.4 % to 36 %. This shift led to average groundwater NO_3-N concentrations declining from 16 mg L^{-1} in 2002 to 6.6 mg L^{-1} in 2011 (Huebsch et al., 2013). However, loads of N loss to groundwater remained high and in excess of 30 kg N ha^{-1}. Factors such as soil type and thickness, water table upper position, underlying geology, climate variability, and management strategies (e.g., mixed pastures, regulated grazing, or N inhibitors) were all important in terms of understanding N dynamics. On another intensive dairy research farm with similar soil and geology, (Jahangir et al., 2013) showed high NO_3-N accumulation in groundwater was enabled by a deep-water table level, high soil and bedrock permeability and a low potential for nitrate-nitrogen (NO_3-N) reduction. The thick unsaturated zone provided perfect conditions for organic N and ammonium (NH_4^+) to be converted to NO_3-N, which accumulated as legacy N (organic matter in soil), which once mineralized could be leached in effective drainage to groundwater with minimal denitrification. However, the role of legacy N stores in the soil and their impact on groundwater NO_3-N concentrations remain a knowledge gap.

Therefore, to investigate the relationship between current groundwater NO_3-N concentrations, historical N application, and legacy N accumulation in a well-drained soil system, soil samples to 1 m depth were collected in summer 2024. The soil sampling campaign was designed to determine whether the legacy N pool in the soil profile has begun to respond to recent reductions in N inputs (i.e., implementation of low, medium and high treatments), and to evaluate whether observed changes in groundwater NO_3-N concentrations and loads reflect these recent modifications or remain influenced primarily by historical N loading.

The objectives of this 24 year (2001-2024) study on an intensive dairy farm were to: (1) compile borehole NO_3-N data and paddock scale farm activity data over the period, (2) use depth specific soil samples to 1 m obtained using soil coring to examine how total N and potential mineralization rates differ in the soil profile in different areas of the farm receiving different inputs perceived as low, medium and high since 2020 as compared to natural background conditions and (3) quantify legacy N to 1 m and discuss its role in affecting groundwater water NO_3-N trajectories over the study period.

3.3 Materials and Methods

3.3.1 Site Description

Our study site is a 48.1-hectare intensive Teagasc Dairy Research farm (locally known as Curtin's Farm) located at the Teagasc Dairy Production Centre, Fermoy, Co. Cork, Ireland (8°15′W, 52°10′N). Positioned on a lowland limestone plateau in southern Ireland, the site features flat topography with negligible runoff and lies up-gradient of the Funshion River, near a public water supply well and down gradient of the River Blackwater (Figure 3-1). The site has a meteorological (MET) station that provides in-situ hourly weather data. The 30-year average (1982–2011) rainfall and effective drainage (ED) data presented by Huebsch et al. (2013) highlight distinct seasonal patterns and inter annual variability. The 30-year average annual rainfall was 1022 mm, which exceeded the 11-year average (2001–2011) of 996 mm. Monthly rainfall peaked in October, with an average of 112 mm, while the lowest rainfall occurred in July at 64 mm. Weather data for the study site were obtained from the Met Éireann online database, which provides access to historical meteorological records. The nearest weather station to Curtin's Research Farm, located approximately 5 km away, was station number 575 (Moorepark). This station has been operational since 2003 and offers hourly, daily, and monthly weather data. Daily data on rainfall (mm), maximum and minimum air temperature (°C), potential evapotranspiration (mm), and evaporation (mm) were compiled for the period 2004–2024. In addition, soil moisture deficit (SMD) estimates were available from Met Éireann for different drainage classes. For this study, SMD values for well-drained soils were used for the period 2011–2024. To extend the SMD dataset, daily SMD values from 2007–2010 were computed using the hybrid SMD model by (Schulte et al. (2005). This model requires daily inputs of rainfall (mm), global radiation (J cm^{-2}), wind speed (m s^{-1}), maximum temperature (°C), and minimum temperature (°C). Due to incomplete weather records before 2007, it was not feasible to apply the SMD model for earlier years, as it does not tolerate missing input data.

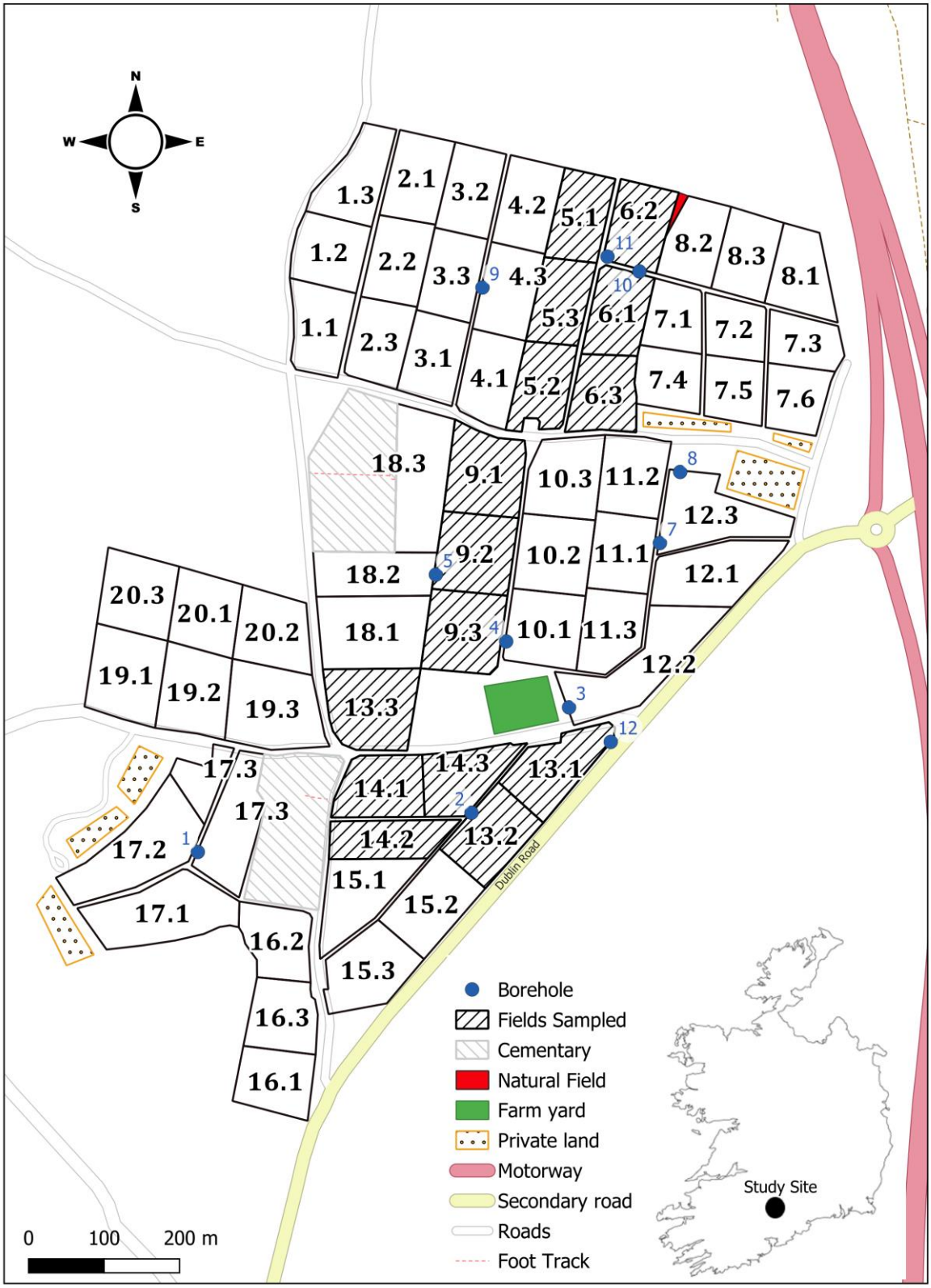


Figure 3- 1 Study site location and figure with area (ha) of each field and location of natural field (i.e., control) and sampled fields within the study.

Previous studies on this site have characterized the soil, sub-soil and geology and interactions between the farming system and NO_3-N transport within and off site. In summary, the soil is classified as freely drained acid brown earth (Haplic cambisol) derived from mixed sandstone-limestone glacial till (Kramers et al., 2009). It has a thickness ranging from 0 to 4.5 m (Bartley & Johnston, 2006), the A horizon comprises 53 % sand, 31 % silt, and 16 % clay, with a dry bulk density of 1.1 g cm^{-3}, total porosity of 52 %, with preferential flow limited to the A-B soil horizons (Kramers et al., 2009). The subsoil is underlain by karstified Waulsortian limestone bedrock at an average depth of 2.5 m (Bartley & Johnston, 2006). The Waulsortian limestone, which constitutes 1.7 % of Ireland's land area (Ryan et al., 2006), is more karstified and less bedded than other limestone formations, such as the Ballysteen Formation (GSI, 2000).

Five paddocks (Paddocks 5, 6, 9, 14 and 15) were selected for our study, with each paddock subdivided into three distinct sub-paddocks (e.g., 5.1, 5.2, 5.3 for Paddock 5) that differed in their N management practices. These differences included variations in fertilizer application rates, grazing intensity (expressed as grazing days ha^{-1}), slurry and dairy soiled water (DSW) application, and the types of sward species grown over time. Detailed farm activity data was available at these paddocks from 2001 to 2024, and this was used to develop our N surpluses. A natural area adjacent to the farm was used as a control area to estimate N accumulation.

Geophysical surveys, including surface conductivity (up to 6 m depth) and resistivity profiling (up to 50 m depth) were conducted to investigate the lateral and vertical variability of subsurface materials, depth to bedrock, and lithological properties (Huebsch et al., 2013). Electrical conductivity (EC) measurements identified three intervals, ranging from low (11–14 mS m^{-1}) to high (17–20 mS m^{-1}), which correspond to variations in overburden composition. High EC values were associated with collapse structures filled with finer materials, such as silt or clay, resulting from the karstified bedrock. These findings also revealed an elongated conduit system trending northwest to southeast in the central part of the farm. Resistivity profiles indicated that karst features, including conduits, air-filled cavities, and fractures, are prevalent in the bedrock and substantially influence the groundwater flow direction. Boreholes (BH) (Table A.1) were installed throughout the farm as in Figure 1 the details of which are summarized in Huebsch et al. (2013). In terms of BH connectivity, tracer experiments indicate that BH 7 is connected to 4, 5 and groundwater exits through in the direction of BH9. BH 8 was not connected to BH 7 nor to BH10-11, however, BH10 and 11 were connected to each other and BHs 2-3 and 12 were connected (Bartley et al. 2004). This connectivity has

implications in terms of how farm activities and N surpluses in one area of the farm affects NO_3-N concentrations in groundwater at different BH locations.

3.3.2 Groundwater Quality Data

A farm-scale groundwater quality monitoring program was initiated in 2002 to assess NO_3-N concentrations across 12 BH on the farm (Figure 1). In addition to NO_3-N, the analysis included measurements of NH_4-N, NO_2-N, total nitrogen (TN), and total oxidized nitrogen (TON), all reported in mg L^{-1} following procedures as outlined in Jahangir et al. (2012). It should be noted that data are unavailable for the period 2017–2021 due to a lapse in sampling during those years. However, the present study utilizes ~1200 sampling points from 2002-2024 for NO_3-N concentrations in groundwater, including ~1200 and ~1160 samples for NO_2-N and NH_4-N, respectively. In the study by Huebsch et al. (2013), NO_3-N concentrations in groundwater showed a substantial decline over the 10-year observation period (2002-2011). Specifically, median NO_3-N concentrations decreased from approximately 16 mg L^{-1} in 2002 to 7.3 mg L^{-1} in 2010 across monitored BHs, with a low of 6.6 mg L^{-1} in 2011. Huebsch et al. (2013) attributes this reduction to regulated N application (i.e., best practice as outlined in National Action Programme (NAP), changes in land use and cropping systems, improved manure management and storage.

To quantify potential N losses to groundwater, NO_3-N concentrations were converted to leaching loads (kg N ha^{-1}) using the method described by Hooker et al. (2008), which incorporates effective drainage (ED) and NO_3^- concentration in leachate. For this study, an ED value of 480 mm yr^{-1} was applied. Long-term studies at the same research site (Fermoy, Co. Cork) reported effective drainage values ranging from 478 to 544 mm yr^{-1} (Huebsch et al., 2013), with Fenton et al. (2017) adopting a representative value of 500 mm yr^{-1} for a nearby site with comparable soils and management. Similarly, Jahangir et al. (2012) found effective drainage within the range of 481–557 mm yr^{-1}, further supporting the use of 480 mm yr^{-1} as a representative benchmark for free-draining soils in southern Ireland. This analysis enabled the reconstruction of a 24-year trajectory of groundwater NO_3-N concentrations and corresponding leaching loads. The resulting time series were plotted to assess temporal patterns in NO_3^- dynamics and to evaluate whether observed concentrations and leaching losses reflected changes in N management practices on the farm. In particular, leaching estimates were compared against the water quality threshold of 30 kg N ha^{-1} (Lakhanpal et al., 2025) to determine whether this target was achieved under both historical and more recent management

regimes. Achieving leaching loads below this threshold results in the receiving water body being classified as having good water quality.

3.3.3 Farm scale N Surplus

All farms received N inputs in the form of mineral fertilizers, as well as slurry and DSW, and detailed data on these sub-paddock specific N inputs were available from the farm activity data in units of kg N ha^{-1} yr^{-1} (Figure 3-2). Mineral fertilizer inputs were higher in the early 2000s, reaching values as high as 360 kg N ha^{-1} yr^{-1}, and transitioned around 2006 to lower values, reaching values as low as 140 kg N ha^{-1} yr^{-1} in some of the paddocks, in recent years. The farm has maintained a herd of Holstein-Friesian cows, with stocking rates ranging from 2.5 to 2.8 livestock units (LU) per hectare. In recent years, these rates have adjusted in response to the intensification of dairy farming practices, aligning with requirements under the EU Nitrates Directive (EU ND, Council Directive 91/676/EEC 2019). The ND is Ireland's program of measures (PoM) used to achieve at least good status in all water bodies under the EU Water Framework Directive (WFD). It is implemented nationally as the NAP and is reviewed every few years to incorporate the latest scientific findings. Since its implementation in 2007, the farm has adopted best nutrient management practices, emphasizing increased slurry use efficiency and reductions in fertilizer N application to meet the levels mandated by statutory instruments. These transition in management practices that occurred in response to EU ND, allowed us to explore the impact of both historical and current management practices on N legacies. The current (2023-2024) inorganic fertilizer inputs were used to classify the paddocks into three distinct N input regimes: low (140 to 170 kg N ha^{-1} yr^{-1}), medium (170 to 220 kg N ha^{-1} yr^{-1}), and high (220 to 300 kg N ha^{-1} yr^{-1}).

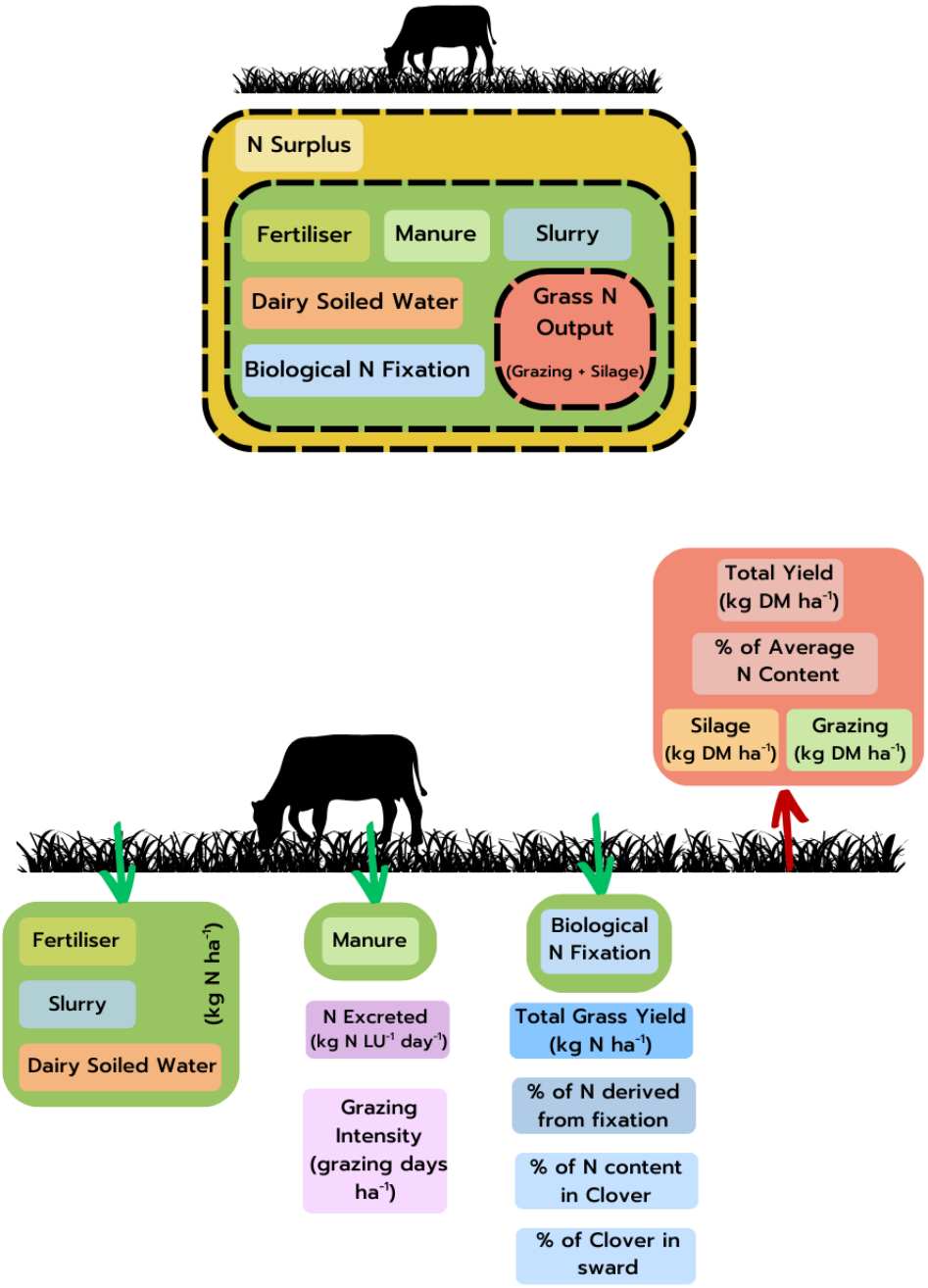


Figure 3- 2 Schematic representation of the nitrogen (N) surplus approach, illustrating all parameters and historical datasets used to estimate farm-scale N surpluses over a 24-year period.

In addition to inorganic fertilizer, the paddocks also received slurry and DSW from when the animals are housed indoors, specifically slurry generated during winter is reapplied to the land during the subsequent grazing season. The frequency and intensity of N-rich inputs such as slurry and DSW applications increased markedly after 2010, followed by a gradual decline in subsequent years. The paddocks also receive manure inputs from when the animals

are grazing on the fields. With historical stocking rates exceeding 3 LU ha^{-1}, while a transition to improved management practices in 2013 has contributed to reduced stocking rates averaging approximately 2.5 LU ha^{-1}. The farm activity data provided information on paddock and year-specific grazing intensity from 2013 – 2024 (grazing days ha^{-1} yr^{-1}), which is a measure of pasture utilization representing the cumulative number of livestock grazing days per hectare over a defined period. To backfill grazing data for the years 2001–2012, the mean grazing days ha^{-1} from 2013 to 2020 were used. This reference period was selected because no major changes in farm N management occurred prior to 2020. This approach provided a reasonable estimate of historical grazing pressure during the early years of the study. The grazing intensity was used to calculate the paddock specific stocking rate (LU ha^{-1} yr^{-1}), by dividing grazing days ha^{-1} by the average grazing season length of 280 days. While grazing duration in Ireland varies annually due to regional climate and soil drainage class (Cahill et al., 2023; O'Loughlin et al., 2016), a 280-day average was deemed appropriate for the farm, where grazing practices typically exceed the national average. According to (Läpple et al., 2012), the national average was 223.2 days, and 233.3 days in the south of the country based on National Farm Survey (2009) data. Finally manure N output per paddock (2001–2024) was estimated by multiplying grazing intensity (grazing days ha^{-1}) by daily N excretion values (358.4 g N LU^{-1} day^{-1}), based on the findings of Bougouin et al. (2022). In that study, average N excretion per cow was reported as 184.0 g day^{-1} in feces (SD ± 50.38) and 175.5 g day^{-1} in urine (SD ± 66.22), yielding a total of 358.4 g day^{-1} (SD ± 96.14).

Grass N output was calculated by summing grazing and silage tonnage (kg DM ha^{-1}) and multiplying by their respective N content: 3.2 % for grazed grass and 2.5 % for silage (DairyNZ, 2013). As grazing and silage tonnage data were only available from 2013 to 2024, the same backfilling approach, using the 2013–2020 mean was applied to estimate values for 2001–2012. The biological nitrogen fixation (BNF) was estimated using annual sward composition data provided for each paddock (2001–2024). Swards included perennial ryegrass (PRG), perennial ryegrass with white clover (PRGWC), and mixed-species swards (i.e., white clover, forbs, and grasses). White clover was introduced only after 2021; hence, no BNF was assumed prior to that year. Clover yield was estimated by multiplying total grass yield (kg DM ha^{-1}) by the percentage of clover in each paddock per year. Clover yield was then multiplied by an N content of 4.6 %, and BNF was estimated assuming 55 % of total clover N was derived from fixation, using a value of 26 kg N fixed per t DM (Widdup et al., 2001). Total N inputs were calculated as the sum of fertilizer, slurry, DSW, manure, and BNF. These were then

compared with N outputs (grass N removal via grazing and silage) to derive the full N surplus over the 24-year period. Atmospheric N deposition was not included in this study, as it represents <0.02 % of the total N inputs to the system. At our study site, deposition was estimated at 13.4 kg N ha^{-1} yr^{-1} (UK Centre for Ecology & Hydrology, n.d.), which falls within the regional range of 10–15 kg N ha^{-1} yr^{-1} (Aherne et al., 2014; Kelleghan et al., 2021).

Fine-resolution historical farm data were collated from 2001 to 2024 to estimate N surplus at the sub-paddock scale for all 15 sub-paddocks (5.1, 5.2, 5.3…) over a 24-year period (Figure 3-3). This dataset includes detailed paddock-level information on annual N inputs from fertilizers, slurry, and DSW; grazing days; silage and grazing yields; sward species composition; and reseeding rates (Figure 3-2). All components of the N surplus such as fertilizer, DSW, slurry, manure, grass N output from grazing and silage were accounted for, except for environmental losses such as leaching losses, volatilization, and denitrification for this study.

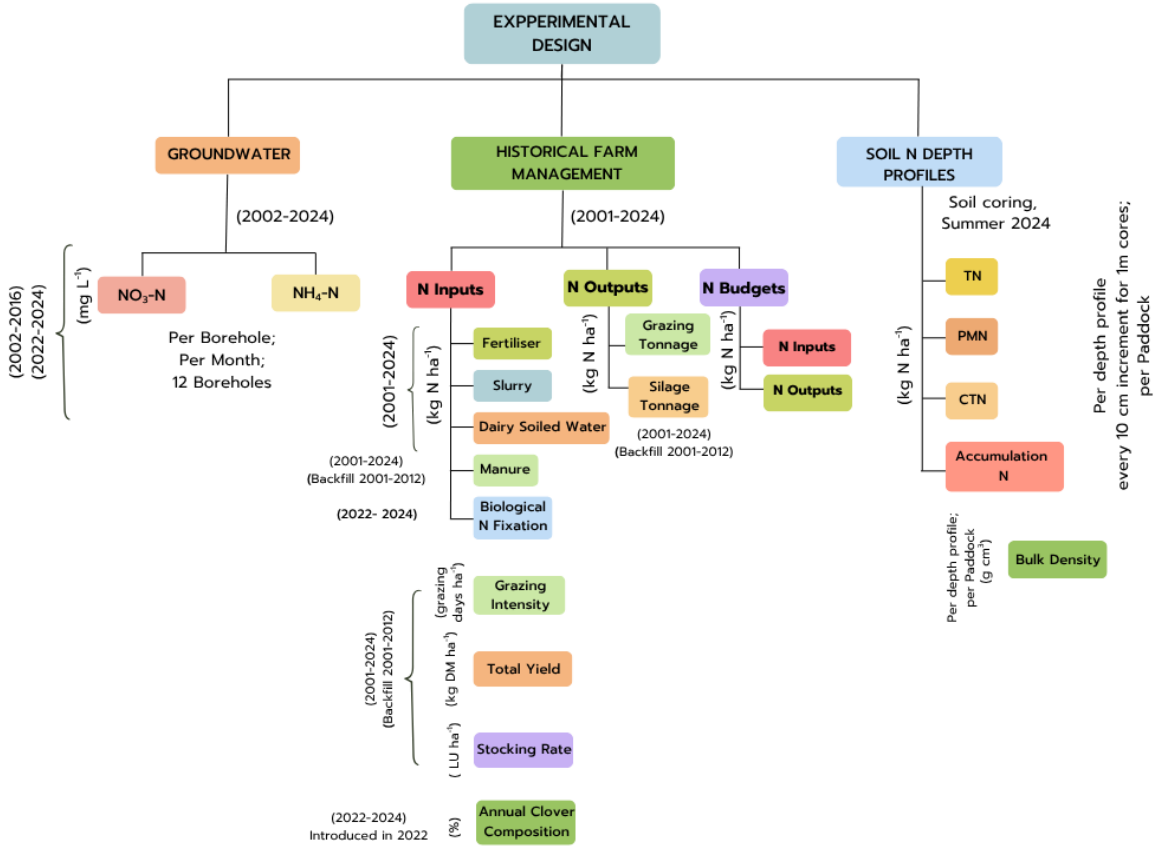


Figure 3- 3 Conceptual framework of the study experimental design showing the organization of all nitrogen-related data used. Historical data sources (e.g., groundwater nutrient records, farm N management) and newly collected soil profile data (e.g., total nitrogen-TN, potential mineralizable nitrogen-PMN, cumulative total nitrogen-CTN, and accumulation nitrogen- Accumulation N) are grouped to illustrate their temporal coverage and contribution to understanding N dynamics in intensive dairy systems. Indicate the flow of information and the relationships among measured and derived N variables.

3.3.4 Soil Profile sampling and in-depth analysis

3.3.4.1 Soil coring to 1 m

Soil cores were taken at the 15 sub-paddocks, following a W-shaped transect pattern with five randomly selected sampling points to ensure a spatially representative estimate of treatment effects (Wall & Plunkett 2021). This sampling framework supports an integrated assessment of short-term and long-term N dynamics across soil and groundwater compartments under contrasting management histories. Samples were extracted using a 1m depth (50 mm diameter) Van Walt™ Percussion Window Sampling System driven into the ground with a petrol pneumatic hammer (Atlas Copco™ Cobra TT) and equipped with lifting gear and ball

clamps for sample retrieval. Each 1 m soil core was divided into 0.1 m segments, manually retrieved, and stored in labelled sample bags. For each of the five sampling points within a paddock, samples were composited in situ by depth, resulting in depth-stratified composite samples for each paddock. Additionally, a natural area with no agricultural inputs or management was selected as a control site (henceforth referred to as control site), where a single core was extracted and equally sub-divided into 0.1 m segments. In total, 750 soil samples were collected prior to compositing. After compositing by depth and paddock, the dataset was reduced to 160 laboratory samples, including the control site core. The fresh (field-moist) samples were kept refrigerated (<4 °C) with minimum disturbance and processed for laboratory analyses as soon as possible (within 48 hours), in which samples were first weighed and subsequently oven-dried at 40 °C for 7 days until constant weight, in order to calculate gravimetric soil water content (%; w:w). Following drying, all samples were mechanically sieved to <2 mm and thoroughly homogenized for laboratory analyses.

3.3.4.2 Soil Total N and potential mineralizable N

To determine potential mineralizable N (PMN), we followed the short-term (7-day) anaerobic incubation method described by Bremner & Keeney (1966) and Schomberg et al. (2009). The method provides a direct measure of the quantity of organic N mineralized to inorganic N over a defined period under controlled environmental conditions (Ros et al., 2011). Biological measures of the mineralization potential of soil organic matter nitrogen (SOM-N) using anaerobic incubation methods are widely regarded as standard practices for understanding N dynamics. These methods effectively reflect the biological activity of soil-dwelling organisms that facilitate the mineralization of N from labile organic N pools (Dahnke & Vasey, 1973).

Two sets of equivalent dried subsamples (5 g each) were transferred into 16 x 150 mm screw-capped test tubes and waterlogged with 12.5 mL of deionized water to create anaerobic conditions. One of the two sets was immediately extracted, while the second was incubated at 40 °C for 7 days (AI-7) before extraction. The resulting soil slurries were extracted by adding 25 mL of 2 M KCl and 60 minutes of orbital shaking, followed by filtration (Whatman No. 42 filter paper), and inorganic nitrogen (ammonium-nitrogen (NH_4-N), nitrite-nitrogen (NO_2-N), nitrate-nitrogen (NO_3-N)) concentrations (mg L-1) were measured colorimetrically using an Aquakem™ 600A analyzer (Aquakem 600A, 01621, Vantaa, Finland). The amount of N

mineralized was calculated as the difference in NH_4^+ concentrations between the initial extraction (time zero) and the extraction after 7 days. Extractable inorganic nitrogen (NO_2-N + NO_3-N and NH_4-N) was determined using a method adapted from Keeney & Nelson (1982) and Mulvaney (1996). Soil subsamples (50 g) were mixed with 100 mL of 2 M KCl in a 250-mL conical container, shaken on an orbital shaker for 1 hour, and the resulting suspension filtered through Whatman No. 42 filter paper. Concentrations of extractable NO_2-N, NO_3-N and NH_4-N were quantified by colorimetric analysis using a SKALAR SAN++ continuous flow analyser (Skalar Analytical B.V., Breda, The Netherlands).

To assess PMN, NH_4-N and TON concentrations from both day 0 and day 7 were converted to mg kg^{-1} based on soil dry mass to calculate total mineral nitrogen (TMN). PMN was determined by subtracting day 0 TMN from day 7 TMN, representing the PMN (mg kg^{-1}) in the soil. Finally, PMN was expressed on a per-day basis by dividing by 7, following the approach described by Hübsch (2015). Additional soil subsamples were further analyzed for pH in deionized water (1:2.5; w:v), and ball-milled (23Hz for 120 seconds, Retsch™ MM 200) in order to determine soil TN and total organic carbon (TOC) concentrations (%; w:w) on 0.25 g of sample by dry elemental combustion analyzer (LECO™ CNS928) after a 24 h hydrochloric acid pre-treatment for removal of inorganic C.

TN concentrations (mg kg^{-1}) obtained from the LECO analysis were multiplied by depth specific dry bulk density, which was derived from soil wet weight, soil dry weight, and core dimensions for each depth interval. In a small number of cases the value for dry bulk density in the upper two depths were combined. In the control core the dry bulk density was calculated for depth intervals to 50 cm. This provided estimates of TN in kg N ha^{-1} for all depths and for each 1 m depth profile. N accumulation (kg N ha^{-1}) at each depth was further calculated by subtracting the TN of the corresponding control core from that of the individual paddock cores for the respective depths, allowing comparison across 1 m profiles.

3.3.5 Data Analysis

To assess long-term trends in groundwater quality, monthly NO_3-N concentration data from 2002 to 2024 were compiled and analyzed. The primary objective was to evaluate the trajectory of NO_3-N concentrations in groundwater relative to the European Commission's maximum admissible concentration (MAC) for drinking water supplies of 11.3 mg NO_3-N L^{-1}. Descriptive statistics including mean, median, standard deviation, minimum, and maximum

values were calculated for both NO_3-N and NH_4-N concentrations across the monitoring period. To analyze trends in groundwater NO_3-N concentrations over time, we applied the Mann-Kendall trend test (Mann, 1945; Kendall, 1975), a non-parametric method commonly used for detecting significant trends in environmental time series. This test is robust to non-normal distributions and missing values, making it suitable for our dataset, which includes variability in sampling frequency and temporal gaps. The magnitude of the trend was quantified using Sen's slope estimator (Sen, 1968), which calculates the median of all pairwise slopes in the time series. This provides an estimate of the rate of change in NO_3-N concentration over time, expressed in mg L^{-1} yr^{-1} (for annual data) or mg L^{-1} mo^{-1} (for monthly data). Trend analyses were performed for two periods: 2002–2016, representing the earlier monitoring phase with complete time series data, and 2022–2024, covering the recent post-intervention phase. Statistical calculations were conducted in R using the Kendall and trend packages.

A quadrant analysis was conducted to explore the relationship between N management history and subsoil N accumulation over the top 50 cm depth, inspired by the approach used by Xu et al. (2018). Quadrant analysis is a visual classification tool that enables the comparison of observations across two independent variables by grouping them into four quadrants. It has been used in agricultural systems research to examine interactions between management intensity and environmental outcomes (e.g. King et al., 2018). In this study, it was applied to assess how combinations of historical and current N inputs relate to subsoil N accumulation under differing management trajectories.

Current N input was calculated as the average annual input from fertilizer, DSW, slurry, manure, and BNF over 2023 to 2024, while historical input was defined as the mean annual input over the longer period from 2001 to 2024. Paddocks were grouped into four categories based on whether their historical and current inputs were above or below the overall mean values: (1) high historical–high current, (2) high historical–low current, (3) low historical–high current, and (4) low historical–low current.

3.4 Results

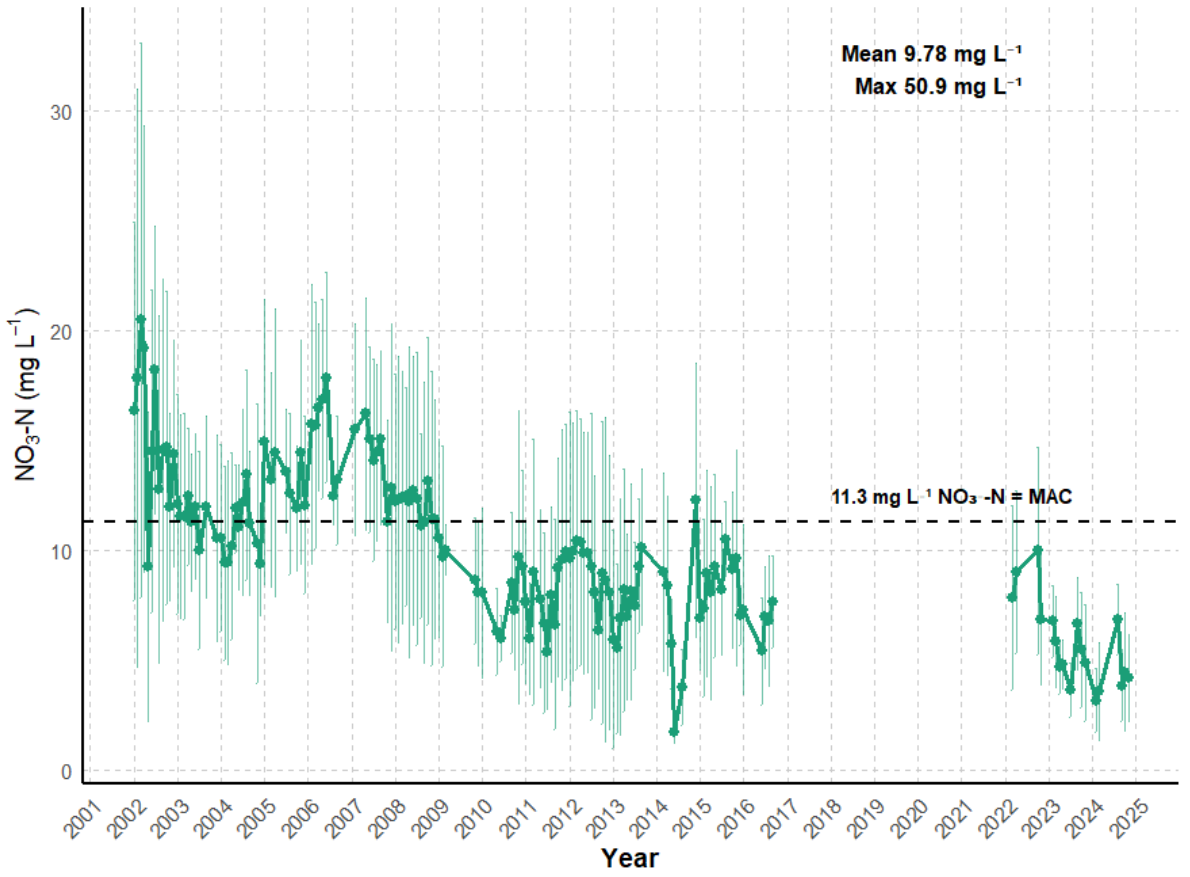
3.4.1 Groundwater N concentrations

Table 3-1 shows us that over the 22-year monitoring period (2002–2024), a total of 1198 samples were analyzed for NO_3-N (Table 1), with a mean concentration of 9.78 mg L^{-1}, a median of 9.13 mg L^{-1}, and a standard deviation (SD) of 5.86 mg L^{-1}. Concentrations ranged from below the detection limit (≤0.02 mg L^{-1}) to a maximum of 50.9 mg L^{-1}, often exceeding the EU drinking water threshold of 11.3 mg L^{-1} (equivalent to 50 mg L^{-1} NO_3^-). NH_4-N, measured in 1168 samples, had a mean of 0.31 mg L^{-1}, a median also ≤0.02 mg L^{-1}, and an SD of 0.84 mg L^{-1}, with highly elevated values indicating contamination events from surface to groundwater up to 11 mg L^{-1}.

Table 3- 1 Total number of samples, mean, median, standard deviation, maximum and minimum groundwater concentration in mg L^{-1} from 2002-2024 across the entire farm.

N Species Concentration	***Total No. of Samples***	***Mean***	***Median***	***Standard Deviation***	***Maximum***	***Minimum***
NO_3-N (mg L^{-1})	1198	9.78	9.13	5.86	50.9	≤ 0.02
NH_4-N (mg L^{-1})	1168	0.31	≤ 0.02	0.84	11	≤ 0.02

Temporal patterns in NO_3-N concentrations are shown in Figure 3-4, which presents monthly values by BH from 2002 to 2024. Substantial spatial and temporal variability was observed, with concentrations ranging from below the detection limit to values exceeding 50 mg L^{-1}. Elevated NO_3-N levels were most frequent during the early monitoring period (2002 to 2010), particularly in BH 4, BH 5, and BH 8. A marked decline in NO_3-N concentrations was observed after 2010. During the most recent monitoring period (2022 to 2024), concentrations remained low, with an average of 5.64 mg L^{-1}. Frequent exceedances of the groundwater NO_3-N MAC were recorded prior to 2010, with average concentrations reaching 12.5 mg L^{-1}. From 2011 onward, values declined consistently, with recent averages of 7.63 mg L^{-1}. Although values remained below MAC, they appeared to plateau in the most recent years.



Abbreviations used: N: Nitrogen, NO_3-N: Nitrate, MAC: Maximum Admissible Concentration.

Figure 3- 4 Monthly nitrate-N (NO_3-N) concentrations by borehole from 2002-2024, with missing data from 2017-2021.

Mann-Kendall trend tests revealed statistically significant negative trends in NO_3-N concentrations across both annual (2002–2016: $\tau = -0.638$, $p = 0.0011$) and monthly time series (2002–2016: $\tau = -0.516$, $p < 0.001$; 2022–2024: $\tau = -0.516$, $p = 0.0031$) (Table 3-2). Corresponding Sen's slope estimates indicated median rates of decline of -0.524 mg L^{-1} yr^{-1} (annual 2002–2016), -0.060 mg L^{-1} mo^{-1} (monthly 2002–2016), and -0.266 mg L^{-1} mo^{-1} (monthly 2022–2024), with 95 % confidence intervals provided in each case.

Table 3- 2 Results of Mann-Kendall trend tests and Sen's slope estimates for annual and monthly groundwater nitrate-N (NO_3-N) concentrations over different time periods. Sen's slope indicates the median rate of change in NO_3-N concentrations, with 95 % confidence intervals (CI) for each estimate.

Period	*Mann-Kendall τ*	*p-value*	*Sen's Slope*	*95 % CI (Low–High)*
Annual (2002–2016)	–0.638	0.0011	–0.524	–0.785 to –0.257
Monthly (2002–2016)	–0.516	<0.001	–0.060	–0.071 to –0.049
Monthly (2022–2024)	–0.516	0.0031	–0.266	–0.411 to –0.120

3.4.2 N Surpluses over the years

3.4.2.1 Per-paddock inputs, outputs and surplus

The total N inputs, grass N outputs, and the resulting N surplus (input minus output) for each paddock is represented in Table 3-3. Grass N output varied considerably across paddocks, ranging from ~360 to over 460 kg N ha^{-1} yr^{-1} (Table 3). Notably, paddocks 5.2 and 5.3 exhibited the highest outputs, yet still accumulated large N surpluses, as evidenced by their substantial N surplus values. The N surplus, shown as a separate orange bar, provides an estimate of potential surplus N retained or lost from the system.

Table 3- 3 Paddock-scale cumulative nitrogen inputs, outputs, and surplus from 2001 to 2024. Values represent total nitrogen (kg N ha⁻¹ yr⁻¹) applied or removed over a 24-year period per paddock. Inputs include fertilizer, slurry, dairy soiled water (DSW), manure, and biological nitrogen fixation (BNF). Outputs refer to grass nitrogen removed via grazing and silage. The N surplus is calculated as total N inputs minus grass N output.

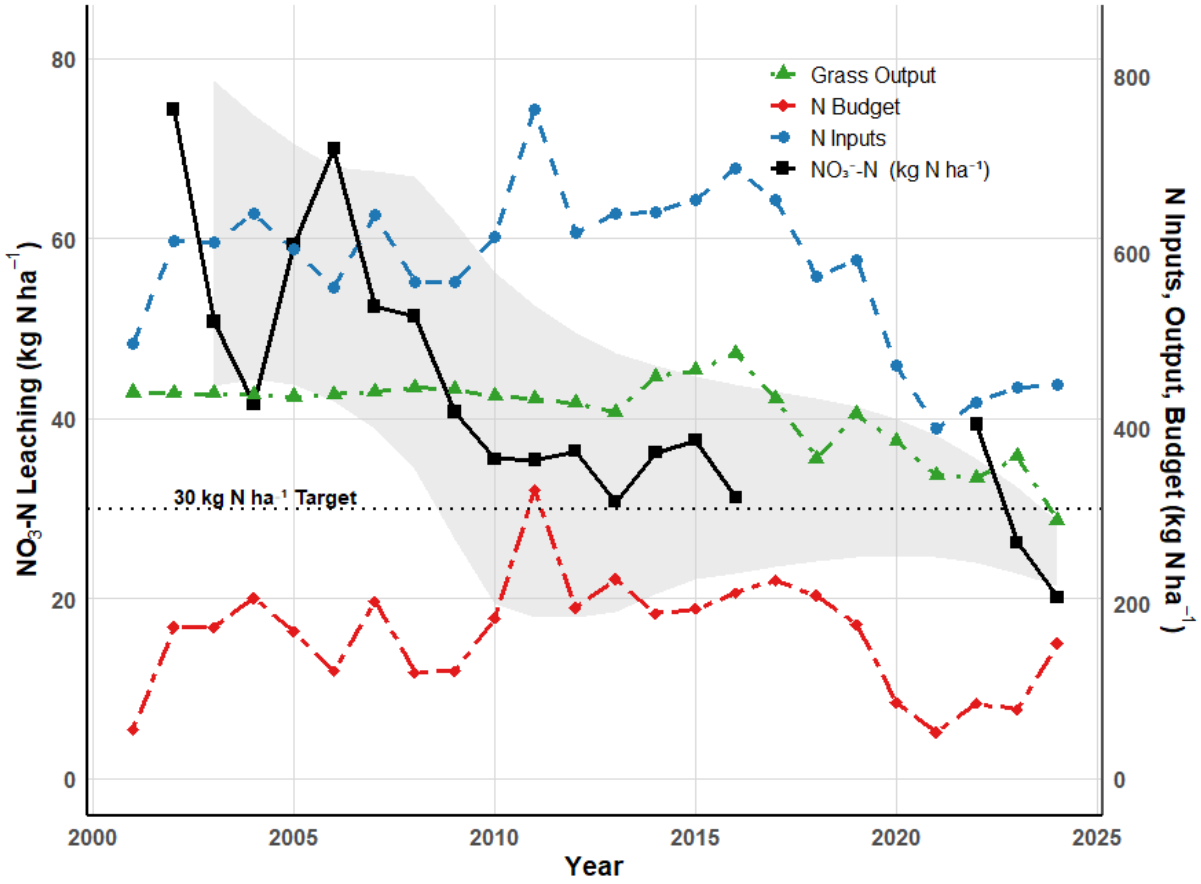
Paddocks	*Fertilizer*	*Slurry*	*DSW*	*Manure*	*BNF*	*Total N Inputs*	*Grass N Output*	*N Surplus*
	(kg N ha^{-1} yr^{-1})							
5.1	259.57	86.32	26.04	260.28	0.00	632.21	439.00	193.21
5.2	240.06	98.32	25.00	281.38	7.97	652.73	398.10	254.64
5.3	245.10	93.39	18.33	227.38	5.39	589.59	410.67	178.93
6.1	240.45	106.47	15.01	227.72	0.00	589.65	366.95	222.70
6.2	219.67	97.49	10.21	201.38	4.62	533.38	388.30	145.08
6.3	222.29	89.74	8.30	286.88	2.54	609.75	407.39	202.35
9.1	256.99	46.33	27.79	208.33	0.00	539.44	393.73	145.72
9.2	232.24	41.35	22.71	245.43	4.67	546.40	385.00	161.40
9.3	232.59	48.89	26.26	232.16	7.43	547.33	423.09	124.25
13.1	255.42	54.50	4.88	273.49	0.00	588.28	415.24	173.04
13.2	236.61	62.31	4.25	179.78	7.04	489.99	422.96	67.04
13.3	247.08	72.72	7.99	240.56	4.10	572.45	427.61	144.83
14.1	263.16	43.89	9.50	301.81	0.00	618.36	451.13	167.23
14.2	251.75	56.95	9.83	237.99	6.44	562.96	465.97	96.99
14.3	240.10	57.05	7.11	341.42	7.74	653.41	469.02	184.40

Abbreviations used: N: Nitrogen, DSW: Dairy Soiled Water, BNF: Biological Nitrogen Fixation.

Fertilizer accounted for the largest proportion of N inputs in all paddocks, contributing between 40 % and 55 % of total N. Slurry and manure were also major contributors, particularly in paddocks 6.1, 5.3, and 14.3. DSW made moderate contributions in some paddocks, while BNF accounting for less than 5 % of total N inputs. Cumulative N inputs ranged from ~490 kg N ha⁻¹ yr⁻¹ in paddock 13.2 to over 650 kg N ha⁻¹ yr⁻¹ in paddock 14.3. The largest cumulative N surpluses were observed in paddocks 5.2, 6.1, and 6.3, each exceeding 200 kg N ha⁻¹ yr⁻¹. Conversely, paddocks 14.2 and 13.2 had more moderate N surplus (~60-90 kg N ha⁻¹ yr⁻¹), reflecting a closer balance between inputs and outputs.

3.4.2.2 Time series trends across 24 years

To assess annual variation in N dynamics at the farm scale, trends in N inputs, grass N output, N surplus, and modelled NO_3-N leaching from 2001 to 2024 are shown in Figure 3-5. N inputs ranged from approximately 400 - 700 kg N ha^{-1} yr^{-1}, with highest values observed between 2006-2015 and a general decline in later years. Grass N output remained within a range of approximately 300-500 kg N ha^{-1} across the monitoring period. The N surplus, calculated as inputs minus outputs, ranged from approximately 100 - 300 kg N ha^{-1} yr^{-1}, with lowest values recorded in the recent years of the dataset. Modeled mean NO_3-N leaching values exceeded 70 kg N ha^{-1} in 2002 and remained above 30 kg N ha^{-1} through most of the period from 2003 to 2016. Values were below 30 kg N ha^{-1} in 2013 (30.8), 2016 (31.2), 2023 (26.2), and 2024 (20.1). The lowest annual leaching estimate was recorded in 2024, with a mean of 20.1 kg N ha^{-1} and a standard deviation of 1.6 kg N ha^{-1}. From 2023 onward, modeled mean NO_3^--N leaching values remained consistently below the 30 kg N ha^{-1} threshold. Earlier years exhibited greater inter-annual and spatial variability, with standard deviations ranging up to 23.7 kg N ha^{-1} in 2007.



Abbreviations used: N: Nitrogen, NO_3-N: Nitrate.

Figure 3- 5 Annual trends in nitrate-N (NO_3-N) leaching (black line, left axis), total nitrogen (N) inputs (blue), grass N output (green), and N surplus (red) from 2001 to 2024. Values represent annual means calculated across all monitored paddocks. Shaded area around NO_3-N line indicates standard deviation across monitored paddocks. Dashed line marks the target threshold of 30 kg NO_3-N leached ha^{-1}.

3.4.3 Depth profiles of Soil Total Nitrogen and PMN across paddocks

The depth soil profiles of TN (down to 1 m depth and stratified by TN application rates (in kg N ha^{-1}) averaged over 2023–2024) across five managed paddocks (5, 6, 9, 13, and 14) were compared to the control (see Figure 1, taken from non-agricultural natural field) reference profile (Figure 3-6). Results show greater TN in upper soil layers (0–40 cm), with magnitude increasing under higher N application rates. TN values ranged from 83.6 kg N ha-1 to 5223.4 kg N ha-1 along the depth profiles for the different paddocks. TN concentrations were highest

in the surface 0–10 cm across all paddocks, aligning with zones of N input and organic matter accumulation. With majority of the TN observed in the top 20 cm, TN concentrations consistently decreased with depth, suggesting limited vertical movement. Paddocks receiving high N inputs e.g., 5 and 6, with rates up to 310 kg N ha-1 yr-1 exhibited elevated TN concentrations throughout the soil profile compared to lower-input paddocks. In Paddock 5, TN increased proportionally with increasing application rates across all depths, reaching up to 290 kg N ha-1 yr-1. Similarly, in Paddock 6, peak TN levels (with N application rate of 310 kg N ha-1 yr-1) were observed and maintained to a depth of 50 cm. In Paddock 9, TN concentrations in the 0–20 cm layer followed incremental input trends, but subsurface values showed greater variability. The control reference site exhibited consistently lower TN values, particularly below 20 cm. Paddock-specific patterns were also observed: Paddock 13 showed relatively uniform TN across depths and application levels, whereas Paddock 14 demonstrated deeper TN accumulation under higher input scenarios.

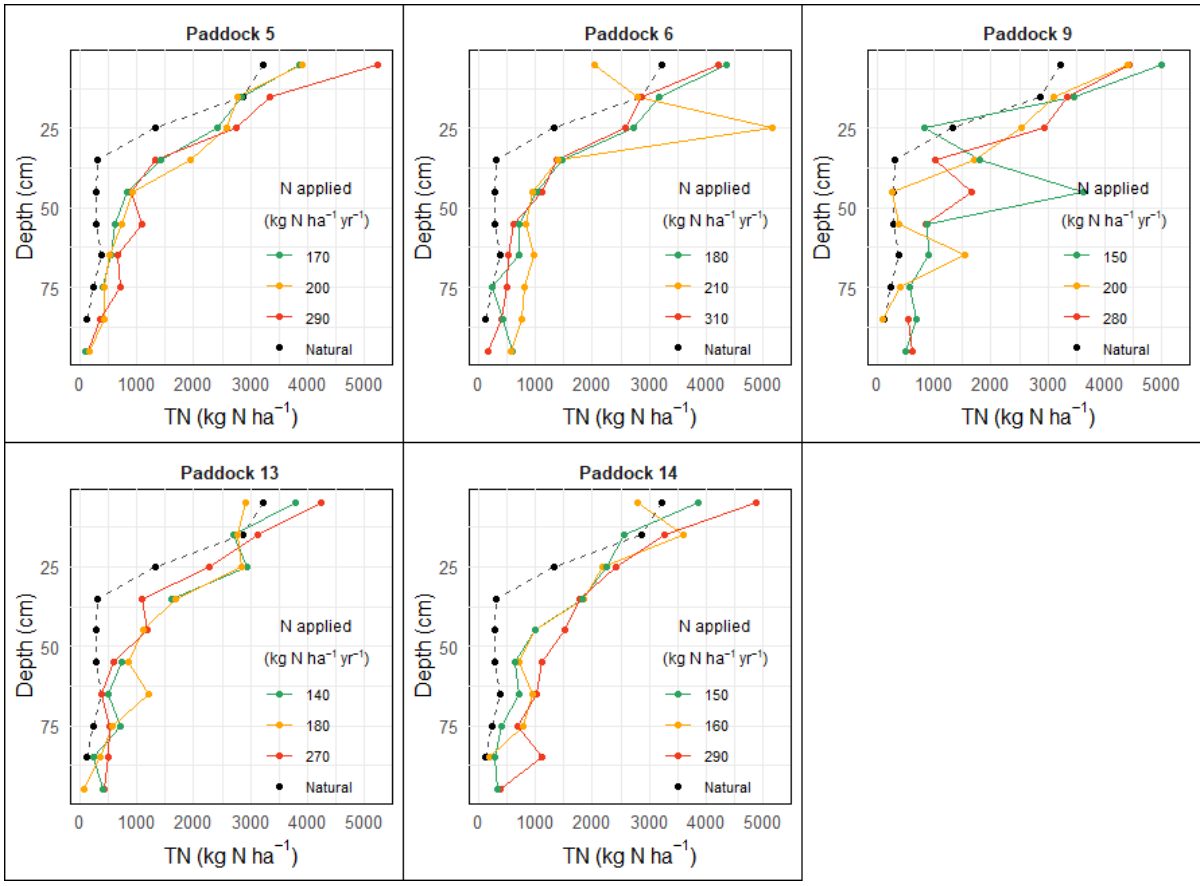


Figure 3- 6 Depth profiles of Total Nitrogen (TN) to 1 m across paddocks under varying N application rates (fertilizer, slurry, DSW). Each panel represents a paddock, showing the vertical distribution of TN (kg N ha⁻¹) measured at 10 cm intervals. Colored lines represent paddocks with varying average N applied (kg N ha⁻¹ yr^{-1}) during 2023–2024, while the dashed black line indicates TN distribution under control (non-fertilized) reference conditions.

The vertical distribution of PMN was assessed across five managed paddocks (5, 6, 9, 13, and 14) and a control reference site. All paddocks exhibited distinct PMN enrichment in the surface layers (0–10 cm), with concentrations declining markedly with depth. PMN was mostly available in the top 50cm with an average of 383.3 kg N ha^{-1} and only 2 kg N ha^{-1} available below the 50 cm depth (Figure 3-7). This pattern was consistent across all management regimes. Paddocks receiving higher N inputs exhibited greater PMN concentrations in the upper soil profile. For example, in Paddock 5, surface PMN ranged from approximately 60 to over 160 kg N ha⁻¹, corresponding to application rates of 170 to 290 kg N ha^{-1} yr^{-1}. Similar associations were observed in Paddocks 6 and 14, where maximum PMN values in the top 10 cm aligned with the highest N application rates (310 and 290 kg N ha^{-1} yr^{-1}, respectively). Below 30 cm, differences in PMN between application levels diminished

substantially, and by 50 cm depth, values converged across all treatments. The control site consistently exhibited lower PMN concentrations in the top 30 cm compared to the managed paddocks, while deeper values (>50 cm) were comparable across all sites. Paddock 9 showed greater variation in surface PMN among treatments, particularly in the 0–20 cm range. Paddocks 13 and 14 showed more stable PMN profiles with less separation across input levels.

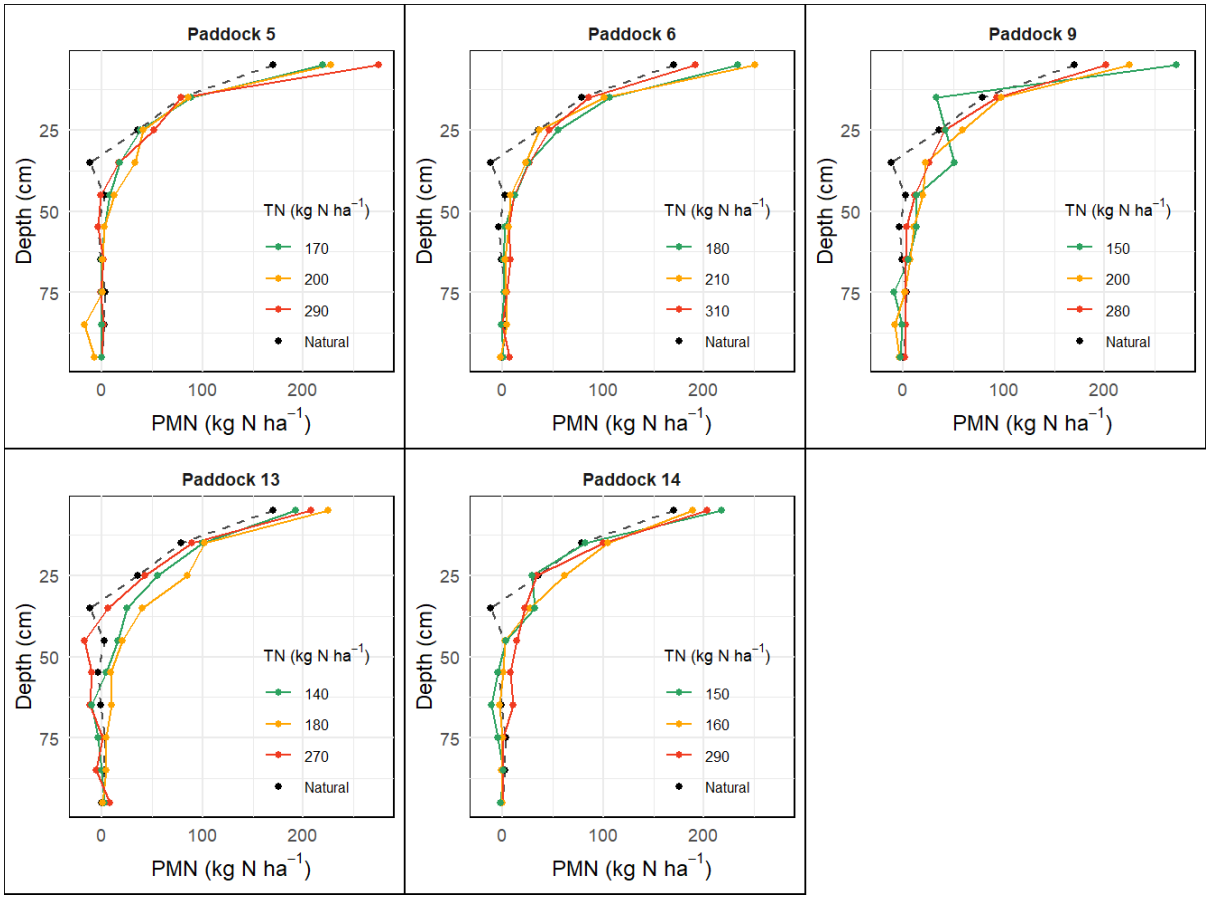


Figure 3- 7 Depth profiles of Potential Mineralizable Nitrogen (PMN) to 1 m across paddocks under varying N application rates (fertilizer, slurry, DSW). Each panel represents a paddock, showing PMN (kg N ha^{-1}) concentrations measured at 10 cm increments from surface to 1 m depth. Colored lines represent paddocks with varying average N applied (kg N ha^{-1} yr-1) during 2023–2024, while the dashed black line indicates PMN distribution under control (non-fertilized) reference conditions.

3.4.4 Nitrogen accumulation as a function of N input categories

We estimated depth specific N accumulation by subtracting paddock and depth specific TN values from corresponding values at the reference site. Accumulated N content within the soil profile differed substantially across paddocks and was strongly influenced by total N input

rates (Figure 3-8). In high N input paddocks (220–300 kg N ha^{-1}), soil N accumulation was notably elevated in the upper 0–50 cm, with peaks reaching over 1,500 kg N ha^{-1} between 25–35 cm depths. Medium-input paddocks (170–220 kg N ha^{-1}) showed more distributed N accumulation, with moderate peaks (~1200 kg N ha^{-1}) also observed at 25–35 cm, followed by gradual declines beyond 50 cm. In contrast, low-input paddocks (140–170 kg N ha^{-1}) exhibited the lowest accumulation values across all depths, with peak concentrations (~900 kg N ha^{-1}) observed around 35–45 cm, and little to no retention below 50 cm. Variation in N accumulation was more pronounced at greater depths within each input class, as indicated by higher standard deviations in subsurface layers (>50 cm).

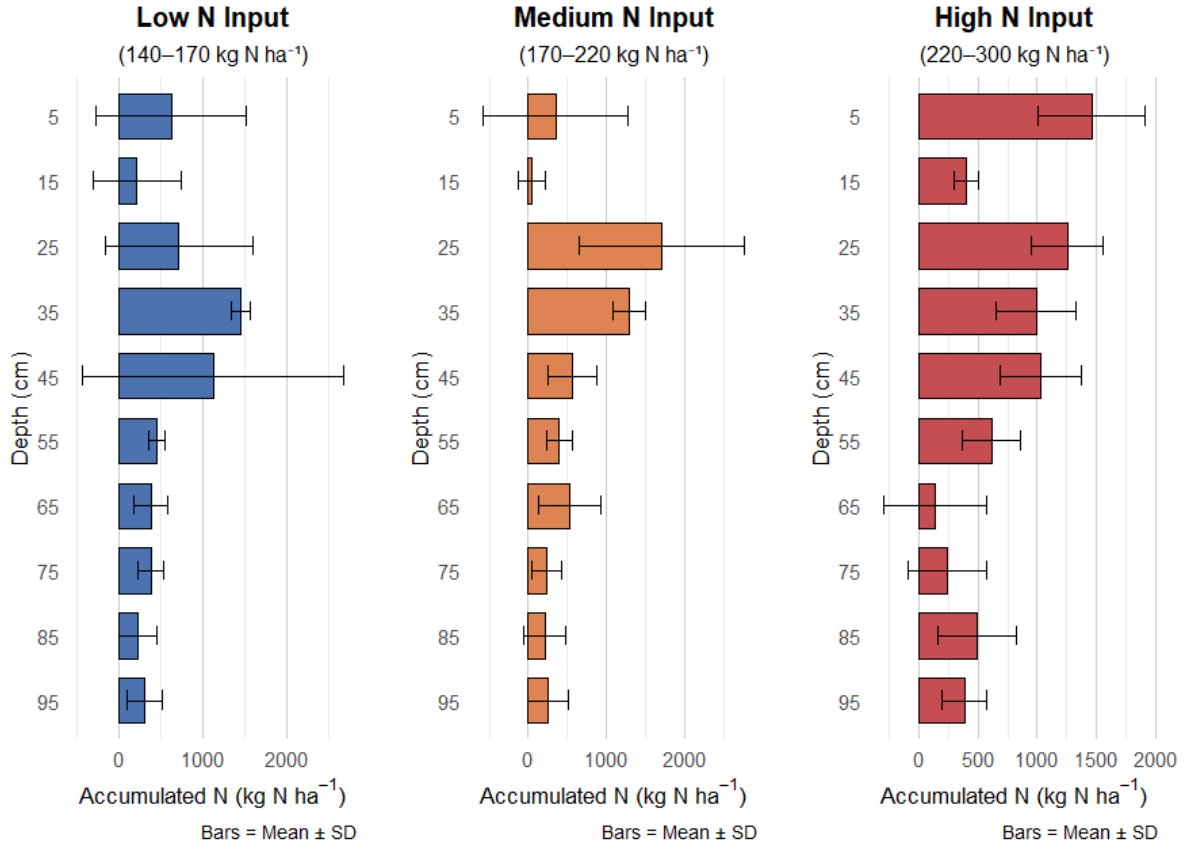


Figure 3- 8 Vertical distribution of accumulated nitrogen (N) across three N input classes (Low: 140–170 kg N ha^{-1}, Medium: 170–220 kg N ha^{-1}, High: 220–300 kg N ha^{-1}) to 1 m soil depth. Each panel represents the mean accumulated N (kg N ha^{-1}) by 10 cm depth increments from the surface to 1 m, calculated across replicate paddocks in each N input class. Bars denote mean values; error bars represent ± standard deviation (SD).

3.4.5 Nitrogen accumulation as a function of current inputs and historical N surplus

Results of the quadrant analysis (Figure 3-9) showed that paddocks in the upper-right quadrant (i.e., characterized by both high historical and high current N applications) consistently exhibited the greatest subsoil N accumulation. Historical N inputs were calculated as the mean annual inputs over a 24-year period (2001–2024), while current N inputs represent the average of the most recent two years (2023–2024). Conversely, paddocks in the lower-left quadrant, with low historical and low current inputs, generally displayed the lowest accumulated N values, but some still retained >6000 kg N ha^{-1}. Paddocks in the upper-left quadrant (high historical but low current inputs) also demonstrated elevated subsoil N accumulation (e.g., 3403 kg N ha^{-1}), while those in the lower-right quadrant (low historical but high current inputs) showed variable outcomes, with N accumulation ranging from moderate to among the highest recorded (e.g., 3,957 kg N ha^{-1} and 4314 kg N ha^{-1}).

Results of the quadrant analysis (Figure 10) showed that paddocks in the upper right quadrant, defined by both high historical (mean 2001 to 2024) and high current (average 2023 to 2024) N inputs, exhibited the highest subsoil N accumulation over the top 50 cm depth. These paddocks also had the largest point sizes, indicating greater measured N accumulation as high as 5825 kg N ha^{-1}.

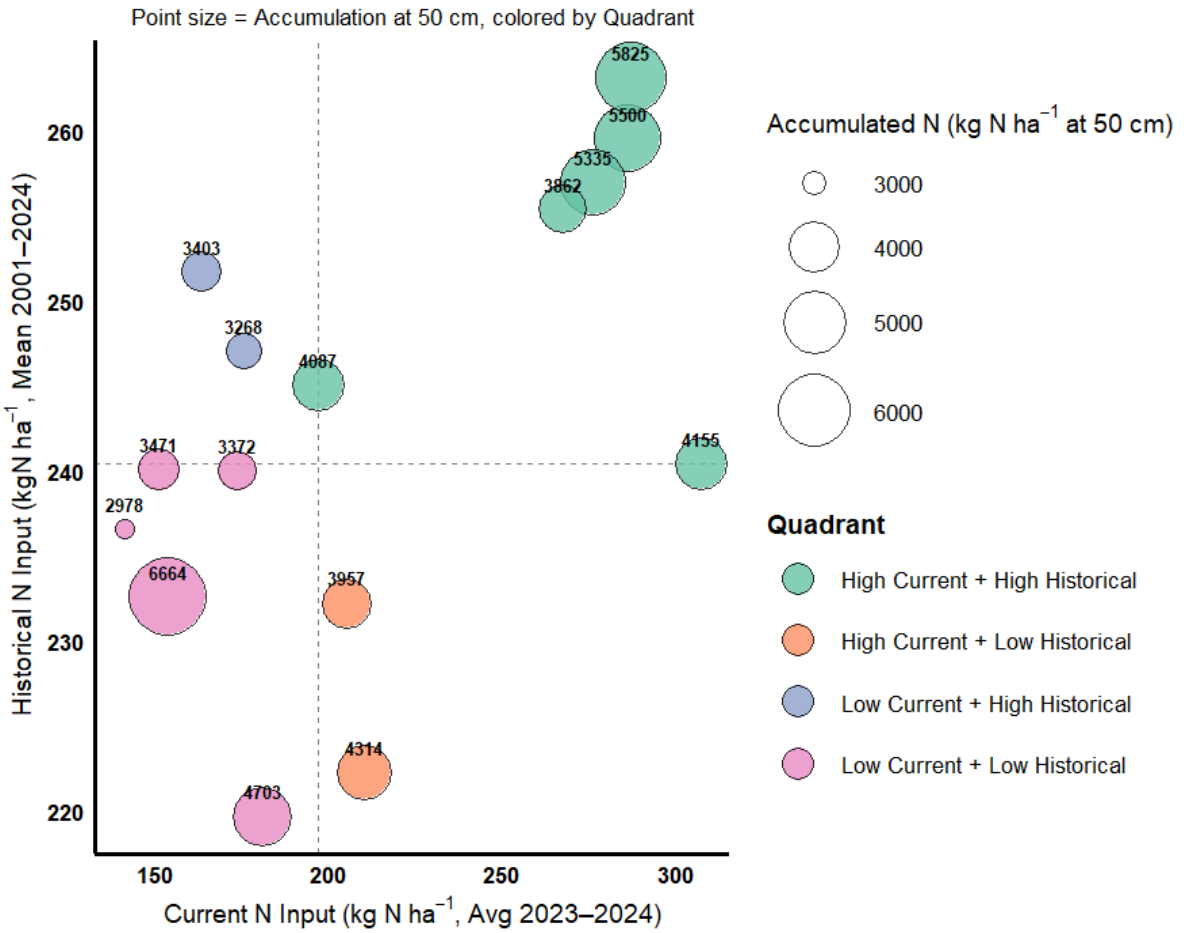


Figure 3- 9 Quadrant analysis figure of nitrogen (N) application and subsoil N accumulation (over the top 50 cm depth) across paddocks. The scatter plot classifies paddocks based on current N input (2023–2024) and historical mean N input (2001–2024). Dashed lines denote mean values used to delineate quadrants: High Current + High Historical (green), High Current + Low Historical (orange), Low Current + High Historical (blue), and Low Current + Low Historical (pink). Point size is scaled to represent the magnitude of N accumulation over the top 50 cm depth (kg N ha^{-1}), and numeric labels denote the cumulative N budget (kg N ha^{-1}) for each paddock. Abbreviations used: Avg=average.

3.5 Discussions

3.5.1 N inputs, outputs, surplus, and soil core overview

Long-term N surplus at the intensive dairy farm site reveals persistent surpluses, particularly in paddocks such as 5.2, 5.3, and 14.3, where grass N output remained modest despite high cumulative N inputs. This imbalance suggests applied N often exceeded plant uptake capacity, promoting continued accumulation or loss through leaching and gaseous pathways. These patterns mirror trends observed in other intensive pasture systems

(McDonald, 2014; Forrestal et al., 2025). N inputs, mainly from fertilizers, slurry, and DSW, varied widely across paddocks, influenced by management history. While paddocks like 13.2 and 9.2 exhibited low N surplus, others like 6.2 and 14.3 showed diminishing returns at high application rates, consistent with concerns about soil N saturation (Weitzman et al., 2022). We acknowledge that the farm N surplus estimates presented here are subject to uncertainty due to variability in input data and potential overlaps in defining outputs. While a full quantitative uncertainty analysis was not feasible given available records, the consistency between the N surplus estimates, measured soil N accumulation, and observed groundwater NO_3-N concentrations provide confidence in the robustness of the results. Vertical TN distribution highlights surface enrichment (0–30 cm) due to repeated surface applications, with TN in the top 10 cm reaching 3.54 g kg^{-1}, consistent with Kilworth soil subclasses 26–28 (McDonald, 2014). PMN was also concentrated in these layers, reflecting active mineralization zones. Deeper TN accumulation (50–75 cm), particularly in Paddocks 5 and 6, signals legacy N migration beyond the rooting zone, with implications for long-term leaching risk (Leal et al., 2025; Miao et al., 2023). These findings echo Page et al. (2003) and Weier & MacRae (1993), who observed that deep NO_3^- accumulation stems from surface leaching rather than in situ mineralization. TN accumulation varied twofold (5,000–9,300 kg N ha^{-1}) under similar annual inputs (~200 kg N ha^{-1} yr^{-1}), underlining NUE inefficiencies and the influence of soil structure, microbial activity, and legacy management (Nakagawa et al., 2021).

3.5.2 Groundwater trends and N surplus reduction

Groundwater monitoring confirmed that NO_3-N dominates DIN in this free-draining system, comprising 84–90 % of inorganic N (Jahangir et al., 2013). Limited denitrification due to deep water tables and high oxygen levels restricts NO_3^- attenuation (Jahangir et al., 2012). Denitrification was not a major N loss pathway at this well-drained study site. Jahangir et al. (2013) reported from a nearby research site that only ~4 % of delivered N was removed via denitrification, resulting in groundwater NO_3-N concentrations of 12–15 mg L^{-1}. The low extent of denitrification reflects limited reaction progress in these soils, leading to elevated net NO_3-N concentrations. NO_3^- concentrations declined sharply after 2010, aligning with improved NUE and reduced N inputs, this decline has continued over the years but at a much slower rate. Concentrations now remain below the EU MAC of 11.3 mg L^{-1} and leaching loads have dropped from over 70 kg N ha^{-1} in the early 2000s to below 30 kg N ha^{-1} by 2024, which is the target for good water quality (Lakhanpal et al., 2025). Mann-Kendall tests confirmed

significant declining trends in NO_3-N for both 2002–2016 and 2022–2024, with Sen's slopes showing declines of –0.524 mg L^{-1} yr^{-1} (annual) and –0.060 to –0.266 mg L^{-1} mo^{-1} (monthly). Despite improvements, legacy N continues to leach slowly. Boreholes 4, 5, and 8 exceeded MAC in early years, reflecting historical management variability. This supports previous work on time lags between reduced N inputs and groundwater response (Goulding et al., 2000; Hooker et al., 2008; Premrov et al., 2014). Paddocks with high cumulative inputs, such as 6.2 and 14.3, still show elevated leaching risk. Historic inputs >400 kg N ha^{-1} yr^{-1}, as reported by Dillon et al. (1995), likely contribute to current subsurface N stores.

3.5.3 Soil cores, groundwater links, and the case for legacy N monitoring

2024 soil coring provided essential insight into legacy N stores influencing groundwater trends. PMN and TN stratification revealed that surface inputs shaped upper 30 cm profiles, while deeper enrichment reflected long-term accumulation. Groundwater NO_3-N (11.1–73.6 kg N ha^{-1} yr^{-1}) match those observed in Huebsch et al. (2013) and McAleer et al. (2017) under similar high-input grassland systems. Quadrant analysis confirmed that paddocks with high historical and high current inputs (e.g., 5.1, 6.2) had the highest subsoil N accumulation (4,155–5,825 kg N ha^{-1}). Paddocks with low historical and low current inputs showed the lowest accumulation (<3,000 kg N ha^{-1}), while those with high historical but reduced current inputs retained substantial N (e.g., 3,403 kg N ha^{-1}). Especially, one paddock classified as having low historical and current inputs showed unexpectedly high accumulation (>6,000 kg N ha^{-1}), which may reflect high legacy surpluses or slower N turnover rates. These patterns highlight persistent legacy effects and variability due to soil type, rooting depth, and hydrology (Yang et al., 2020; Ren et al., 2023). Jahangir et al. (2013) and O'Connell et al. (2004) describe soil and water properties in this area that support slow vertical movement and minimal denitrification. Variability in subsoil accumulation, even in similarly managed paddocks (e.g., 13.3 vs. 14.2), emphasizes the need for site-specific mitigation. Episodic weather events (e.g., drought followed by rewetting) further modulate NO_3^- mobility and loss potential (Tyson et al., 1997). Legacy N clearly plays a central role in sustaining NO_3^- concentrations even after input reductions. This underscores the need for deep soil sampling, fine-scale nutrient management, and long-term monitoring frameworks. A one-size-fits-all strategy is insufficient; targeted interventions tailored to site history and subsurface data are necessary for improving NUE and protecting groundwater.

3.5.4 Implications of findings and future requirements

Our findings on subsoil N accumulation reinforce growing evidence that agricultural soils act as long-term reservoirs of legacy N, contributing to delayed improvements in groundwater quality despite reductions in surface inputs. Van Meter et al. (2016) demonstrated that substantial N can accumulate in subsurface layers over decades, with cumulative net N retention in some U.S. basins reaching on the order of 1400 kg N ha^{-1} over ~50 years (i.e., ~28 kg N ha^{-1} yr^{-1}) as reported in their broad mass-balance estimates. This dynamic aligns with our findings in high-input paddocks, where deep N accumulation below 50 cm persists despite recent input reductions. Nakagawa et al. (2021) reported extremely large vadose-zone NO_3-N inventories in intensively managed orchards, ranging from 15,322 to 26,583 kg N ha^{-1}, underscoring the magnitude of legacy pools that can persist under long fertilization histories. Similarly, Liu et al. (2022) quantified NO_3-N stocks of 118 to 6,303 kg N ha^{-1} in the deep vadose zone of the North China Plain, with accumulation increasing at an average rate of ~157 kg N ha^{-1} per meter of depth; in orchard systems they observed up to 7,113 kg N ha^{-1} within the 0–10 m profile. Zhou et al. (2016) demonstrated that semi-humid croplands in China also store large amounts of NO_3-N in the 0–4 m profile, with accumulations of ~453 kg N ha^{-1} under wheat, 749 kg N ha^{-1} under maize, 1,191–1,269 kg N ha^{-1} under vegetables, and up to 2,155 kg N ha^{-1} in orchards; comparable amounts were detected below 4 m. At the catchment scale, Sarrazin et al. (2022) further showed that legacy storage is shaped not only by inputs but also by subsurface hydrology, delaying N export for decades. Zhou et al. (2016) showed semi-humid croplands in China can accumulate ~453 to 2,155 kg N ha^{-1} in the 0–4 m soil profile (depending on crop type), with comparable amounts potentially deeper. At the catchment scale, Sarrazin et al. (2022) used a legacy-driven model for the Weser Basin (1960–2015) and estimated a total legacy buildup of ~491 kg N ha^{-1}; of that, ~230–780 kg N ha^{-1} is stored in the soil "source zone" and ~10–105 kg N ha^{-1} in the subsurface (hydrologic) legacy domain.

These studies demonstrate that cumulative N inputs, even when short-term balances appear neutral, can result in substantial subsoil accumulation, typically on the order of 10^0–10^3 kg N ha^{-1} over decadal timescales. This is consistent with Van Meter et al.'s estimate of ~1,400 kg N ha^{-1} legacy storage. It also explains, the intensive dairy system examined in this study showed markedly greater accumulation, exceeding 5,000 kg N ha^{-1} in well-drained, high-input intensive dairy farming system. To address this, improved N stewardship is essential. While strategies like reduced synthetic N use, nitrification inhibitors, and restricted grazing

(Lakhanpal et al., 2025) are promising, our results highlight the need for site-specific approaches. Soil drainage, climate, and management history must inform planning; broad generalizations are insufficient.

Recent national assessments by the Environmental Protection Agency (EPA, 2024) strongly reinforce the findings of this study. While groundwater NO_3-N concentrations may appear to be improving or stabilizing, further reductions are increasingly constrained by persistent legacy N stores. Despite improvements in water quality from 2023 to 2024, NO_3^- concentrations remain elevated in the southeast of Ireland, including in areas under intensive dairy management. Although current mitigation strategies appear effective in reducing NO_3-N loads to below the commonly referenced target of 30 kg N ha^{-1} yr^{-1}, concentrations in groundwater, while mostly beneath the EU drinking water standard of 11.3 mg NO_3-N L^{-1}, still frequently exceed the ecological guideline of 8.5 mg NO_3-N L^{-1}, which is associated with negative impacts on sensitive estuarine ecosystems. This observation suggests that legacy N acts as a persistent source, effectively topping up groundwater NO_3^- concentrations and maintaining levels above the ecological threshold even as recent inputs decline. In this way, legacy N is not only prolonging the recovery trajectory but also preventing full compliance with stricter ecological targets. It underscores the narrow margin between achieving the drinking water standard and achieving ecologically protective concentrations. As policy emphasis increasingly shifts toward the latter (for example, 8.5 mg NO_3^--N L^{-1} in groundwater or 2.5 mg L^{-1} in estuaries), the importance of quantifying, managing, and eventually depleting the legacy N pool becomes paramount. These findings suggest that further regulatory measures focused solely on reducing current inputs may yield diminishing returns unless they are accompanied by strategies that explicitly target the reduction or stabilization of subsurface legacy N stores.

Future research should explore a broader range of agro-climatic zones, incorporate long-term fertilizer histories, and examine soil layers beyond 1 m to assess deep N storage. The integration of deep-rooted, biologically active species such as plantain and chicory could access subsoil N stores and reduce surface fertilizer needs. In-field mineralization and leaching assessments are also needed to complement lab-based findings. Risch et al. (2017) showed that actual mineralization in the field may exceed lab estimates, emphasizing the need for site-specific data. Collectively, these findings support long-term monitoring and adaptive nutrient

management strategies that consider both legacy effects and current practices to reduce environmental losses and improve fertilizer use efficiency in intensive grassland systems.

3.6 Conclusions

This study shows that long term N loading in intensively managed dairy systems has led to persistent subsurface N accumulation and delayed improvements in groundwater quality. Denitrification contributed minimally at this well-drained site, resulting in persistently elevated groundwater NO_3-N concentrations. Groundwater monitoring confirmed that NO_3-N is the dominant form of dissolved inorganic N, with concentrations frequently exceeding safe drinking thresholds in the early 2000s. Limited denitrification in well drained soils restricted subsurface NO_3^- removal. While NO_3-N levels declined after 2010, the trend plateaued in recent years, reflecting legacy effects and time lags in NO_3^- transport. Paddock scale N surplus revealed sustained surpluses across the farm, with cumulative inputs exceeding outputs by up to 255 kg N ha^{-1} yr^{-1} in high input paddocks. While recent years showed narrowing surplus, annual averages confirmed continued leaching risks. Leaching declined over time, approaching the 30 kg N ha^{-1} target in the most recent period. Soil coring to 1 meter depth identified substantial subsoil accumulation of TN and PMN, especially below 30 to 50 cm. These concentrations were strongly correlated with long term N inputs, highlighting the influence of historical loading. Vertical stratification of TN and PMN indicates that legacy N persists independently of recent management changes. Quadrant analysis of historical and current N inputs showed that paddocks with sustained high inputs had the highest subsoil N accumulation. Even where current inputs were reduced, elevated subsoil N levels persisted, underscoring the long-term impact of legacy N. Variability in accumulation patterns also pointed to the role of soil properties and hydrology in modulating outcomes. Together, these findings highlight the importance of integrating legacy N considerations into nutrient management. Improving NUE, reducing surpluses, and targeting subsoil processes are critical for long term sustainability. Continued monitoring, deep soil and in-field mineralization assessments and site-specific strategies will be essential to address both historical impacts and future risks.

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Chapter 4

Legacy Nitrogen Accumulation and Mineralization in Two Grassland Soil Profiles with Contrasting Drainage, Texture, and Hydrological Characteristics

This chapter is a mirror of the following article in preparation.

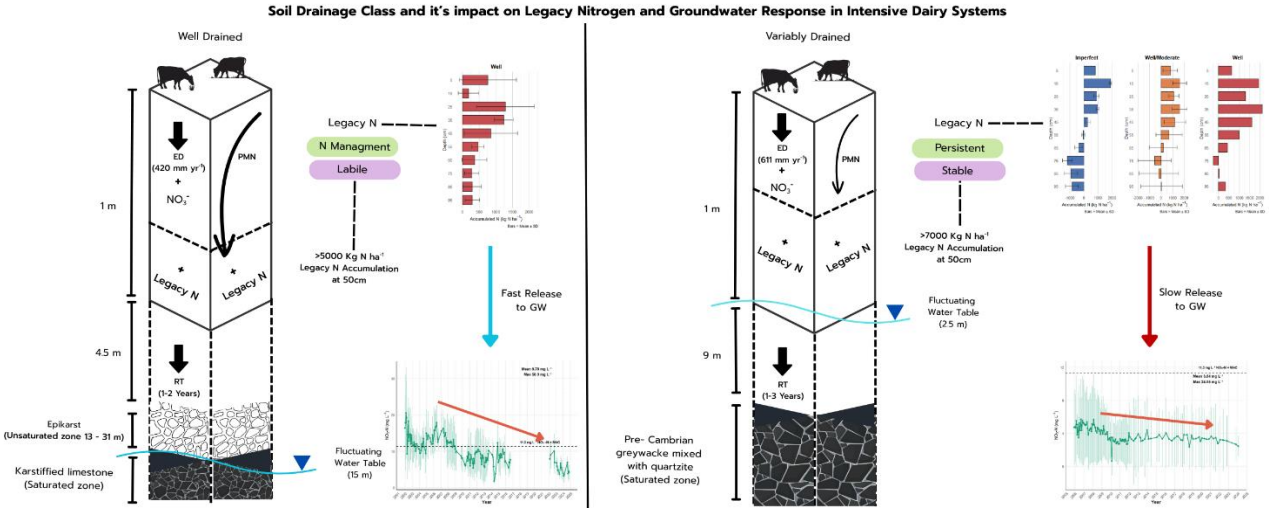
Lakhanpal, G., Fenton, O., Basu, N.B., Horan, B., Richards, K.G., Cardenas-Alvarez, G., Lopez-Sangil, L., Murphy J. and Condon, T., Legacy Nitrogen Accumulation and Mineralization in two Grassland Soil Profiles with Contrasting Drainage, Texture, and Hydrological Characteristics.

4.1 Overview

Intensive grass-based dairy systems generate large nitrogen (N) surpluses, yet the fate of this surplus and its accumulation in soils remain poorly quantified across different drainage and hydrological settings. This study combines 18–24 years of farm-scale N budget, groundwater monitoring, and soil profile data from two long-term research farms in Ireland—Curtins, a well-drained site underlain by karstic limestone with low denitrification potential, and Johnstown Castle (JC), a variably drained site with finer-textured soils, shallow groundwater, and moderate-to-high denitrification potential. At Curtins, historical surpluses exceeding 300 kg N ha^{-1} yr^{-1} created substantial but labile legacy pools (3,000–6,500 kg N ha^{-1} in the upper 50 cm), leading to groundwater NO_3-N concentrations above 50 mg L^{-1} in the early 2000s. Concentrations declined below the EU drinking-water threshold (11.3 mg L^{-1}) by 2024 following major input reductions, reflecting the rapid responsiveness of well-drained, low-denitrification systems once surpluses are curtailed. In contrast, at JC, lower surpluses (~100–150 kg N ha^{-1} yr^{-1}) still produced larger soil accumulations (4,000–11,000 kg N ha^{-1}), driven by higher silt and clay content, shallow fluctuating water tables, and denitrification-mediated buffering that limited leaching losses (mean ≈ 3.6 mg L^{-1} NO_3-N) but promoted long-term N retention and slower mineralization. Even the drier, well-drained paddocks at JC used more intensively for grazing showed higher manure-derived inputs and deeper N accumulation. These contrasting outcomes demonstrate that N accumulation is a multifactor process governed

not only by drainage but also by soil texture, hydrogeology, groundwater depth, and denitrification potential. Well-drained systems potentially act as “fast transmitters,” prone to leaching but capable of rapid recovery when inputs decline, whereas variably drained systems potentially act as “slow retainers,” protecting groundwater in the short term but storing persistent legacy pools that delay long-term recovery. From a policy perspective, these findings highlight that uniform mitigation timelines under the EU Nitrates and Water Framework Directives are unrealistic. Site-specific approaches that incorporate legacy N storage, soil drainage class, and lagged recovery dynamics are essential to design credible mitigation strategies, realistic environmental targets, and adaptive nutrient management frameworks for Ireland’s intensive dairy sector.

Graphical Overview



4.2 Introduction

Human activity has profoundly altered the global nitrogen (N) cycle, driving sharp increases in reactive nitrogen (Nr) availability and associated pressures on terrestrial and aquatic environments (Galloway et al., 2008; Fowler et al., 2013). Intensification of agriculture, and in particular pasture-based dairy systems, has been a major contributor to excess N surpluses (Lassaletta et al., 2014; Oenema et al., 2009). While these systems provide important economic outputs, they are characterized by low nitrogen use efficiency (NUE), with a large proportion of inputs susceptible to loss via leaching, gaseous emissions, or legacy accumulation in soils (Zhang et al., 2015; Velthof et al., 2014). The resulting diffuse pollution threatens groundwater quality, accelerates eutrophication in rivers and coastal waters, and contributes to

climate forcing through nitrous oxide (N_2O) emissions (Gruber & Galloway, 2008; Winiwarter et al., 2018).

Within dairy systems, the distribution and fate of N surpluses are strongly conditioned by both static site factors such as soil type, texture, drainage class, and hydrogeology, and dynamic influences such as rainfall patterns, effective rainfall, water table depth, and management intensity (Fenton et al., 2011; Baily et al., 2011; Jahangir et al., 2012). Previous research in Ireland has shown that changes in fertilizer inputs and grazing management can improve NUE and reduce average groundwater nitrate (NO_3^-) concentrations, yet the extent of these improvements is highly site-dependent (Treacy et al., 2008; Huebsch et al., 2013; Fenton et al., 2017). For instance, deep, well-drained soils with permeable geology and fluctuating water tables are prone to rapid NO_3^- accumulation and leaching due to high mineralization and nitrification potential, with limited capacity for denitrification (Jahangir et al., 2013; Clagnan et al., 2019). In contrast, poorly drained soils with shallow water tables are more likely to retain N in ammonium (NH_4^+) form under anaerobic conditions but can also act as zones of NO_3^- reduction (Jahangir et al., 2012; Baily et al., 2011). These contrasting drainage classes create divergent pathways for N storage, transformation, and legacy accumulation, highlighting the need to explicitly consider both hydrogeology and drainage when evaluating long-term N dynamics (Van Meter et al., 2016; Ascott et al., 2017; Ascott et al., 2021).

At a policy level, these dynamics have direct implications for the European Union's (EU) Water Framework Directive (WFD; 2000/60/EC), which aims for all water bodies to achieve at least "good status." In Ireland, agriculture remains the most widespread pressure on water quality, with over 60 % of at-risk water bodies impacted by nutrient pollution (EPA, 2024). The Nitrates Directive (ND) (91/676/EEC) and the associated National Action Program (NAP) have sought to reduce N losses through improved nutrient management and stocking controls, yet time lags from legacy N storage complicate progress toward meeting water quality targets (Acott et al., 2021; Van Meter et al., 2016). This underlines the need for research that links farm-level N surpluses, soil drainage classes, and hydrogeological settings to long-term groundwater trajectories, providing an evidence base for both farmers and regulators to anticipate delayed responses to policy interventions.

Legacy N in soils, consisting of both organic matter and inorganic pools that have accumulated under past management, represents a latent but poorly understood source of diffuse pollution. Although research often focuses on surface soils, evidence suggests that

subsoils can store substantial quantities of reactive N and actively contribute to mineralization and nitrification. Page et al. (2003) showed that Vertosols can accumulate NH_4^+ through limited nitrification and subsoil ammonification, while Weier & MacRae (1993) demonstrated that mineralization and nitrification processes extend to depths of 1.8 m under pasture. More recently, Risch et al. (2019) highlighted the variability of mineralization rates globally, with soil texture, temperature, and microbial biomass identified as key drivers, and laboratory-based estimates often overestimating field mineralization. However, systematic evaluation of vertical N soil profiles under contrasting hydrological settings in intensive grassland systems remains scarce. In particular, little is known about how soil drainage class and hydrogeological context shape the storage, turnover, and mobilization of legacy N at depth. Addressing this gap is critical for predicting long-term risks to groundwater quality and for designing nutrient management strategies that account for both surface and subsurface legacy pools.

Recent work on well-drained intensive dairy farms has demonstrated how legacy N pools can be quantified using multi-decadal fine-scale farm management records, soil coring, and long-term groundwater NO_3^- data. Lakhanpal et al. (2025) combined these approaches to reconstruct site-level N surpluses and evaluate how historical agricultural practices have shaped groundwater quality trajectories. Their findings demonstrated that legacy N storage within the soil profile plays a critical role in determining the lag between improved management practices and observable reductions in groundwater NO_3^- concentrations. This insight motivated the next stage of investigation: moving beyond quantification of the legacy N pool to identify the static and dynamic factors that control its size, stability, and mobilization potential. By addressing this, it becomes possible not only to assess how much reactive N is stored belowground, but also to understand what governs its persistence and release.

The present study builds on this foundation by evaluating vertical N profiles (0–1 m) and quantifying N transformations contributing to legacy accumulation across two intensively managed dairy farms in Ireland with contrasting drainage and hydrogeological settings. Importantly, we also developed multi-decadal N surplus estimates for both sites to contextualize current soil and subsoil N storage within the long-term balance between inputs and outputs. The specific objectives were to: (i) quantify net mineralization, potentially mineralizable nitrogen (PMN), and total N accumulation across soil profiles; (ii) evaluate how hydrological contrasts, including shallow versus deep water tables and poorly versus well-drained soils, influence subsoil N transformations and legacy storage; and (iii) integrate these findings with site-specific N surplus trajectories to assess the drivers of legacy N accumulation.

We hypothesized that poorly drained soils with shallow water tables would accumulate more NH_4^+ due to inhibited nitrification, while well-drained soils would support greater mineralization and nitrification at depth. Furthermore, we expected laboratory PMN assays to overestimate field mineralization, particularly under poorly drained conditions. By combining field measurements, laboratory incubations, and multi-decadal surplus calculations, this study provides new insight into how site conditions shape legacy N pools in grassland systems and their long-term implications for groundwater quality. Importantly, while current nutrient management frameworks under the ND guide farmers, policymakers, and regulators, none explicitly incorporate legacy N. This study therefore contributes to emerging international discussions by Ascott et al., 2021 on how legacy N must be recognized and managed within future agri-environmental policy to ensure that groundwater quality targets are achieved.

At both EU and national levels, policy and regulatory frameworks must begin to explicitly acknowledge the role of legacy N and associated time lags in water quality recovery. Beyond existing PIP and NIP mapping tools provided by the EPA, an additional layer is required to represent how legacy N manifests differently across systems; with overland flow and artificial drainage pathways driving phosphorus and shallow NH_4^+ losses, while infiltration-dominated systems remain vulnerable to NO_3^- leaching. This builds on the soil drainage class framework of Clagnan et al. (2019) but extends it by incorporating the delayed impacts of legacy N that are not yet reflected in current nutrient management strategies.

4.3 Materials and Methods

4.3.1 Site Description

4.3.1.1 Curtins Site

The Curtins study site is a 48.1 ha intensive dairy research farm situated on a lowland limestone plateau in southern Ireland (Teagasc Dairy Production Centre, Fermoy, Co. Cork; 52°10′N, 8°15′W). The landscape is characteristically flat with negligible overland flow and lies up-gradient of the Funshion River, in proximity to a public water supply well, and down-gradient of the River Blackwater (Figure 4-1). A dedicated on-site meteorological (MET) station provides hourly weather records, which have been supplemented with long-term climate data. Thirty-year averages (1982–2011) reported by Huebsch et al. (2013) indicate a mean annual rainfall of 1022 mm, exceeding the 11-year mean of 996 mm (2001–2011). Rainfall

distribution is strongly seasonal, with maxima in October (112 mm) and minima in July (64 mm).

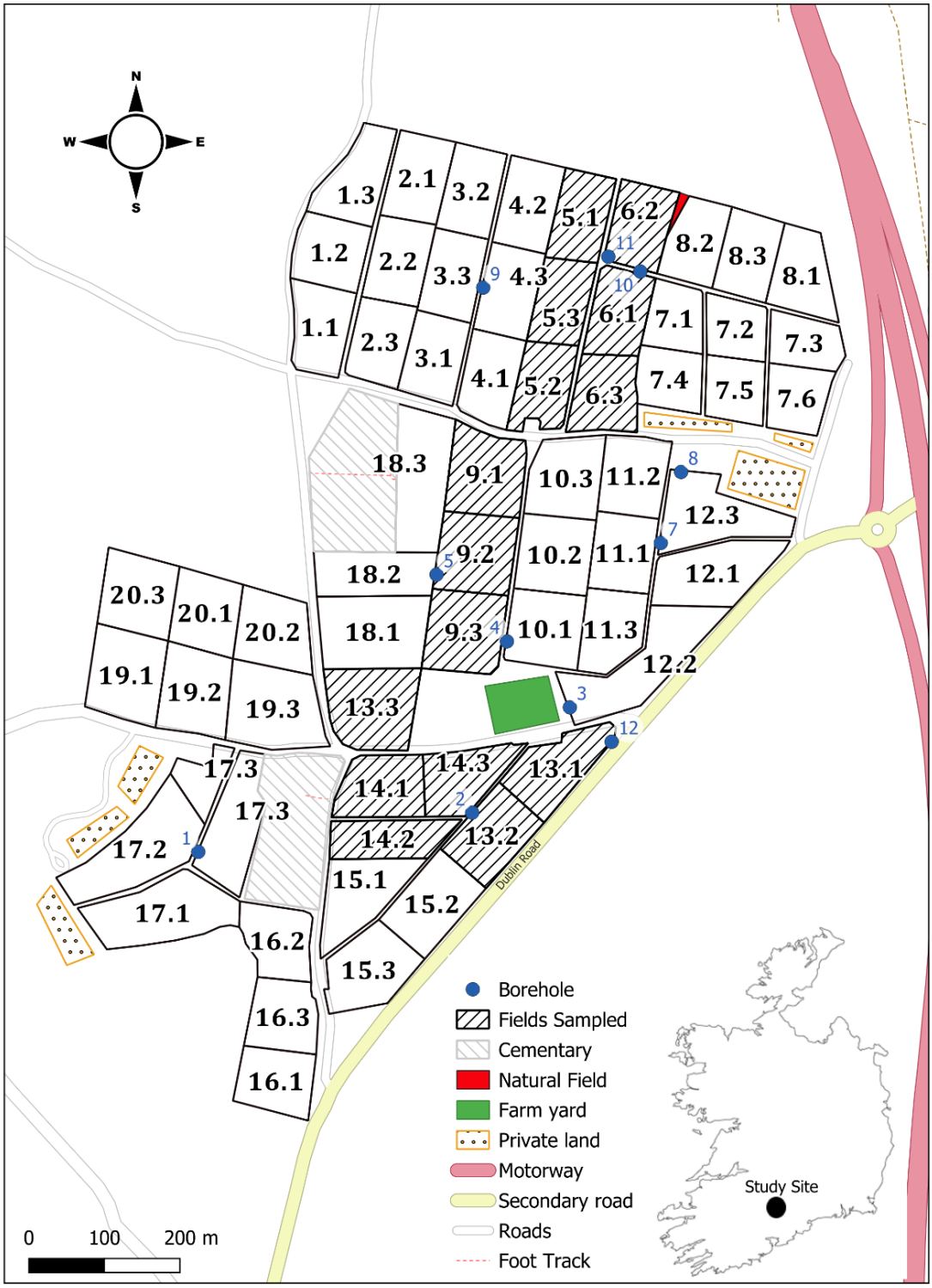


Figure 4- 1: Study site 1 Curtins, location and figure with area (ha) of each field and location of natural field (i.e., control) and sampled fields within the study.

Weather data used in this study were primarily obtained from Met Éireann's online database. The nearest official station, Moorepark (station number 575), located ~5 km from Curtins, has provided hourly, daily, and monthly climate records since 2003. Daily datasets of rainfall (mm), maximum and minimum air temperature (°C), potential evapotranspiration (mm), and evaporation (mm) were compiled for the period 2004–2024. Soil moisture deficit (SMD) values for well-drained soils were sourced from Met Éireann for 2011–2024. To extend the record, daily SMD estimates for 2007–2010 were reconstructed using the hybrid model of Schulte et al. (2005), which requires inputs of rainfall, global radiation, wind speed, and maximum and minimum temperatures. Due to incomplete climate records prior to 2007, the model could not be reliably applied to earlier years.

Curtins research farm has been extensively studied in terms of soil, subsoil, geology, and the interactions between farming practices and NO_3-N transport in groundwater within and beyond the site. The dominant soil type is a freely drained acid brown earth (Haplic Cambisol) developed from mixed sandstone–limestone glacial till (Kramers et al., 2009). Soil depth varies between 0 and 4.5 m (Bartley & Johnston, 2006). The surface A horizon consists of 53 % sand, 31 % silt, and 16 % clay, with a dry bulk density of 1.1 g cm^{-3} and a total porosity of 52 %. Preferential flow pathways are largely restricted to the A–B horizons (Kramers et al., 2009). The subsoil is underlain by Waulsortian limestone, a karstified formation that occurs at an average depth of 2.5 m (Bartley & Johnston, 2006). This formation, which represents about 1.7 % of Ireland's land area (Ryan et al., 2006), is more highly karstified and less bedded than other limestone units such as the Ballysteen Formation (GSI, 2000).

For this study, five paddocks (5, 6, 9, 14, and 15) were selected. Each paddock was subdivided into three sub-paddocks (e.g., 5.1, 5.2, 5.3), reflecting differences in N management practices. These included variation in fertilizer input, grazing intensity (expressed as grazing days per hectare), application of slurry and dairy soiled water (DSW), and changes in sward composition over time. Detailed farm activity records from 2001 to 2024 were available for these paddocks and were used to construct N budgets. A nearby unmanaged area served as a control to estimate background N accumulation.

Geophysical investigations provided additional insights into subsurface characteristics and groundwater flow pathways. Surveys of surface conductivity (to 6 m depth) and resistivity profiling (to 50 m depth) revealed both vertical and lateral variability in overburden composition, bedrock depth, and lithological properties (Huebsch et al., 2013). Electrical

conductivity (EC) values ranged from 11–14 mS m^{-1} (low) to 17–20 mS m^{-1} (high), corresponding to differences in subsurface material. Elevated EC values were associated with collapse features filled with fine-grained sediments such as silt and clay, linked to the karstified limestone bedrock. These surveys also identified a conduit system trending northwest to southeast across the central part of the farm. Resistivity data indicated that karst features, including conduits, fractures, and air-filled cavities, are widespread and exert strong control on groundwater flow direction. The soils at Curtins are predominantly well-drained, which minimizes opportunities for denitrification and results in a high leaching potential. Combined with the karstified limestone subsoil, this hydrogeological setting makes the site highly vulnerable to NO_3-N transfer to groundwater, with limited natural attenuation capacity.

Groundwater monitoring boreholes (BHs) (Table A.1) have been installed across the site (locations summarized in Huebsch et al., 2013). Tracer studies demonstrated complex connectivity between BHs, with BH7 linked to BH4, BH5, and discharging in the direction of BH9. In contrast, BH8 showed no connection to BH7 or to BH10–11, although BH10 and BH11 were hydraulically connected. Similarly, BH2, BH3, and BH12 formed another connected group (Bartley et al., 2004). This pattern of connectivity has important implications for N transport, as management practices and N surpluses in one part of the farm can directly influence groundwater NO_3-N concentrations at multiple monitoring points.

4.3.1.2 Johnstown Castle Site

The Johnstown Castle research farm is located in County Wexford, southeast Ireland (52°17′30″ N, 6°29′50″ W). The site covers 72.95 ha (Figure 4-2) and is situated in a cool maritime climate, with a long-term average annual temperature of 10 °C and mean annual precipitation of 1038 mm (1981–2011, Rosslare synoptic station). Rainfall is most intense between September and November. An on-site meteorological station records daily weather variables, which are used in hydrological modelling to estimate soil moisture deficits and effective drainage.

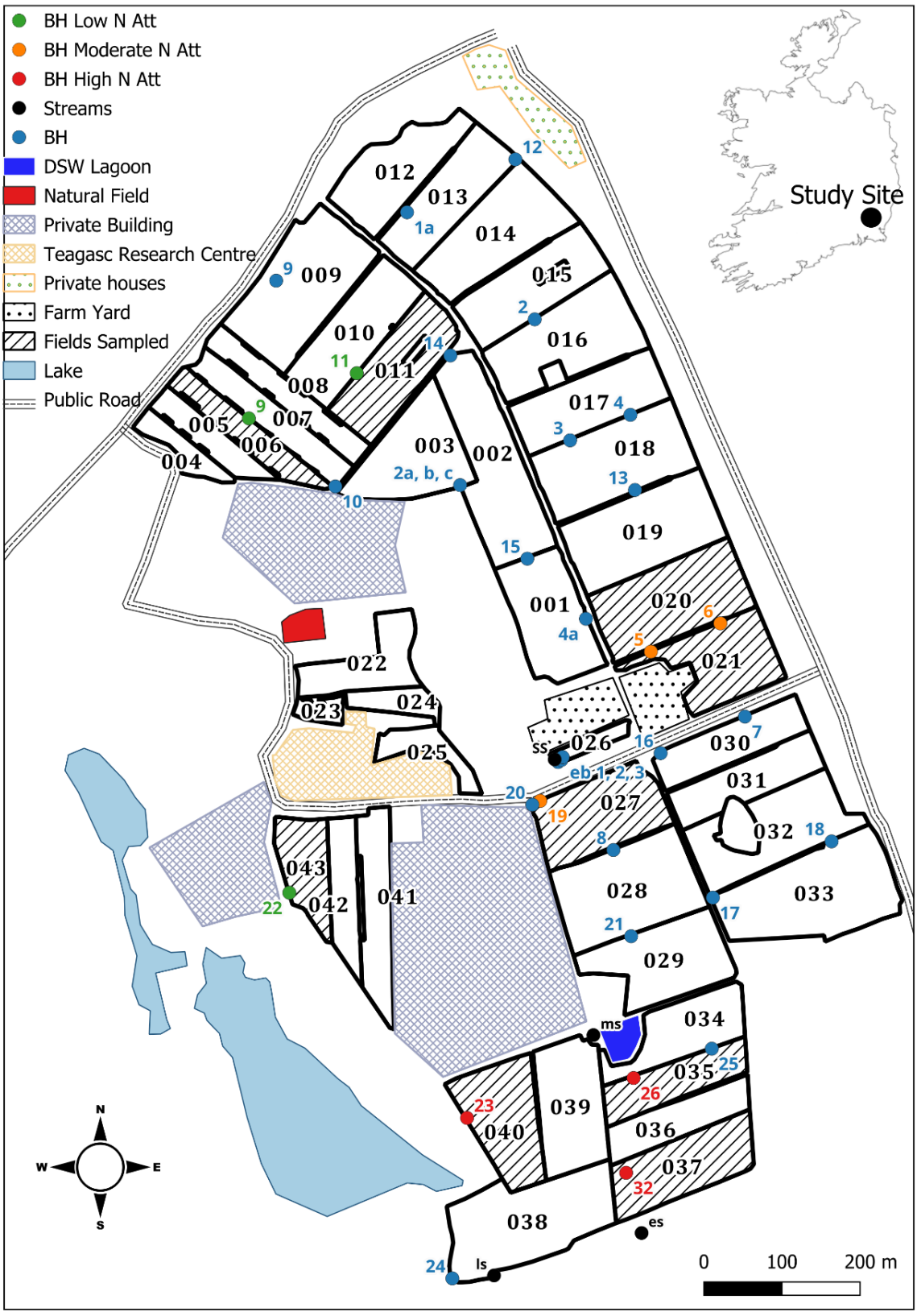


Figure 4- 2: Study site 2 Johnstown Castle, location and figure with area (ha) of each field and location of natural field (i.e., control) and sampled fields within the study.

Abbreviations used: N: Nitrogen, BH: Borehole, DSW: Dairy Soiled Water, Att: Attenuation.

Soils are predominantly fine loam to clay loam, classified as Brown Earths and Gleyic Cambisols derived from Irish Sea till (Gardiner & Ryan, 1969). Small pockets of sandy soils are also present. Drainage classes range from well-drained to poorly drained (Figure 4-3), creating strong spatial contrasts in hydrological response across the site (Jahangir et al., 2013b). Subsoils are of moderate permeability, with saturated hydraulic conductivity values between 5×10^{-8} and 5×10^{-4} m s^{-1}, but are interspersed with gravel and sand lenses that locally enhance drainage (Fenton et al., 2009; Jahangir et al., 2013b). Soils at Johnstown Castle span imperfect, moderately well-drained, and well-drained drainage classes. This heterogeneity provides greater potential for denitrification, particularly in imperfectly drained areas with shallow groundwater tables, thereby buffering NO_3-N transfer to groundwater but promoting long-term N storage in soils.

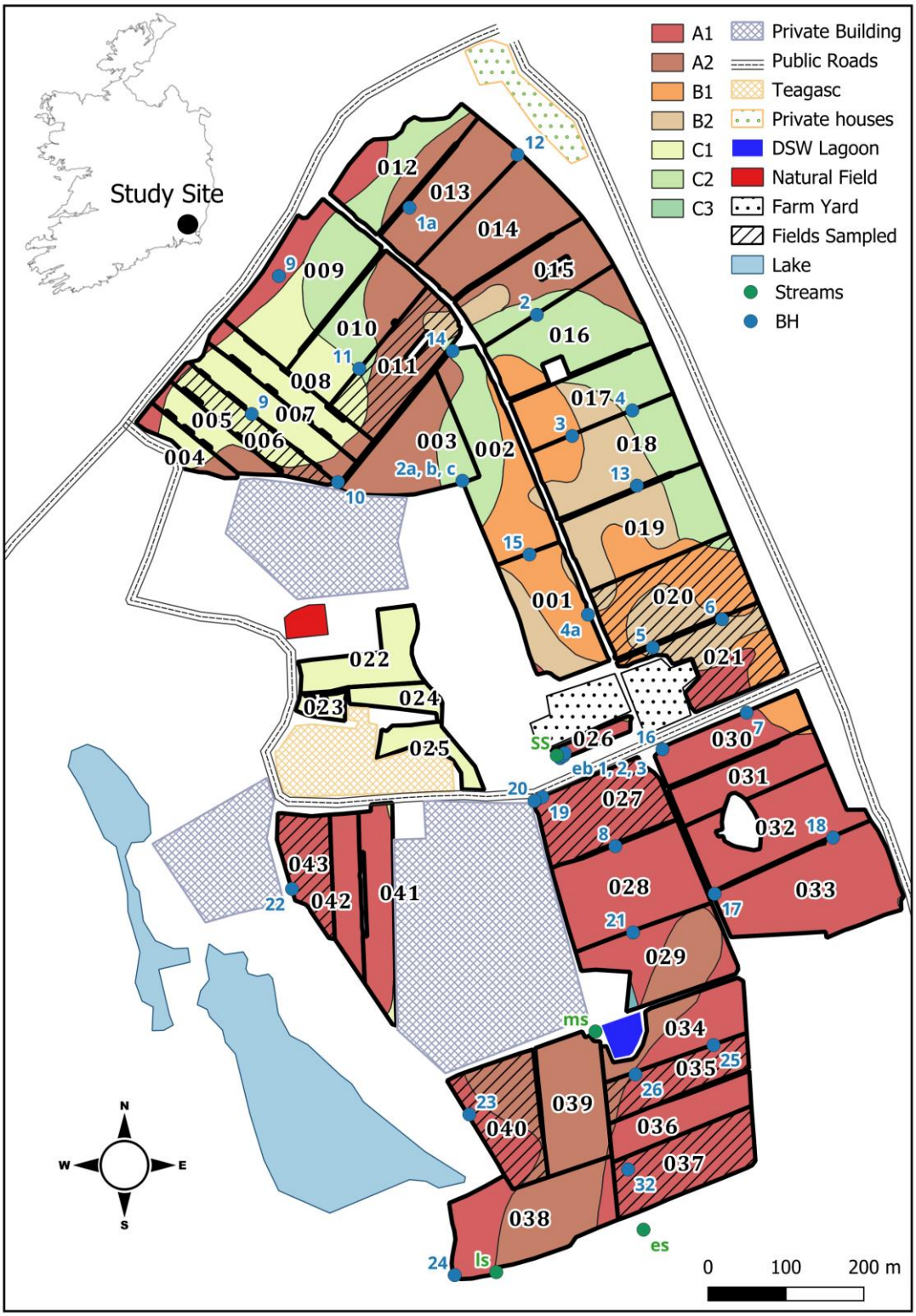


Figure 4- 3: Study site 2 Johnstown Castle, soil drainage class and sample paddocks with boreholes (BH) within the study site.

Hydrogeological investigations indicate that water residence times in the unsaturated zone vary between 1 and 3 years, with a mean of 1.5 years occurring 85 % of the time (Baily et al., 2011). This introduces a lag between management interventions at the surface, the

storage and mineralization of N in the soil profile, and subsequent leaching to groundwater. Groundwater levels fluctuate seasonally, averaging 2.8 ± 1.7 m below ground level (bgl) between 2005 and 2014. The deepest water tables occur in summer (3.0 m bgl), while the shallowest are recorded in winter (2.3 m bgl). Bedrock consists of Pre-Cambrian greywacke interbedded with quartzite at depths of 10–12 m. It forms a poorly productive aquifer with an estimated recharge of 1–50 mm yr^{-1} (Baily et al., 2011).

For this study, nine paddocks were selected: 6, 11, 20, 21, 27, 35, 37, 40, and 43. Each paddock had distinct N management histories between 2007 and 2024, including differences in fertilizer inputs, slurry and DSW applications, manure management, and contributions from biological nitrogen fixation (BNF). The paddocks also spanned a range of soil drainage classes, from imperfectly drained to moderately well-drained and well-drained (Jahangir et al., 2013). Grassland soils at the site range from poorly and imperfectly drained Gleys to well-drained silty clay loams, underlain by dense gravels intermixed with clay at depths of 0.6–10.0 m.

Hydrogeological monitoring infrastructure includes a network of multi-level piezometers and boreholes (BHs) (n=38) (Table A.2), though some have become inactive over time due to drying or mechanical failure. The paddocks selected for this study coincided with specific BHs (Figure 4-3), allowing paired soil and groundwater monitoring: Paddock 6 with BH9, Paddock 11 with BH11, Paddock 43 with BH22, Paddock 27 with BH19, Paddocks 20 and 21 with BH20 and BH5, Paddock 40 with BH6, Paddock 35 with BH23, and Paddock 37 with BH26 and BH32. Soils within these paddocks are classified as sandy, fine loamy, or coarse over fine loamy, with major groups belonging to the Brown Earth and Gley classes.

The combination of contrasting soil drainage classes, variable soil textures, and differing N management histories across paddocks (Table 4-1) provides a strong basis for examining how static and dynamic factors influence legacy N pools and their mobilization potential.

Table 4- 1: Site description including location, hydrogeology and sampling locations

Attribute	Curtins Site (Fermoy, Co. Cork)	Johnstown Castle Site (Co. Wexford)
Location & Area	48.1 ha intensive dairy research farm, Teagasc Dairy Production Centre, Fermoy (52°10′N, 8°15′W).	72.95 ha dairy research farm, Teagasc Johnstown Castle, Wexford (52°17′30″ N, 6°29′50″ W).
Topography & Setting	Flat lowland limestone plateau; negligible runoff; up-gradient of Funshion River, near public water supply well; down-gradient of River Blackwater.	Gently sloping terrain; subsurface drainage linked between up- and down-gradient units; hydrological heterogeneity across drainage classes.
Climate	Cool maritime climate; 30-year mean rainfall 1022 mm (1982–2011); 11-year mean 996 mm (2001–2011); peak rainfall in October (112 mm), lowest in July (64 mm).	Cool maritime climate; mean annual precipitation 1038 mm; mean annual air temperature 10 °C (1981–2011); peak rainfall Sept–Nov.
Meteorological Data	On-site MET station; supplemented by Moorepark station (~5 km away, active since 2003); provides rainfall, temperature, PET, evaporation. Soil Moisture Deficit (SMD) data (2011–2024), reconstructed for 2007–2010 via hybrid model.	On-site MET station; daily rainfall, wind, sunshine used for hydrological modelling. Supports estimation of soil moisture deficit (SMD) and effective rainfall across drainage classes.
Soils	Freely drained acid brown earth (Haplic Cambisol) on sandstone–limestone till; depth 0–4.5 m; A horizon: 53 % sand, 31 % silt, 16 % clay; bulk density 1.1 g cm^{-3}; porosity 52 %.	Predominantly fine loam–clay loam; Brown Earths & Gleyic Cambisols on Irish Sea till; smaller areas of sandy soils. Drainage classes: imperfect, moderately well, well-drained.
Subsoil & Geology	Waulsortian limestone at ~2.5 m depth; highly karstified, with conduits and collapse features controlling groundwater flow.	Subsoil: moderate permeability ($5 \times 10^{-8} – 5 \times 10^{-4}$ m s^{-1}), interspersed with gravel and sand lenses. Bedrock: Pre-Cambrian greywacke with quartzite, 10–12 m depth; poorly productive aquifer (recharge 1–50 mm yr^{-1}).
Hydrogeology	Karst features with high connectivity between boreholes; conduit system NW–SE; preferential flow largely within A–B horizons.	Water residence time in unsaturated zone: 1–3 years (mean 1.5 years 85 % of the time). Groundwater table: avg. 2.8 ± 1.7 m bgl (2005–2014); seasonal fluctuation 2.3–3.0 m bgl.
Paddocks Sampled	Five paddocks: 5, 6, 9, 14, 15; subdivided into sub-paddocks (e.g., 5.1, 5.2, 5.3).	Nine paddocks: 6, 11, 20, 21, 27, 35, 37, 40, 43.

N Management Records	Detailed 2001–2024; variations in fertilizer, slurry, DSW, grazing intensity, and sward composition.	Detailed 2007–2024; variation in fertilizer, slurry, DSW, grazing intensity, and sward composition; paddocks classified by drainage class.
Groundwater Monitoring	12+ boreholes; complex connectivity (e.g., BH7–BH4–BH5–BH9, BH2–BH3–BH12 clusters); tracer experiments confirm conduit-driven flow.	38 boreholes/piezometers, some inactive; paddock-to-borehole mapping allows paired soil–groundwater monitoring (e.g., Paddock 6 with BH9, Paddock 43 with BH22).
Unique Features	Karstified limestone geology with rapid subsurface transport pathways; strong focus on nitrate leaching under free-draining soils.	Contrasting soil drainage classes (poor to well drained) create spatial variability in nitrate attenuation, ammonium accumulation, and water table responses.
Denitrification Potential	Low denitrification potential (homogenous)	Relatively high denitrification potential (heterogeneous)

Abbreviations used: N: Nitrogen, BH: Borehole, DSW: Dairy Soiled Water, PET: Potential Evapotranspiration.

4.3.1.3 Nitrogen Cycling in Varying Soil Drainage Class

The rates and dominant pathways of N transformation are governed by a range of environmental factors, including soil physicochemical characteristics, substrate availability, hydrology, vegetation cover, land management, and climate (Saggar et al., 2013). Among these, soil texture and drainage class exert a particularly strong influence on N speciation and attenuation potential.

At the Curtins site (Figure 4-4), characterized by well-drained soils and a deep-water table, NO_3^- is the predominant N species transported to groundwater. The absence of artificial drainage systems and the presence of oxic, freely draining conditions favor the rapid nitrification of mineralized NH_4^+, derived from organic matter mineralization, fertilizer application, and biological fixation. Given the limited potential for denitrification and low carbon availability, NO_3^- undergoes minimal biogeochemical attenuation and is lost primarily via vertical leaching. Additional N losses occur through NH_3 volatilization and N_2O emissions, although these are secondary to leaching. DNRA (dissimilatory nitrate reduction to ammonium) is not a dominant pathway under these conditions due to the high redox potential and low C:NO_3^- ratio. The site exemplifies a system in which Nr inputs are converted and mobilized with limited transformation or retention within the soil profile.

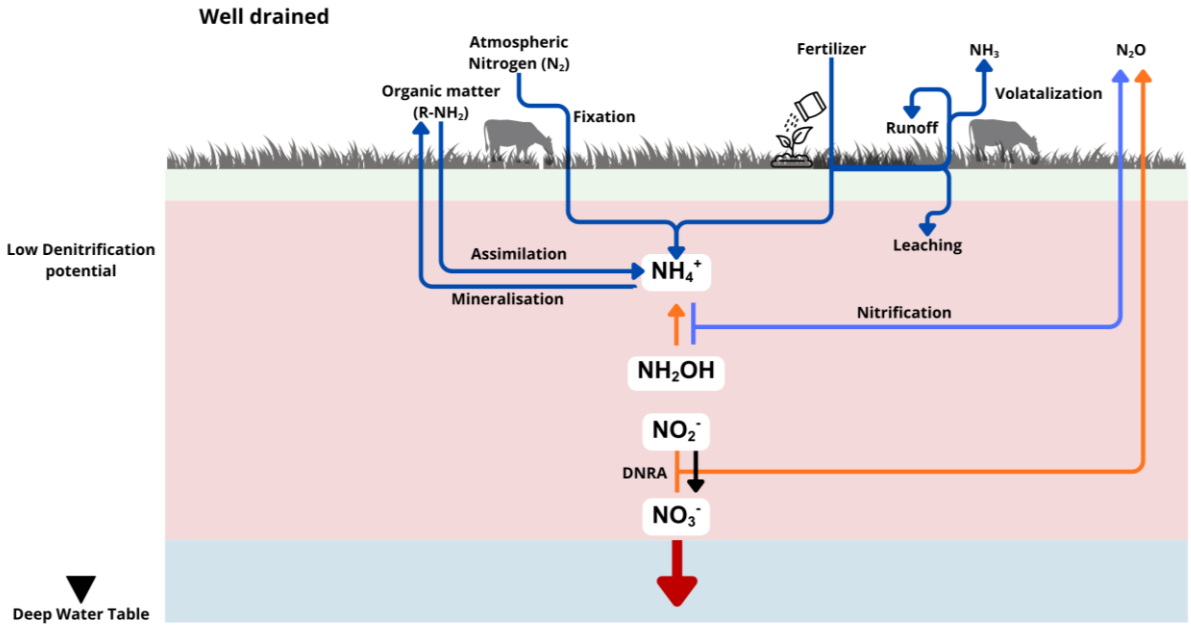


Figure 4- 4: Nitrogen cycling and transformation pathways in a well-drained agricultural grassland soil (i.e., indicative of Curtins Dairy Farm).

In contrast, the Johnstown Castle site (Figure 4-5) is characterized by pronounced spatial heterogeneity in soil type and therefore soil drainage, ranging from well- to poorly drained conditions, in combination with shallow groundwater and the presence of artificial tile drainage. In well-drained areas, NO_3^- remains the dominant N species and leaching to groundwater is the primary loss pathway. However, in moderately and poorly drained zones, although land drainage has been installed, prolonged soil saturation and reduced redox potential still dominate which promote alternative N transformation pathways.

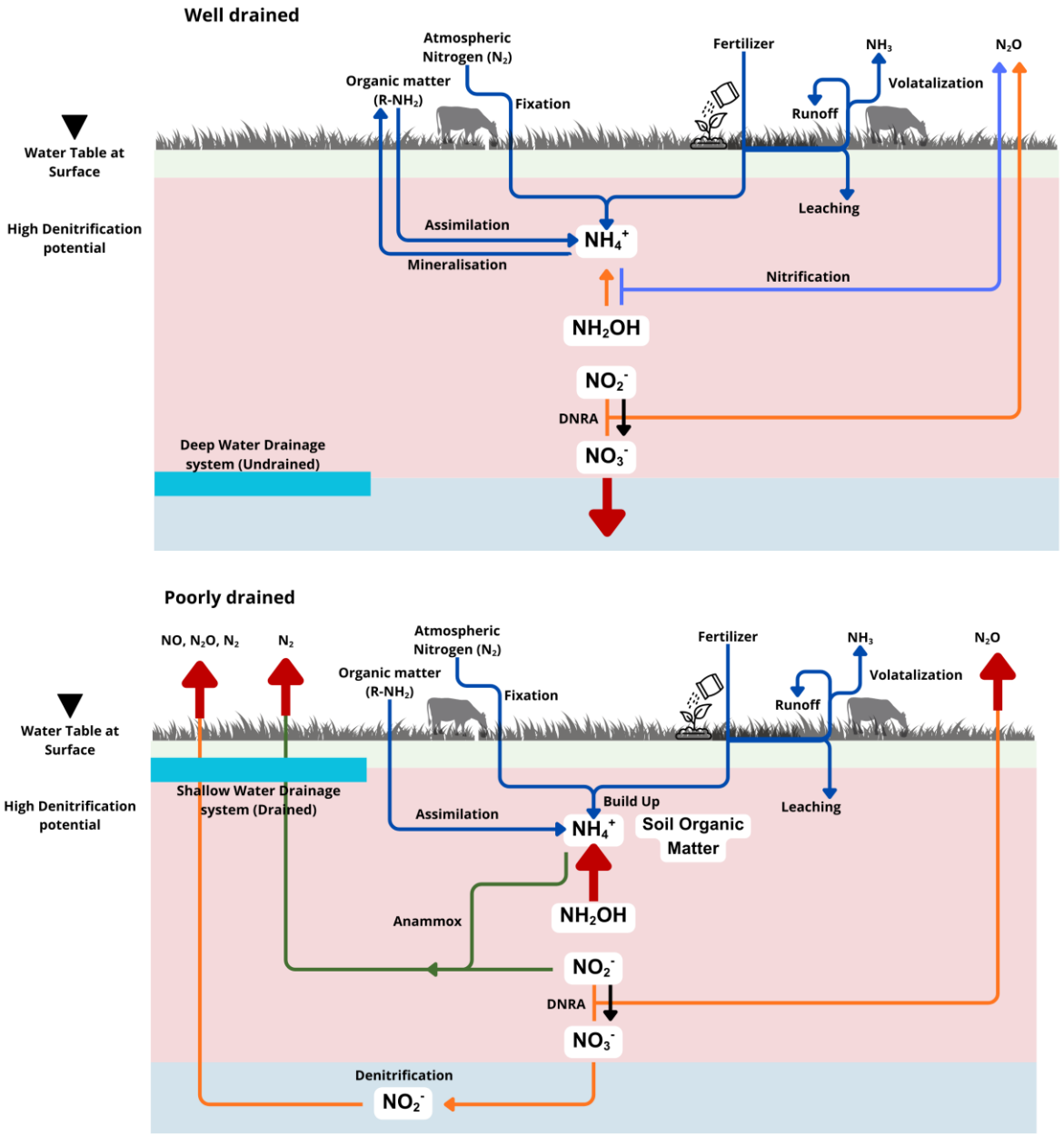


Figure 4- 5: Nitrogen cycling and transformation pathway differences in a heterogeneous drainage soil system from well to poorly drained soils (Johnstown Castle has a mixture of both these scenarios at different parts of the farm due to variations in soil type).

Under poorly drained conditions, NO_3^- is reduced to NH_4^+ via DNRA, resulting in either NH_4^+ accumulation within the soil matrix or lateral transport through subsurface drainage systems. Elevated water tables further enhance denitrification and anaerobic ammonium oxidation (anammox), leading to gaseous N losses as N_2, N_2O, and NO. While these anaerobic processes increase the capacity for in situ N attenuation in poorly drained areas, they also introduce the potential for increased greenhouse gas emissions (Clagnan et al., 2017).

Overall, the contrast between the Curtins and Johnstown Castle sites highlights how landscape-scale variability in drainage and hydrological connectivity fundamentally governs N cycling pathways, determining both the dominant forms of N loss and the balance between leaching, gaseous emissions, and internal retention in agricultural systems.

4.3.1.4 Natural Nitrogen Attenuation in Agricultural Landscapes

Natural attenuation of reactive N in agricultural soils and groundwater refers to the biogeochemical processes that mitigate NO_3^- pollution before it reaches surface waters. These processes are critical in reducing the risk of eutrophication and limiting emissions of N_2O, a potent greenhouse gas. Attenuation pathways are either temporary—including plant uptake, microbial assimilation, and DNRA—or permanent, such as denitrification and anammox.

Denitrification is the primary mechanism for permanent N removal, converting NO_3^- to dinitrogen gas (N_2) through a series of intermediates: $NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$. This microbial mediated process is favored under low redox conditions, limited oxygen availability, and sufficient electron donors (e.g., dissolved organic carbon, Fe^{2+}, or reduced sulfur). DNRA, on the other hand, transforms NO_3^- to ammonium (NH_4^+), retaining N in the system but reducing its mobility. The dominance of one pathway over the other is largely determined by the carbon-to-nitrate ratio and redox potential (Rütting et al., 2011; Jahangir, 2012).

Site Specific Attenuation: Curtins vs Johnstown Castle

Curtins Site- Well drained

The Curtins Site is underlain by free-draining brown earth soils over karstified limestone, with groundwater typically 30 m below ground level. These conditions are highly unfavourable for natural attenuation due to:

- High permeability, rapid recharge, and deep groundwater limiting residence time.
- Elevated DO (10–12 mg L^{-1}) and redox potential (Eh ~250 mV) preventing anaerobic transformations.
- Groundwater nitrate concentrations ranging from 8 to 26 mg NO_3^-–N L^{-1}, exceeding drinking water limits (Clagnan et al., 2017).

- Denitrification accounted for only 4–8 % of the delivered NO_3^-–, and DNRA activity was negligible.

Johnstown Castle Site- Variably drained

The JC site is characterised by a poorly productive aquifer composed of glacial gravels and clay overlying Ordovician bedrock. Shallow groundwater tables (<2 m bgl), low permeability, and suboxic conditions (Eh < 100 mV; DO < 2 mg L^{-1}) create highly favourable conditions for denitrification.

Field-scale and laboratory assessments at JC have demonstrated strong attenuation potential:

- In situ monitoring (2009–2011) showed that 46–77 % of delivered NO_3^-–N was removed via denitrification, reducing concentrations to 1–4 mg L^{-1} (Jahangir, 2012).
- Subsoil incubations revealed that denitrification rates declined with depth but increased significantly with carbon addition (glucose or DOC), highlighting carbon limitation in deeper layers.
- Tracer experiments confirmed an average denitrification rate of 3.92 mg NO_3^-–N L^{-1} d^{-1}, accounting for 24.5 % of injected N.
- DNRA occurred at a lower rate (0.04 mg N L^{-1} d^{-1}), contributing marginally to NO_3^-–attenuation.
- Subsoil profiles showed enhanced N_2 production at depth, indicating more complete denitrification in deeper horizons.

The contrasting outcomes at Curtins and Johnstown Castle underscore the central role of hydrogeological setting in governing nitrogen attenuation. Where conditions are conducive (e.g., shallow water table, low Eh, low DO), denitrification can substantially reduce nitrate loads. Conversely, highly permeable, oxidised environments like Curtins lack the capacity for meaningful in situ NO_3^- removal. Therefore, soil, subsoil and geology-informed nutrient management is critical for mitigating diffuse agricultural pollution.

4.3.2 Groundwater Quality Data

At Curtins, a farm-scale groundwater quality monitoring program was initiated in 2002 to assess nitrate-N (NO_3-N) concentrations across 12 boreholes. In addition to NO_3-N, the

program included ammonium-N (NH_4-N), nitrite (NO_2-N), total nitrogen (TN), and total oxidized nitrogen (TON), all measured in milligrams per liter (mg L^{-1}) following the procedures outlined by Jahangir et al. (2012). Due to a lapse in sampling, no data were collected between 2017 and 2021. For the present study, approximately 1200 groundwater samples collected between 2002 and 2024 were analyzed for NO_3-N, with similar numbers for NH_4-N (~1160) and NO_2-N (~1200). At Johnstown Castle (JC), groundwater monitoring was initiated in 2007 through a network of 38 boreholes and piezometers distributed across the site. As at Curtins, samples were analyzed for NO_3-N, NH_4-N, and NO_2-N using the methods described by Jahangir et al. (2012). The present study utilizes approximately 4000 groundwater samples collected between 2007 and 2024, providing a robust dataset for evaluating long-term N dynamics in shallow aquifers underlying imperfectly to well-drained soils.

To quantify potential N losses to groundwater at both sites, NO_3-N concentrations were converted to leaching loads (kg N ha^{-1} yr^{-1}) using the method of Hooker et al. (2008). This approach combines measured NO_3-N concentrations in leachate with effective rainfall (ER), expressed as the combined runoff and recharge amount. For Curtins, an ER of 500 mm yr^{-1} was applied, consistent with long-term studies in southern Ireland. Huebsch et al. (2013) reported ER values ranging from 478 to 544 mm yr^{-1}, while Fenton et al. (2017) adopted 500 mm yr^{-1} as a representative benchmark for nearby sites with similar soils and management. Similarly, Jahangir et al. (2012) reported a mean ER of 500 mm yr^{-1} for a comparable site during 2009–2010. For JC, the same study reported a mean ER of 611 mm yr^{-1}, which was adopted here as representative of moderately drained soils in southeast Ireland.

This combined analysis enabled the reconstruction of a 24-year trajectory of groundwater quality at Curtins and an 18-year trajectory at JC. Temporal patterns in NO_3-N concentrations and associated leaching loads were evaluated against changes in farm N management practices, with results compared to the water quality target of 30 kg N ha^{-1} yr^{-1} (Lakhanpal et al., 2025). Meeting this threshold is considered consistent with "good" status for receiving water bodies under EU water quality standards.

4.3.2.1 Nitrogen Sources and On-Farm Attribution of Groundwater Contamination

Effective N management in agricultural systems requires a clear understanding of whether elevated groundwater NO_3^- concentrations stem from local farm activity or external sources. At both Johnstown Castle (JC) and Curtins Farm (or nearby DairyGold, DG), detailed hydrological, geochemical, and isotopic investigations provide compelling evidence that on-

farm practices are the predominant drivers of subsurface N pollution. These findings underscore the necessity of site-specific mitigation strategies, rather than regional or background assumptions.

Curtins Farm: High Vulnerability and Low Attenuation in a Karst Aquifer

Curtins Farm is situated over a karstified limestone aquifer overlain by freely draining glacial till. This hydrogeological setting enables rapid vertical percolation of NO_3^-, limiting the residence time necessary for attenuation processes.

- Direct Farmyard Connectivity: Tracer and geophysical studies (Bartley et al., 2006; Huebsch et al., 2015) demonstrate clear hydraulic connectivity between farmyard nutrient sources and monitoring boreholes. This supports the conclusion that observed groundwater contamination is from vertical percolation of applied N within the farm.
- Hydrological Isolation: The site's plateau location and inferred groundwater divides prevent lateral groundwater inflow from surrounding areas. Flow paths are directed downslope toward the Funshion and Blackwater Rivers, minimizing external influence on groundwater composition.
- Boundary Borehole Validation: Monitoring at boreholes near the farm boundary confirms consistently low incoming NO_3^- concentrations, reinforcing the attribution of observed contamination to on-farm practices.
- Minimal External Sources: The only septic system on site serves a small number of non-resident staff, making anthropogenic point-source contributions negligible.

The Curtins site exhibits high NO_3^-–N concentrations in groundwater (8–26 mg L^{-1}), with limited evidence of significant denitrification due to high dissolved oxygen (10–12 mg L^{-1}) and redox potential. Denitrification accounted for only 4–8 % of NO_3^- attenuation, with little or no DNRA activity observed due to the lack of heavy textured soils and drainage systems.

Johnstown Castle (JC): Internal Management Drives Contamination and Attenuation

At JC, an intensively managed dairy research farm, groundwater monitoring reveals a distinct spatial gradient in NO_3^- concentrations, linked directly to internal N applications. The farm contains paddocks with poorly drained or moderately drained soil profiles and these paddocks have herring bone subsurface piped drainage networks connected to open drainage

channels, which connect to an underground piped-stream which finally discharges from the farm. On site there is a high density groundwater monitoring system combining multi-level boreholes and piezometers designed and installed to differentiate incoming background water quality from on-site impacts.

- Low Background Inputs: Boreholes at the upstream boundary (e.g., BH1) consistently record the lowest NO_3^-–N levels, representing background groundwater quality entering the site.
- Elevated On-Farm Contamination: Downstream increases in NO_3^- concentrations coincide with areas of high N input, including slurry spreading and dairy soiled water irrigation. In some zones, elevated DOC levels (8–25 mg L^{-1}) stimulate strong denitrification, creating localized NO_3^- reduction "hot spots."
- Isotopic Confirmation: Stable isotope analyses (Baily et al., 2011; Clagnan et al., 2017) confirm that the NO_3^- originates from organic fertiliser and slurry applied within farm boundaries, with no significant input from external anthropogenic or natural sources.
- Natural Attenuation Evidence: The end-of-pipe (EOP) monitoring locations which are found along the open drainage channel system, representative of net outflows from drained zones, show reduced NO_3^- concentrations relative to groundwater wells, indicating effective in-situ attenuation via denitrification in the upper soil and unsaturated zones.

Together, these data confirm that N loading at JC is dominated by localized management activities, with observed groundwater concentrations reflecting both input intensities and the site's strong natural attenuation capacity.

In both JC and Curtins, elevated NO_3^- concentrations in groundwater can be conclusively attributed to N inputs and management practices within the respective farm boundaries. These are governed by low incoming concentrations at site boundaries, strong isotopic and hydrological evidence of onsite N sources, spatial correlation between N input zones and groundwater contamination and isolation from external point and diffuse sources.

4.3.3 Farm Scale N Surplus

At both Curtins (2001–2024) and Johnstown Castle (2007–2024), N inputs included mineral fertilizers, slurry, and DSW. Detailed paddock- and sub-paddock–specific farm

activity records were available, expressed as kilograms of N per hectare per year (kg N ha^{-1} yr^{-1}). At Curtins, mineral fertilizer applications reached up to 360 kg N ha^{-1} yr^{-1} in the early 2000s but declined after 2006, falling to ~140 kg N ha^{-1} yr^{-1} in recent years. At JC, some paddocks received mineral fertilizer inputs as high as ~550 kg N ha^{-1} yr^{-1} in the earlier years, before declining to ~100 kg N ha^{-1} yr^{-1} more recently.

Both farms maintained Holstein–Friesian herds, but stocking rates differed. At Curtins, stocking rates historically exceeded 3 livestock units (LU) ha^{-1} before stabilizing at ~2.5 LU ha^{-1} after the adoption of improved management in 2013. At JC, stocking rates were consistently lower, typically ranging between 1.5 and 2.5 LU ha^{-1}. Since the introduction of the EU ND (Council Directive 91/676/EEC) and the associated NAP, both farms progressively reduced fertilizer inputs and adjusted stocking rates. Post-2007, best nutrient management practices emphasized greater slurry use efficiency and reduced mineral fertilizer applications, providing an opportunity to evaluate how historical versus current practices influence legacy N accumulation.

For the present study, paddocks at both sites were classified into three inorganic fertilizer input regimes: low (140–170 kg N ha^{-1} yr^{-1}), medium (170–220 kg N ha^{-1} yr^{-1}), and high (220–300 kg N ha^{-1} yr^{-1}). In addition to fertilizer, paddocks also received slurry and DSW, applied mainly after winter housing. At Curtins, slurry applications increased after 2010 and later declined, whereas at JC no clear temporal trend was evident. Manure inputs occurred directly from grazing animals. Farm records at Curtins provided paddock- and year-specific grazing intensity data for 2013–2024, expressed as grazing days ha^{-1} yr^{-1}. For 2001–2012, grazing intensity was backfilled using the 2013–2020 mean, as no major management changes occurred during this period. Grazing intensity was then converted to stocking rate (LU ha^{-1} yr^{-1}) using an assumed average grazing season length of 280 days, consistent with practices at both sites and longer than the national average of ~223 days (Läpple et al., 2012). At JC, grazing intensity and yield data were less complete, and missing years were backfilled using a three-year rolling mean.

Manure N output was calculated by multiplying grazing intensity by the daily N excretion rate of 358.4 g N LU^{-1} day^{-1} (Bougouin et al., 2022), which accounts for both fecal (184 g day^{-1}) and urinary (175.5 g day^{-1}) N. Grass N output was estimated from grazing and silage yields (kg dry matter, DM ha^{-1}), using N concentrations of 3.2 % for grazed grass and

2.5 % for silage (DairyNZ, 2013). At Curtins, yield data prior to 2013 were backfilled using the 2013–2020 mean; at JC, missing yield data were estimated using three-year averages.

Biological nitrogen fixation (BNF) was estimated from annual sward composition data (2001–2024 at Curtins; 2007–2024 at JC). Swards included perennial ryegrass (PRG), perennial ryegrass with white clover (PRGWC), and mixed-species swards (MSS). White clover was introduced only after 2021, so BNF estimates were applied from that year onwards. Clover yield was calculated as a proportion of total grass yield, with N content set at 4.6 %. Assuming 55 % of clover N was derived from fixation, BNF was estimated as 26 kg N fixed per ton of DM (Widdup et al., 2001). At JC, detailed clover composition data were not available, so estimates were based on Curtins values: 11.7 % clover in PRGWC and 13.25 % in MSS.

For each paddock, annual N inputs (fertilizer, slurry, DSW, manure, and BNF) were compared with N outputs (grass N removal via grazing and silage). This allowed reconstruction of multi-decadal N surplus trajectories for both Curtins and Johnstown Castle, providing insight into the accumulation and persistence of legacy N under contrasting soil drainage classes and management histories. Atmospheric N deposition was not included in paddock-scale N surplus estimates, as it contributed negligibly relative to overall farm inputs. Deposition values were derived from the Air Pollution Information System (APIS; UK Centre for Ecology & Hydrology), averaged for 2020–2022, and estimated at 7.07 kg N ha^{-1} yr^{-1} for Curtins and 5.41 kg N ha^{-1} yr^{-1} for Johnstown Castle, based on short vegetation/semi-natural land cover. These values represent less than 0.012 % of the total annual N inputs at either farm. Given this minimal contribution, atmospheric deposition was excluded from further analysis.

4.3.4 Soil Profiling and TN Measurements

At both Curtins and Johnstown Castle (JC), soil cores were collected within selected paddocks following a W-shaped transect, with five randomly chosen points per paddock to capture spatial variability (Wall & Plunkett, 2021). This design enabled an integrated assessment of both short- and long-term N dynamics under contrasting management histories and drainage classes. Samples were obtained using a Van Walt™ Percussion Window Sampling System (50 mm diameter, maximum depth 1 m), driven with a petrol-powered pneumatic hammer (Atlas Copco™ Cobra TT). Each 1 m core was divided into 0.1 m increments, retrieved manually, and placed into labelled sample bags. For each paddock, the

five cores were composited by depth to create depth-stratified composite samples. A natural, unfertilized grassland adjacent to the farms was also sampled and used as a control site. In total, 750 samples were collected before compositing, resulting in 160 depth-composite samples, including the control core. Field-moist samples were stored at <4 °C and processed within 48 hours. Samples were first weighed, oven-dried at 40 °C for seven days to constant mass, and gravimetric water content was calculated. Dried samples were sieved to <2 mm and homogenized before further analyses.

Potentially mineralizable nitrogen (PMN) was determined using the short-term (7-day) anaerobic incubation method of Bremner & Keeney (1966) and Schomberg et al. (2009). Duplicate subsamples (5 g) were placed in screw-capped tubes, waterlogged with 12.5 mL deionised water, and either extracted immediately (time zero) or incubated at 40 °C for 7 days (AI-7). After incubation, samples were extracted with 25 mL of 2 M KCl, shaken for 60 minutes, filtered (Whatman No. 42), and analyzed for NH_4-N, NO_2-N, and NO_3-N using an Aquakem™ 600A analyzer (Vantaa, Finland). PMN was calculated as the difference in extractable NH_4^+ between time zero and AI-7, expressed on a dry mass basis (mg N kg^{-1}), and further normalized to a daily rate by dividing by seven (Hübsch, 2015).

Extractable inorganic N (NH_4-N, NO_2-N, and NO_3-N) was also determined following Keeney & Nelson (1982) and Mulvaney (1996). Soil subsamples (50 g) were shaken with 100 mL of 2 M KCl for one hour, filtered (Whatman No. 42), and analyzed using a SKALAR SAN++ continuous flow analyzer (Skalar Analytical B.V., Breda, Netherlands). Additional subsamples were analyzed for soil pH (1:2.5 soil:water), and finely ground using a ball mill (23 Hz, 120 s, Retsch™ MM 200). Total N (TN) and total organic carbon (TOC) concentrations (% w:w) were measured on 0.25 g subsamples using a LECO™ CNS928 elemental analyzer, following 24 h hydrochloric acid pretreatment to remove inorganic carbon.

Depth-specific dry bulk density was calculated from fresh and oven-dry weights of each core segment. In cases where upper horizons yielded insufficient material, bulk density estimates for the two shallowest depths were combined. For the control core, bulk density was measured to 0.5 m. TN concentrations (mg N kg^{-1}) were converted to stocks (kg N ha^{-1}) for each 0.1 m increment and summed to obtain whole-profile (0–1 m) TN storage. Legacy N accumulation at each depth was calculated by subtracting TN in the control core from that of paddock cores, allowing vertical comparisons of N storage across sites and paddocks.

4.3.5 Data Analysis

The datasets assembled for this study combined multi-decadal records from Curtins (2002–2024) and Johnstown Castle (JC; 2007–2024). Three complementary strands of evidence were integrated: (i) paddock-scale N surplus estimates derived from detailed farm activity records, (ii) long-term groundwater chemistry monitoring, and (iii) vertical N distributions from soil cores sampled to 1 m depth. By linking these datasets, we aimed to evaluate how legacy N pools are shaped by both management history and contrasting drainage regimes.

Groundwater data included concentrations of NO_3-N, NH_4-N, and NO_2-N, monitored across borehole networks at both sites. At Curtins, the dataset spans 24 years, while at JC, 18 years of data were available. Descriptive statistics (mean, median, standard deviation, minimum, maximum) were calculated for all analyses. Temporal trends in NO_3-N concentrations were assessed using the Mann–Kendall non-parametric trend test (Mann, 1945; Kendall, 1975), with the rate of change estimated by Sen's slope (Sen, 1968). This approach is well suited for environmental time series that contain gaps or variable sampling frequency. Results are reported as changes in mg L^{-1} yr^{-1}. Statistical analyses were implemented in R using the Kendall and trend packages.

Farm N surplus trajectories were reconstructed at the paddock level by comparing inputs (fertilizer, slurry, dairy soiled water, manure, and biological N fixation) with outputs (grass N removal via grazing and silage). Current inputs were defined as the mean for 2023–2024, while historical inputs were calculated as the long-term mean across each site's study period (2002–2024 at Curtins; 2007–2024 at JC). To explore interactions between historical and current management, paddocks were grouped using quadrant analysis (Xu et al., 2018), with four categories: (1) high historical–high current, (2) high historical–low current, (3) low historical–high current, and (4) low historical–low current. This framework enabled us to examine how legacy management intensity influences subsoil N accumulation.

Soil profile data provided depth-resolved measures of TN, extractable inorganic N, and potentially mineralizable N (PMN). Results were standardized to kg N ha^{-1} for each 0.1 m depth increment and summed to 1 m, enabling whole-profile comparisons across paddocks. Legacy N accumulation was estimated by subtracting control core values (unmanaged reference) from those of managed paddocks.

A defining feature of the analysis was the explicit consideration of soil drainage class. Curtins is characterized by uniformly well-drained soils (Haplic Cambisols), whereas JC encompasses a mosaic of imperfectly drained, well/moderately drained, and well-drained soils (Jahangir et al., 2013b). This hydrological contrast provided the framework for interpreting patterns in all three datasets. At Curtins, well-drained soils are expected to favor rapid NO_3^- leaching with limited NH_4^+ retention, while at JC, imperfectly drained soils are more prone to nitrate reduction and NH_4^+ accumulation. By comparing two sites with contrasting drainage conditions, the analysis allowed us to disentangle the relative influence of management history versus inherent soil properties on the persistence and mobilization of legacy N.

Together, this multi-scale approach of integrating N surplus, groundwater quality trends, and soil profile inventories offers a comprehensive means of assessing how static (soil type, drainage class, groundwater depth) and dynamic (effective rainfall, fertilizer use, grazing intensity) factors shape the legacy N pool and its potential environmental impacts.

4.4 Results

4.4.1 Groundwater N concentrations

Groundwater monitoring at Curtins (2002–2024) and Johnstown Castle (2007–2024) revealed contrasting concentrations of NO_3-N, NO_2-N and NH_4-N across the two sites (Table 4-2). At Curtins, mean NO_3-N concentration was 9.8 mg L^{-1} (median: 9.1 mg L^{-1}), with values ranging from below detection (≤ 0.02 mg L^{-1}) to 50.9 mg L^{-1}. These concentrations frequently exceeded the EU maximum admissible concentration (MAC) of 11.3 mg L^{-1} for drinking water, particularly during the early monitoring years. In contrast, NO_2-N and NH_4-N concentrations were consistently low, with median values at or below the detection limit (≤ 0.002 and ≤ 0.02 mg L^{-1}, respectively). Occasional spikes were recorded, reaching maxima of 1.9 mg L^{-1} (NO_2-N) and 11 mg L^{-1} (NH_4-N), but these events were rare.

At Johnstown Castle, NO_3-N concentrations were substantially lower, with a mean of 3.6 mg L^{-1} (median: 3.1 mg L^{-1}), ranging from below detection to a maximum of 24.9 mg L^{-1}. Concentrations remained consistently below the EU MAC throughout the monitoring period. As at Curtins, median NO_2-N and NH_4-N concentrations were below detection limits (≤ 0.002 and ≤ 0.02 mg L^{-1}, respectively), although sporadic elevated values were observed, reaching 8.0 mg L^{-1} (NO_2-N) and 22.1 mg L^{-1} (NH_4-N). Variability (standard deviation ~0.21 for NO_2-

N and ~0.97 for NH_4-N) was greater than at Curtins, reflecting the heterogeneity of drainage classes across the site.

Table 4- 2: Summary statistics (total samples, mean, median, standard deviation, maximum, and minimum) of groundwater nitrogen (N) concentrations (mg L^{-1}) at Curtins (well-drained, 2002–2024) and Johnstown Castle (variably drained, 2007–2024).

Site	*N Species Concentration (mg L^{-1})*	*Total No. of Samples*	*Mean*	*Median*	*Standard Deviation*	*Maximum*	*Minimum*
Curtins	NO_3-N	1198	9.78071	9.137	5.865953	50.9	≤ 0.02
	NO_2-N	1208	0.013803	≤ 0.002	0.11235	1.9	≤ 0.002
	NH_4-N	1168	0.311355	≤ 0.02	0.844901	11	≤ 0.02
Johnstown Castle	NO_3-N	3849	3.64	3.11	5.86	24.95	≤ 0.02
	NO_2-N	3845	0.02	≤ 0.002	0.21	8.03	≤ 0.002
	NH_4-N	3868	0.19	≤0.02	0.97	22.08	≤ 0.02

At Curtins, groundwater NO_3-N concentrations in the early 2000s were consistently above the EU MAC for drinking water of 11.3 mg L^{-1}. Median values were ~16 mg L^{-1} in 2002, with individual boreholes recording maxima of 50.9 mg L^{-1} during the early monitoring years (Figure 4-6). Over time, NO_3-N concentrations declined steadily, with median concentrations falling below the MAC from the early 2010s onward. In recent years (2020–2024), concentrations stabilized at ~6–7 mg L^{-1}, remaining well within the compliance threshold. This represents a sustained downward trajectory over the 24-year monitoring period, with the overall mean across all years at 9.7 mg L^{-1}.

At Johnstown Castle, NO_3-N concentrations were substantially lower and more variable across the 18-year record. Mean concentrations averaged 2.8 mg L^{-1}, with maxima of 24.6 mg L^{-1} observed in isolated boreholes. Importantly, site-wide concentrations consistently remained below the EU MAC for the entire monitoring period (Figure 4-6). Temporal variability was greater than at Curtins, but no major long-term declining trend was evident. Instead, NO_3-N concentrations fluctuated around a stable mean, reflecting heterogeneity between drainage classes and paddock management histories.

When comparing the two sites, Curtins exhibited a pronounced long-term reduction in NO_3-N concentrations, whereas Johnstown Castle maintained consistently low levels with no clear directional trend. These contrasting patterns reflect the different hydrological and drainage characteristics of each site, with Curtins representing well-drained soils where groundwater responds rapidly to management changes, and Johnstown Castle characterized by variably drained soils where groundwater NO_3-N concentrations are more strongly buffered.

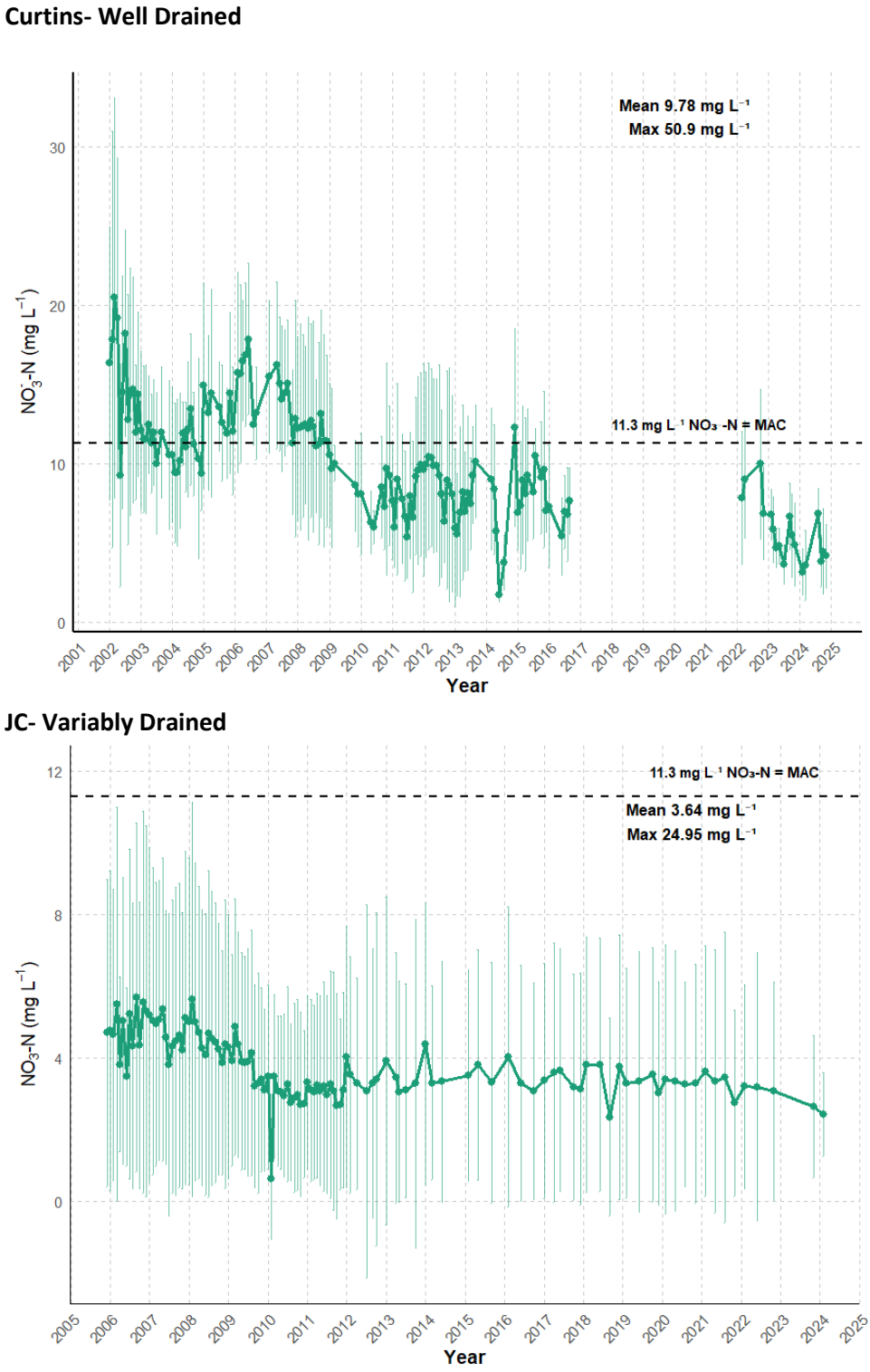


Figure 4- 6: Monthly groundwater nitrate (NO_3–N) concentrations by borehole at Curtins (2002–2024, with missing data from 2017–2021) and Johnstown Castle (2007–2024). The dashed line indicates the EU maximum admissible concentration (MAC) for drinking water (11.3 mg L^{-1}).

Abbreviations used: N: Nitrogen, NO_3-N: Nitrate, MAC: Maximum Admissible Concentration, JC: Johnstown Castle.

The Mann–Kendall test indicated significant declining trends in groundwater NO_3-N concentrations at both Curtins and Johnstown Castle, though with differences in magnitude and temporal response (Table 4-3).

At Curtins, annual data from 2002–2016 showed a strong negative trend (τ = –0.638, p = 0.0011), with Sen's slope of –0.524 mg L^{-1} yr^{-1} (95 % CI: –0.785 to –0.257). This decline was also evident in monthly data for the same period (τ = –0.516, p < 0.001), with a slope of –0.060 mg L^{-1} mo^{-1}. More recent monitoring (2022–2024) confirmed a continued decline, with τ = –0.516 (p = 0.0031) and a slope of –0.266 mg L^{-1} mo^{-1}, suggesting that reductions in groundwater NO_3-N have been sustained beyond the initial monitoring period.

At Johnstown Castle, annual records from 2005–2024 also revealed a significant downward trend (τ = –0.632, p = 0.0001), though with a much smaller Sen's slope (–0.091 mg L^{-1} yr^{-1}, 95 % CI: –0.120 to –0.046). Monthly analysis showed a weaker but still significant decline (τ = –0.449, p < 0.001), with slope estimates of –0.016 mg L^{-1} mo^{-1}.

Table 4- 3: Mann–Kendall trend test results and Sen's slope estimates for annual and monthly groundwater nitrate (NO_3–N) concentrations at Curtins (2002–2024) and Johnstown Castle (2007–2024). Sen's slope represents the median rate of change in NO_3–N concentrations, with corresponding 95 % confidence intervals (CI).

Site	*Period*	*Mann-Kendall τ*	*p-value*	*Sen's Slope*	*95 % CI (Low–High)*
Curtins (Well drained)	Annual (2002–2016)	–0.638	0.0011	–0.524	–0.785 to –0.257
	Monthly (2002–2016)	–0.516	<0.001	–0.060	–0.071 to –0.049
	Monthly (2022–2024)	–0.516	0.0031	–0.266	–0.411 to –0.120
Johnstown Castle (Variably drained)	Annual (2005–2024)	–0.632	0.0001	–0.091	–0.12 to –0.046
	Monthly (2005–2024)	–0.449	<0.001	–0.016	–0.019 to –0.013

4.4.2 Depth profiles in varying soil drainage class

Soil coring to 1 m depth revealed contrasting vertical patterns of TN accumulation between Curtins and Johnstown Castle (Figure 4-7). At Curtins, TN concentrations declined

with depth across all paddocks, with the majority of N stored in the upper 0–30 cm. However, substantial amounts of TN persisted throughout the subsoil profile (30–100 cm), with depth-integrated TN values ranging from ~2,000 to >5,000 kg N ha^{-1} depending on paddock. Paddocks with higher historical N surpluses (e.g., Paddocks 5 and 9) exhibited greater subsoil accumulation, whereas those with lower or more recent reductions in inputs (e.g., Paddock 14) showed comparatively smaller values. Despite differences in input, all paddocks demonstrated measurable N storage below the plough layer.

At Johnstown Castle, TN profiles were more variable across paddocks, reflecting the range of soil drainage classes. In imperfectly and moderately drained soils, TN declined more gradually with depth compared to Curtins, and in some cases, elevated concentrations persisted below 50 cm. In well-drained paddocks, TN distributions were more similar to Curtins, with sharper declines in the upper profile. Across all paddocks, TN values were evident throughout the 1 m profiles, but their magnitude and vertical distribution differed strongly with soil drainage class.

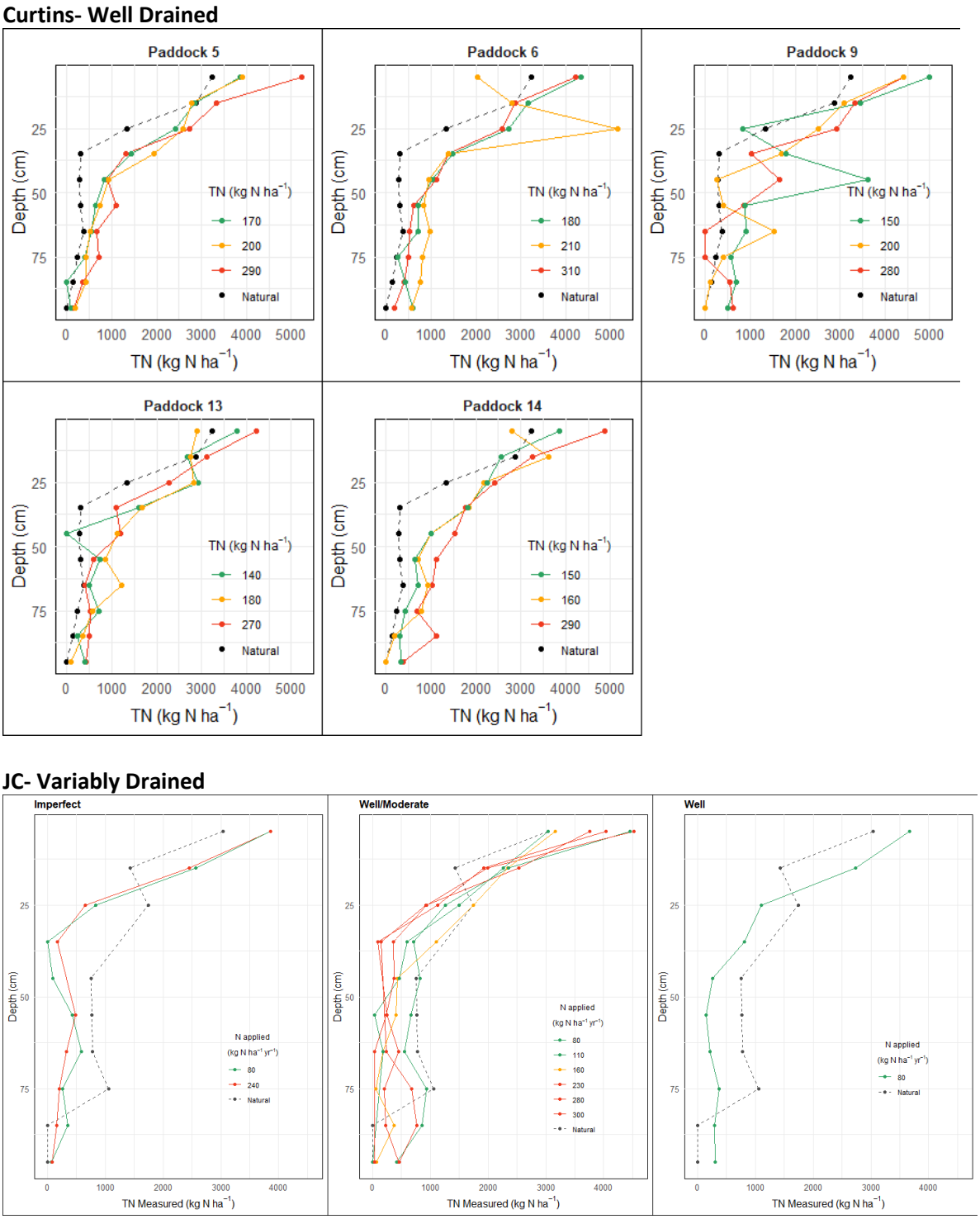


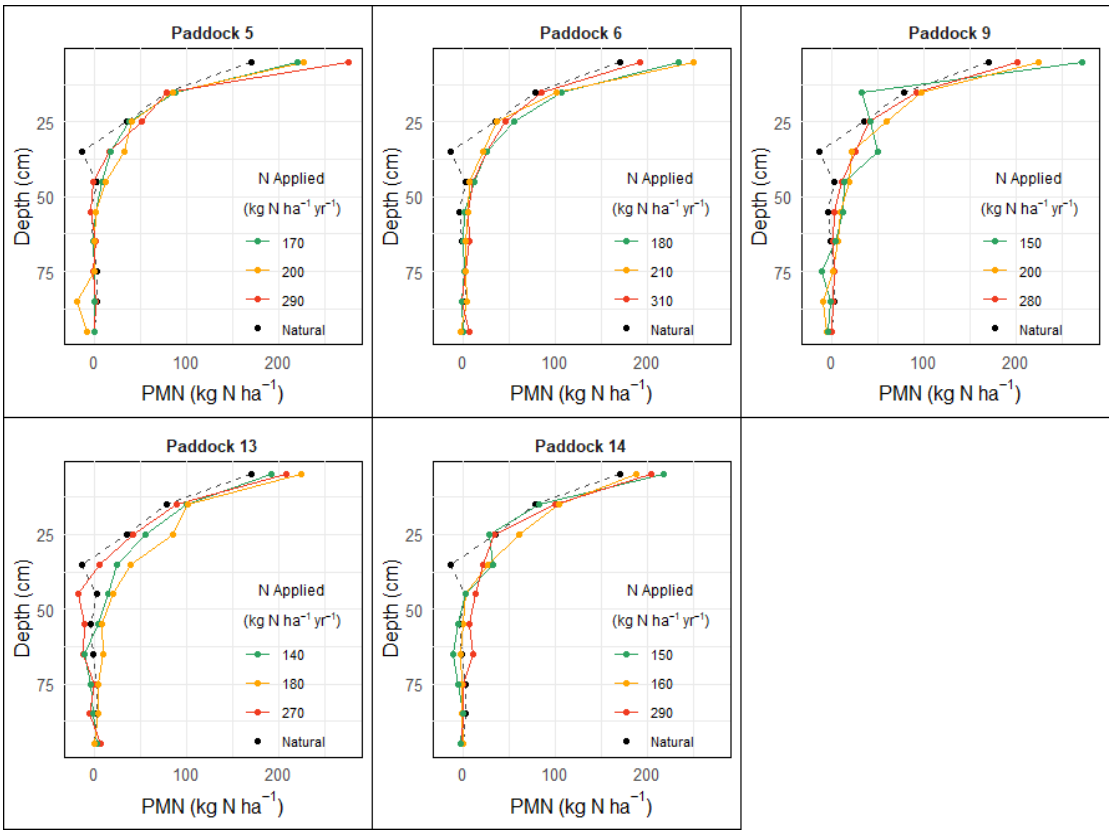
Figure 4- 7: Depth profiles of total nitrogen (TN) to 1 m across paddocks at Curtins (well-drained) and Johnstown Castle -JC (variably drained) under contrasting nitrogen (N) application regimes (fertilizer, slurry, and dairy soiled water, DSW). Each panel represents an individual paddock, with TN (kg N ha^{-1}) measured at 10 cm intervals. Colored lines indicate paddocks grouped by average N inputs (kg N ha^{-1} yr^{-1}) during 2023–2024, while the dashed black line shows TN distribution in the unfertilized control

reference profile

Soil cores to 1 m depth showed marked differences in PMN distribution between Curtins and Johnstown Castle (Figure 4-8). At Curtins, PMN concentrations were highest in the surface soils (0–10 cm), typically exceeding 250 kg N ha^{-1}, and declined sharply with depth across all paddocks. By 50–100 cm, PMN values were consistently <50 kg N ha^{-1}. Across paddocks, the general shape of the profiles was similar, but magnitude differed according to historical N inputs, with paddocks 5 and 9 showing the highest surface PMN values and paddock 14 the lowest. This reflects the strong influence of management intensity on the size of labile N pools in well-drained soils.

At Johnstown Castle, PMN depth distributions were more variable, reflecting differences in drainage class. In imperfectly drained soils, PMN remained elevated (>100 kg N ha^{-1}) at depth (30–60 cm) compared with Curtins, while in better-drained paddocks the decline with depth was sharper. Overall, PMN values in the topsoil were lower than at Curtins, but subsoil PMN pools appeared more persistent across depth increments, particularly in poorly and imperfectly drained profiles.

Curtins- Well Drained



JC- Variably Drained

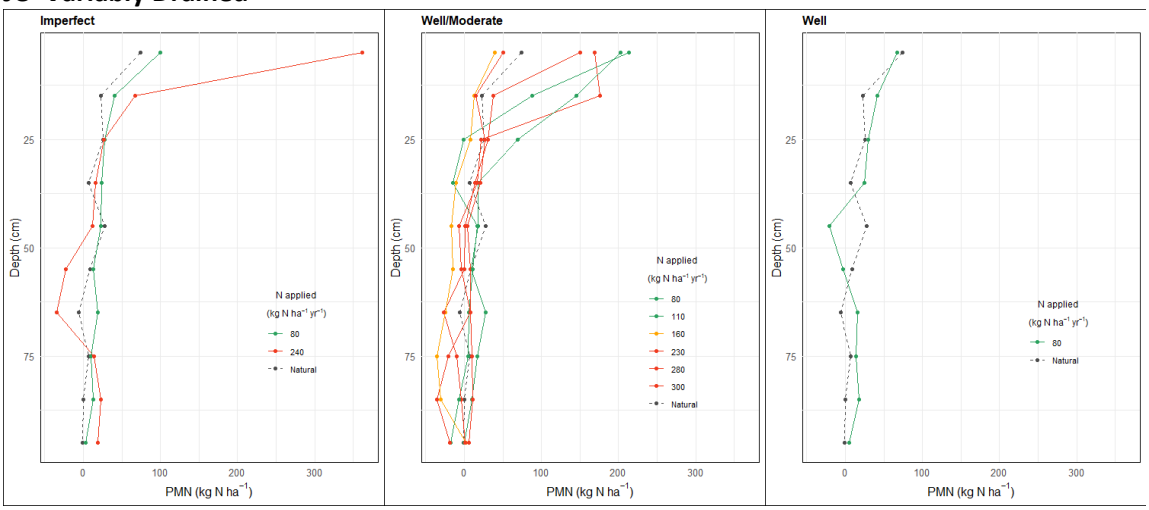


Figure 4- 8: Depth profiles of potential mineralizable nitrogen (PMN) to 1 m across paddocks at Curtins (well-drained) and Johnstown Castle- JC (variably drained) under contrasting nitrogen (N) application regimes (fertilizer, slurry, and dairy soiled water, DSW). Each panel shows an individual paddock, with PMN (kg N ha^{-1}) measured at 10 cm depth increments. Colored lines represent paddocks grouped by average N inputs (kg N ha^{-1} yr^{-1}) during 2023–2024, while the dashed black line denotes PMN distribution

in the unfertilized control reference profile.

Soil N accumulation profiles differed strongly between Curtins and Johnstown Castle, reflecting the influence of drainage class on legacy N storage (Figure 4-9). At Curtins (well-drained), accumulated N was concentrated in the upper 0–30 cm, where mean values exceeded 1500 kg N ha^{-1} in the 5–15 cm and 25–35 cm increments. Below 50 cm, accumulated N declined but remained detectable through to 1 m depth, with values of ~500–800 kg N ha^{-1} in deeper increments. Standard deviations were relatively small across depths, indicating consistent accumulation patterns across sampled paddocks.

At Johnstown Castle, depth distributions varied markedly between drainage classes. In imperfectly drained soils, accumulated N was concentrated in the upper 0–25 cm but declined steeply below this depth, with only modest amounts detected in the subsoil. In well/moderately drained soils, accumulated N was more evenly distributed across the 0–100 cm profile, with 500–1200 kg N ha^{-1} present in most depth increments. By contrast, well-drained soils at Johnstown Castle displayed high accumulations (>1500 kg N ha^{-1}) at both surface and deeper depths (40–80 cm), indicating more vertical movement and storage of legacy N.

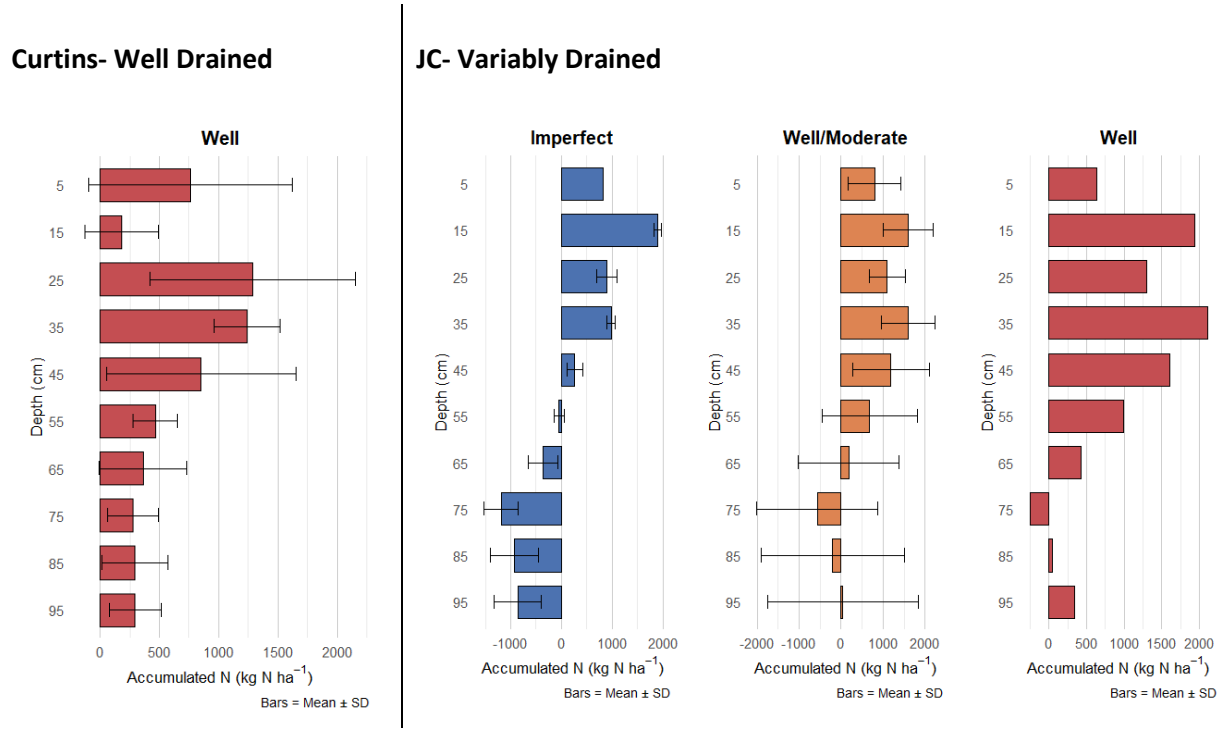


Figure 4- 9: Depth profiles of accumulated nitrogen (N) to 1 m at Curtins (well-drained) and Johnstown Castle- JC (variably drained), grouped by soil drainage class. Panels represent well-drained (Curtins and JC), imperfectly drained (JC), and moderately drained (JC) soils. Bars indicate mean accumulated N (kg N ha^{-1}) by 10 cm depth increments, with error bars denoting ± standard deviation (SD) across replicate paddocks.

Soil N accumulation to 1 m depth showed clear differences between Curtins and Johnstown Castle, as well as across low, medium, and high N input paddocks (Figure 4-10). At Curtins, accumulated N increased systematically with input intensity. Under low N inputs (140–170 kg N ha^{-1} yr^{-1}), accumulation was largely confined to the upper 0–30 cm, with values of ~500–1000 kg N ha^{-1} per depth increment and little accumulation below 50 cm. Under medium inputs (170–220 kg N ha^{-1} yr^{-1}), surface accumulations increased substantially (~1200–1500 kg N ha^{-1} in the top 0–15 cm) with measurable storage down to 70–100 cm. High input paddocks (220–300 kg N ha^{-1} yr^{-1}) displayed the largest accumulations, with >1500 kg N ha^{-1} in surface increments and considerable amounts (>5000 kg N ha^{-1} total accumulation) persisting through the subsoil over the top 50 cm depth.

At Johnstown Castle, accumulated N patterns were more variable, reflecting the site's heterogeneity in soil drainage. Under low N inputs (80–170 kg N ha^{-1} yr^{-1}), N was concentrated in the upper 0–25 cm, with limited storage at depth. Medium inputs (170–220 kg N ha^{-1} yr^{-1}) resulted in higher accumulations across the profile, particularly in the top 30 cm (~1200–1500

kg N ha^{-1}), though some paddocks showed modest subsoil accumulation. Under high N inputs (220–300 kg N ha^{-1} yr^{-1}), total N accumulation for the 1m depth exceeded 2000 kg N ha^{-1} in surface layers, with subsoil pools also evident, though generally lower than those observed at Curtins.

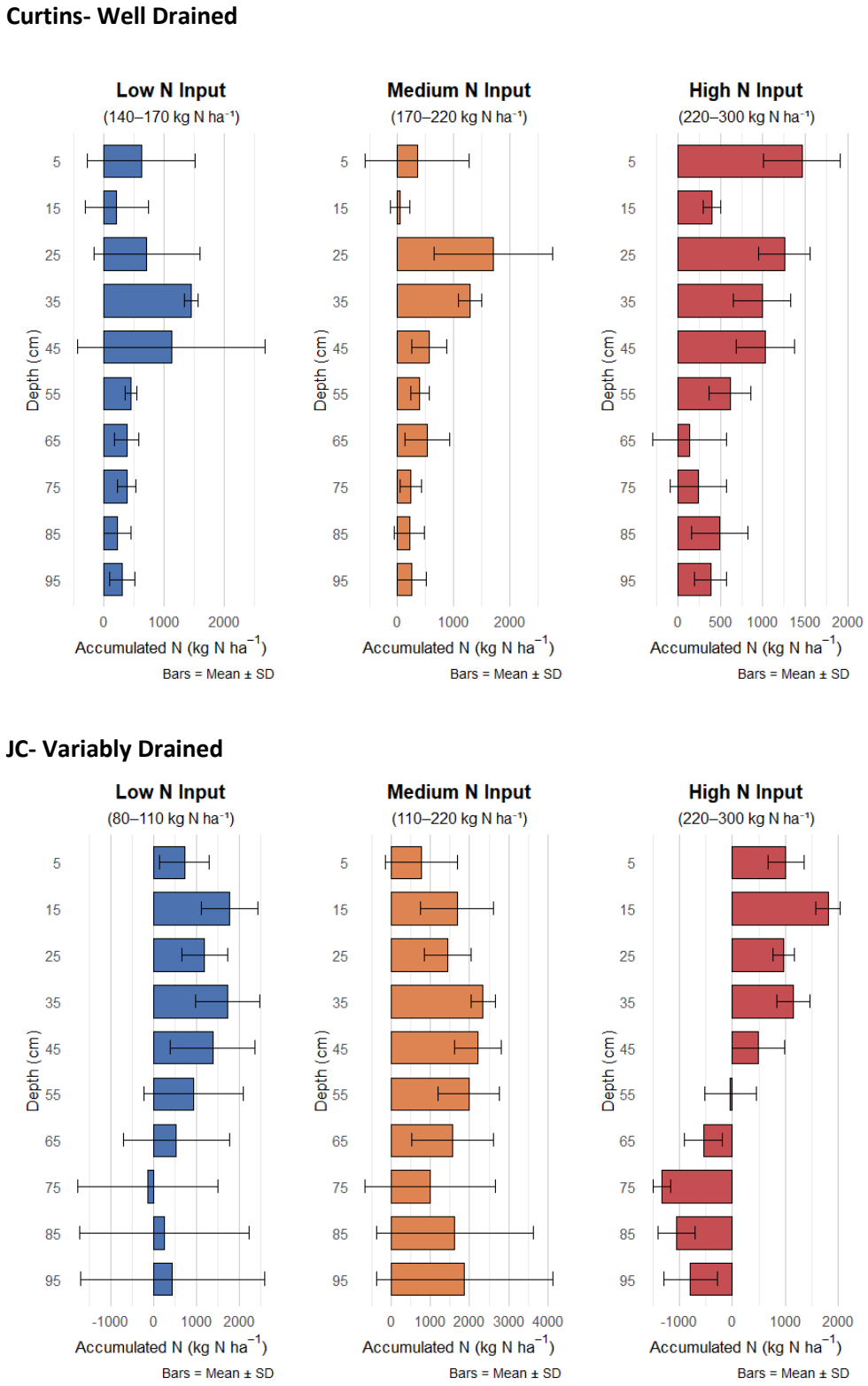


Figure 4- 10: Depth profiles of accumulated nitrogen (N) to 1 m at Curtins (well-drained) and Johnstown Castle- JC (variably drained), stratified by N input class. Panels show low (140–170 kg N ha⁻¹ yr⁻¹), medium (170–220 kg N ha⁻¹ yr⁻¹), and high (220–300 kg N ha⁻¹ yr⁻¹) input categories. Bars represent mean accumulated N (kg N ha⁻¹) by 10 cm depth increments, with error bars denoting ± standard deviation (SD) across replicate paddocks.

4.4.3 N Surplus and trends over the years

At Curtins, N surpluses were consistently higher in the early 2000s, exceeding 400 kg N ha^{-1} yr^{-1} in several years, and showed a declining trend after ~2006 (Figure 4-11). Recent years (2020–2024) exhibited lower surpluses of ~200–250 kg N ha^{-1} yr^{-1}. Grass N output increased modestly over the same period, stabilizing around 350–400 kg N ha^{-1} yr^{-1}. Estimated NO_3^- leaching losses followed the surplus trend, peaking at ~60 kg N ha^{-1} yr^{-1} in the early monitoring years and gradually declining to ~20–25 kg N ha^{-1} yr^{-1} in recent years. Notably, Curtins achieved the 30 kg N ha^{-1} yr^{-1} water quality target in most years after 2010.

At Johnstown Castle, N surpluses were consistently lower than at Curtins, ranging mostly between 200–300 kg N ha^{-1} yr^{-1}, with no strong downward trend over time. Grass N outputs were stable at ~350–400 kg N ha^{-1} yr^{-1}. NO_3^- leaching losses were generally <20 kg N ha^{-1} yr^{-1}, with several years recording values <10 kg N ha^{-1} yr^{-1}, consistently below the 30 kg N ha^{-1} yr^{-1} threshold.

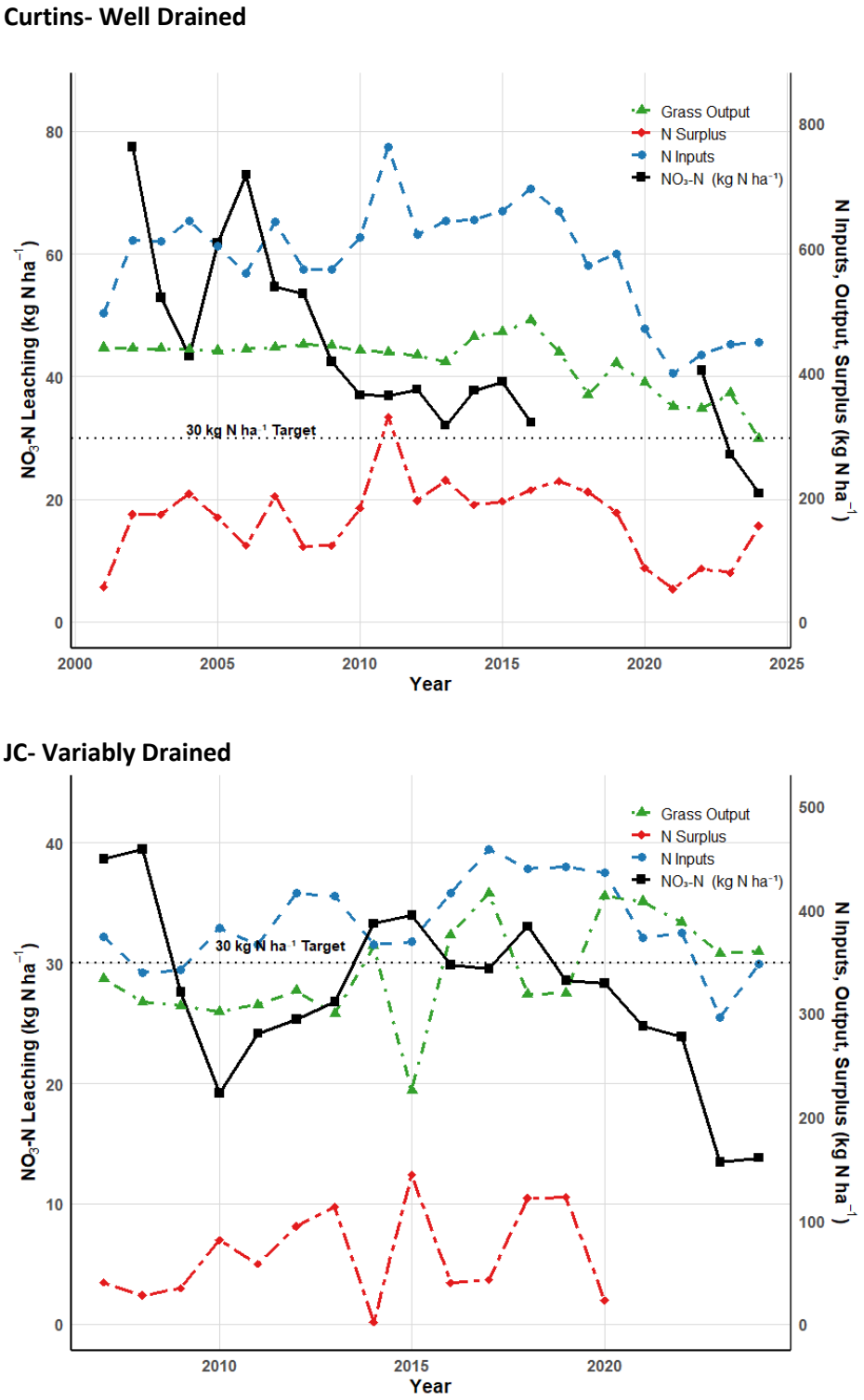


Figure 4- 11: Annual trajectories of nitrate leaching (NO_3-N; black line, left axis), total nitrogen (N) inputs (blue), grass N outputs (green), and N surpluses (red) at Curtins (well-drained) and Johnstown Castle- JC (variably drained) from 2001–2024 and 2007–2024, respectively. Lines represent annual means across monitored paddocks whereas, the dashed horizontal line indicates the water quality threshold of 30 kg NO_3-N ha^{-1} yr^{-1}.

At Curtins, accumulated N over the top 50 cm depth ranged between ~3000 and 6500 kg N ha^{-1}, with most paddocks falling within the 3500–5000 kg N ha^{-1} range (Figure 4-12). N surpluses spanned from ~100 to >250 kg N ha^{-1} yr^{-1}, and higher surpluses (>150 kg N ha^{-1} yr^{-1}) were often associated with greater subsoil accumulation. However, the relationship was not strictly linear, suggesting that additional factors beyond surplus magnitude (e.g., drainage connectivity, soil texture, karst features) influence vertical N storage.

At Johnstown Castle, accumulated N over the top 50 cm depth was generally higher than at Curtins, ranging from ~4000 to over 11,000 kg N ha^{-1}. Despite these higher pools, annual N surpluses were consistently lower, rarely exceeding 150 kg N ha^{-1} yr^{-1}. Some paddocks even recorded near-zero or negative balances while still exhibiting substantial legacy N in the subsoil, indicative of historical accumulation and slow turnover under imperfectly and moderately drained conditions.

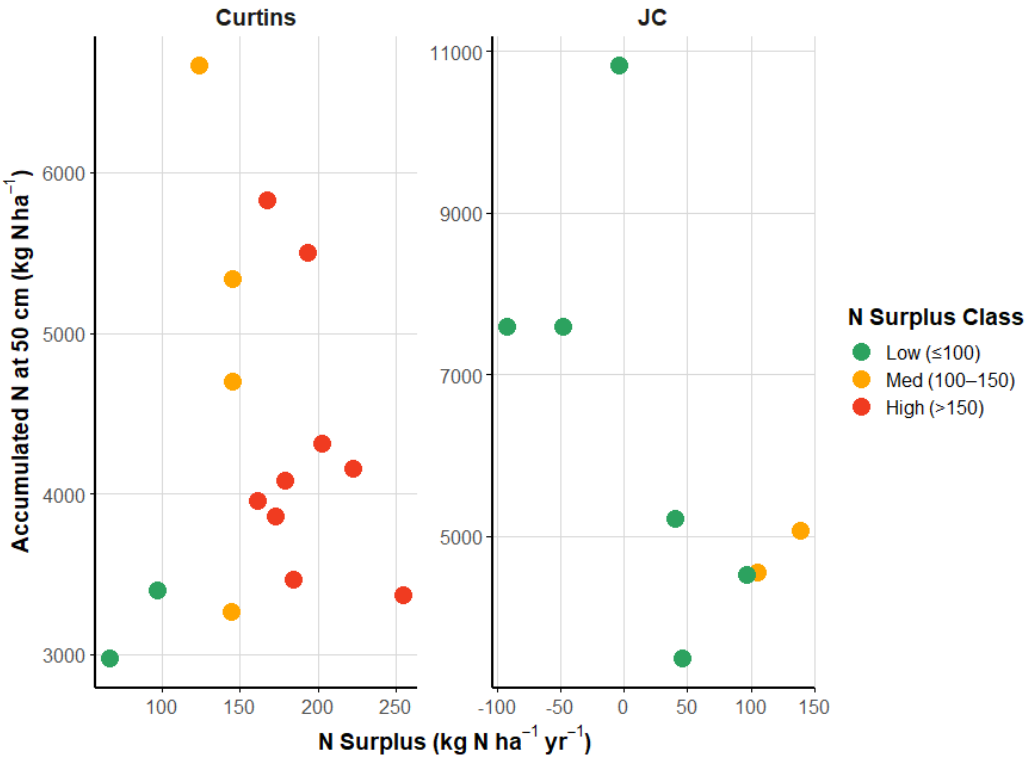


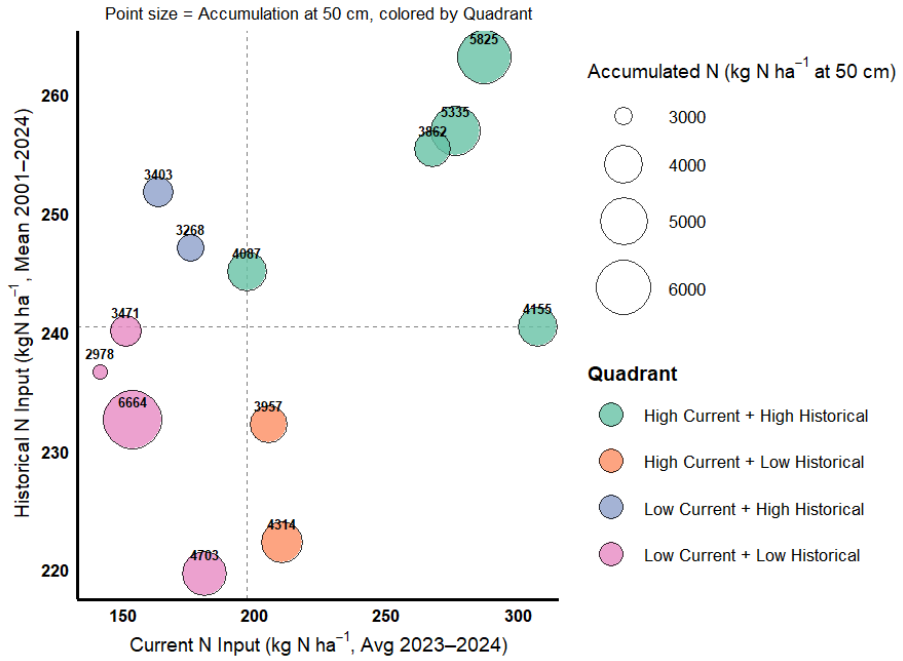
Figure 4- 12: Nitrogen (N) surplus and accumulated N over the top 50 cm soil depth at Curtins (well-drained) and Johnstown Castle- JC (variably drained). Points represent paddocks, grouped by N surplus class (low, medium, high).

4.4.4 Nitrogen Accumulation as a function of N input and current and historical application

At Curtins, most paddocks fell into the high historical–high current quadrant, reflecting the site's legacy of intensive management combined with ongoing high inputs (Figure 4-13). These paddocks also displayed relatively large, accumulated N pools over the top 50 cm, typically between 4000–6000 kg N ha^{-1}. A smaller number of paddocks were situated in the high historical–low current quadrant, showing reduced recent inputs but still carrying substantial legacy N accumulation. Paddocks in the low historical–low current quadrant were less common and corresponded to the lowest observed N accumulation.

At Johnstown Castle, a contrasting pattern emerged. The majority of paddocks were located in the high historical–low current quadrant, indicating that although recent N inputs (2023–2024) are modest (<200 kg N ha^{-1} yr^{-1}), substantial legacy pools (>7000 kg N ha^{-1} over the top 50 cm) remain. Only a few paddocks fell within the high current–high historical quadrant, and even these were associated with lower accumulation compared to Curtins. The distribution highlights a disconnect between current inputs and stored subsoil N, consistent with the site's poorer drainage and longer residence times.

Curtins –Well Drained



JC- Variably Drained

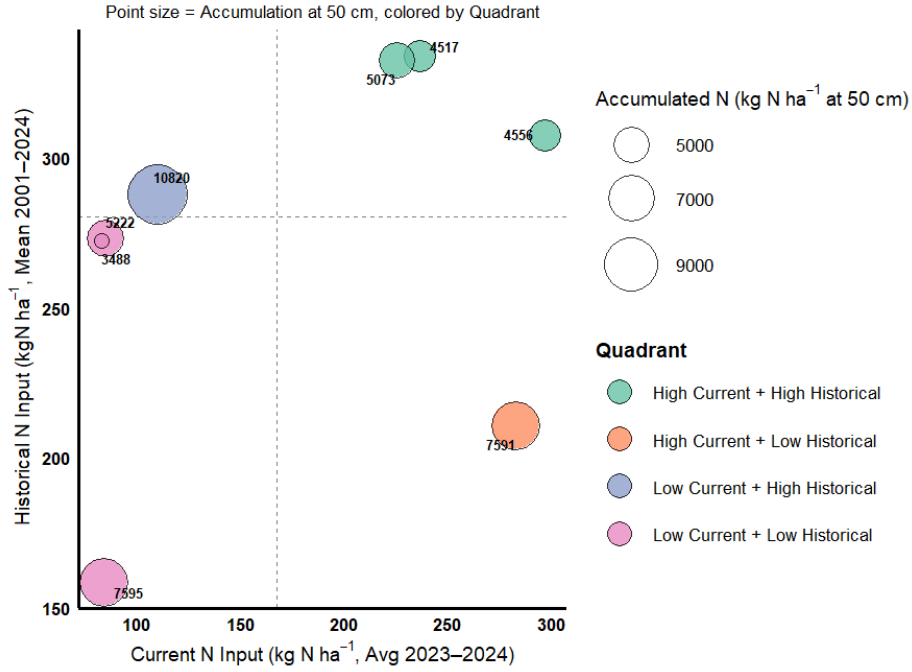


Figure 4- 13: Quadrant analysis of nitrogen (N) inputs and subsoil N accumulation (over the top 50 cm depth) at Curtins (well-drained) and Johnstown Castle- JC (variably drained). Paddocks are classified by current (2023–2024) and historical (2001–2024) N inputs, with quadrants defined as High Current + High Historical, High Current + Low Historical, Low Current + High Historical, and Low Current + Low Historical. Point size indicates N accumulation (kg N ha^{-1}) over the top 50 cm depth.

4.5 Discussions

4.5.1 Groundwater trends

The contrasting characteristics of Curtins (well-drained, karstic, low denitrification potential) and Johnstown Castle (JC) (variably drained, finer-textured, moderate–high denitrification potential) provide a valuable framework for understanding how soil type, drainage, hydrogeology, and redox dynamics govern both NO_3-N mobility and legacy N persistence in intensive dairy systems. These differences explain much of the observed variability in groundwater NO_3-N concentrations, leaching behavior, and rates of water-quality recovery across Irish agricultural landscapes.

At Curtins, the combination of free-draining loam soils, shallow unsaturated zones, and karstified limestone bedrock produces highly transmissive pathways for NO_3-N movement. These hydrogeological conditions, coupled with limited denitrification capacity (Jahangir et al., 2012, 2013a), make the site particularly sensitive to N surpluses and management changes. Historical data confirm this vulnerability: groundwater NO_3-N concentrations in the early 2000s reached a median of ~16 mg L^{-1} and peaks above 50 mg L^{-1}, far exceeding the EU MAC for drinking water of 11.3 mg L^{-1} (Huebsch et al., 2013). These levels reflect the cumulative effect of fertilizer applications exceeding 300 kg N ha^{-1} yr^{-1} under intensive management during that period.

Following the implementation of measures under the EU ND, mineral N fertilizer use declined to ~140–220 kg N ha^{-1} yr^{-1} after the mid-2000s. This reduction corresponded with a steady and significant decrease in groundwater NO_3-N, reaching 6–7 mg L^{-1} by 2020–2024, with a Sen's slope of –0.52 mg L^{-1} yr^{-1} (2002–2016). This trend demonstrates Curtins' rapid hydrological responsiveness: short residence times (~1–2 years) and low soil organic matter turnover allow groundwater quality to improve relatively quickly once inputs are reduced. However, Curtins' thin, free-draining soils also mean that N retained within the upper 50 cm (3,000–6,500 kg N ha^{-1}) remains highly mobile and strongly coupled with groundwater flow. These characteristics render such systems highly vulnerable to NO_3-N pollution, but also fast responders when management improves.

In contrast, Johnstown Castle represents a buffered, variably drained system where the interplay of soil texture, hydrology, and denitrification markedly alters N cycling. The soils here are finer-textured, with higher silt and clay contents that enhance N retention through mineral–organic associations, NH_4^+ fixation, and restricted macropore flow. These soils are

underlain by imperfectly drained subsoils, a shallow and fluctuating water table, and moderate to high denitrification potential (Jahangir et al., 2013; Clagnan et al., 2019), all of which contribute to a more closed N cycle.

As a result, groundwater NO_3-N concentrations at JC remained consistently below the EU MAC, with a mean of 3.6 mg L^{-1} and median of 3.1 mg L^{-1} (n = 3849), despite long-term surpluses averaging 100–150 kg N ha^{-1} yr^{-1}. The maximum recorded concentration of 24.9 mg L^{-1} occurred only during isolated episodes, reflecting limited vertical transport and strong redox regulation. Occasional peaks in reduced N forms; NO_2-N up to 8.0 mg L^{-1} and NH_4-N up to 22 mg L^{-1} suggest fluctuating redox conditions that periodically suppress nitrification and promote denitrification. Artificial drainage networks further modify subsurface flow, promoting lateral movement and surface runoff rather than deep NO_3-N transport.

Despite these protective mechanisms, substantial legacy N accumulation was observed in the JC soil profile, with total N accumulation in the upper 50 cm reaching 4,000–11,000 kg N ha^{-1}, markedly higher than at Curtins. This accumulation likely reflects both physical protection of organic N in clay-rich aggregates and periodic anoxia that limits mineralization. Consequently, JC acts as a slow retainer, maintaining low groundwater NO_3-N concentrations but holding large, long-lived N stores that may only be gradually mineralized over decades. Temporal trend analysis confirms a slow but statistically significant decline (Sen's slope = –0.09 mg L^{-1} yr^{-1}, 2005–2024), consistent with the site's longer water residence times and denitrification buffering.

Our observations corroborate the findings of Clagnan et al. (2019) and Jahangir et al. (2013), who demonstrated that imperfectly drained soils with shallow groundwater tables and active denitrification zones can effectively attenuate NO_3-N losses, albeit at the expense of accumulating organic and mineral N pools in subsoils. The episodic occurrence of NH_4-N peaks at JC supports this mechanism, indicating alternating oxic–anoxic cycles that both suppress mineralization and promote temporary NH_4^+ build-up. Similarly, studies by Forrestal et al. (2025) showed that leaching from grazed grasslands is strongly modulated by soil type, with free-draining soils exhibiting higher NO_3^- fluxes and imperfectly drained soils favoring N retention and denitrification—a dynamic that aligns closely with our comparative findings.

Together, these results reveal a fundamental trade-off between N mobility and storage across Irish soil types. Well-drained sites like Curtins function potentially as fast transmitters: they quickly translate management surpluses into groundwater NO_3-N signals but also show

rapid recovery when inputs decrease. Variably drained, clay-rich sites like JC most likely function as slow retainers: they suppress immediate leaching but accumulate large legacy N reserves that can delay water-quality improvements for decades.

This distinction has direct implications for Ireland's compliance with EU environmental directives. Under the EU WFD, member states must achieve "good status" for all waters, yet Ireland remains among the countries with the highest share of agricultural pressures, particularly in the south and southeast regions dominated by intensive dairy systems similar to Curtins and JC. Current six-year reporting cycles within the WFD and ND are too short to reflect the slow turnover of legacy N in variably drained soils, risking misinterpretation of delayed responses as policy failure.

From a management and policy perspective, the findings underscore the importance of drainage-informed nutrient strategies. In well-drained systems like Curtins, interventions should focus on reducing fertilizer inputs, improving slurry management, and maintaining nutrient-use efficiency, as improvements can be realized within relatively short timeframes. In variably drained systems like JC, strategies should prioritize long-term stabilization and gradual drawdown of stored N through lower surpluses, multi-species swards, enhanced soil structure, and targeted monitoring of mineralization dynamics.

To support evidence-based policy, Ireland must expand monitoring frameworks to include deep soil sampling (to 1 m) alongside existing catchment and groundwater networks. Current monitoring primarily focuses on surface and outlet water quality, neglecting the substantial subsurface N reservoirs that drive lagged responses. Integrating soil N data into national monitoring would allow more accurate recovery forecasts and more credible progress reporting under the WFD.

In summary, the contrasting results at Curtins and Johnstown Castle demonstrate that legacy N dynamics are not uniform but context-dependent, emerging from the interaction of soil texture, drainage, denitrification potential, hydrology, and management. Well-drained karstic sites respond rapidly but remain vulnerable to leaching; variably drained clay–silt systems buffer groundwater but store persistent legacy pools. Understanding and managing these contrasts is central to achieving Ireland's WFD and ND targets and to aligning its dairy sector with evolving EU sustainability frameworks.

4.5.2 N Surplus at the 2 varying sites

The farm-scale N surplus trajectories further illustrate the interaction between management intensity, soil texture, and drainage regime in shaping N accumulation and transfer. At Curtins, surpluses were initially high in the early 2000s, consistent with intensive inputs of up to 360 kg N ha^{-1} yr^{-1} on well-drained loam soils underlain by karstic limestone. These conditions characterized by rapid percolation, short residence times, and limited denitrification potential translated high surpluses almost directly into groundwater NO_3-N loads. Following the implementation of the EU ND and the adoption of improved nutrient management (reduced chemical fertilizer, greater slurry recycling, and refined timing of applications), surpluses declined sharply, accompanied by a marked reduction in groundwater NO_3-N concentrations. By the late 2010s, Curtins consistently tracked below the 30 kg N ha^{-1} yr^{-1} leaching target, demonstrating how reductions in inputs were rapidly transmitted to groundwater under free-draining, fast-response conditions.

At Johnstown Castle, surpluses in several paddocks were of similar magnitude to Curtins (~100–150 kg N ha^{-1} yr^{-1}), yet leaching losses remained substantially lower. This contrast reflects the influence of soil texture and hydrology: the higher silt and clay content, imperfect drainage, and shallow fluctuating water table at JC promote denitrification and longer residence times, effectively buffering the transfer of surplus N to groundwater. However, these same factors encourage N retention in the soil profile and subsoil layers, leading to persistent accumulation despite reduced inputs.

Together, these results reinforce a key distinction between input-driven accumulation and soil property-controlled transport. At Curtins, N surpluses are rapidly transmitted to groundwater, whereas at JC, they are retained within the soil matrix, representing a long-term source of mineralizable N that may be released gradually over time. This highlights that reducing inputs alone may not immediately reduce groundwater N concentrations in variably drained systems, where soil and hydrological conditions drive slower responses.

4.5.3 Legacy N Accumulation over decades in varying sites

The soil core profiles provide further insight into how soil texture, hydrogeology, and drainage class mediate the depth distribution and persistence of legacy N. At Curtins, most legacy N was concentrated within the upper 50 cm (3,000–6,500 kg N ha^{-1}), but substantial pools also occurred below this depth, consistent with rapid percolation and limited retention

capacity in the well-drained loam soils overlying karstic limestone. These deeper accumulations reflect historical surpluses exceeding 300 kg N ha^{-1} yr^{-1} and the highly transmissive nature of the subsurface, which facilitates downward movement of NO_3^- and dissolved organic N. The interactions between the unsaturated zone and groundwater explains why groundwater NO_3-N concentrations at Curtins declined rapidly once inputs were reduced. However, the low denitrification potential below the rooting zone (Jahangir et al., 2013) means that subsoil legacy N remains a continuing risk to groundwater, even after management improvements.

At Johnstown Castle, profiles across imperfectly, moderately, and well-drained soils revealed a wide range of accumulation patterns driven by soil texture, drainage, and management. Imperfectly drained, finer-textured soils (with higher silt and clay content) retained larger N stores in surface layers, reflecting restricted downward transport and greater physical and chemical stabilization of organic matter. Moderately drained soils displayed more uniform distributions, while the relatively well-drained paddock, situated in the drier portion of the farm showed deeper migration similar to Curtins. This paddock also received disproportionately higher manure deposition due to management practices that shifted grazing to this area when other, wetter regions of the farm were unsuitable. As a result, total N inputs from both fertilizer and organic sources were locally elevated, contributing to greater N accumulation at depth.

Overall, JC exhibited larger total accumulations (>7,000–11,000 kg N ha^{-1} over the top 50 cm) compared with Curtins, but this difference arises primarily from slower turnover and stronger retention rather than higher inputs across the entire site. Denitrification and redox buffering under the shallow and fluctuating groundwater table suppress mineralization and promotes the long-term storage of organic N, particularly in finer-textured areas.

The PMN profiles reinforce these contrasts. At Curtins, PMN declined sharply with depth, consistent with aerobic turnover and the predominance of labile organic fractions subject to mineralization and leaching. In contrast, JC showed sustained PMN values at depth, particularly in imperfectly drained soils, reflecting anaerobic microsites and slower organic matter decomposition. This pattern indicates that legacy N at Curtins is more labile and responsive to management, whereas at JC, legacy N is more stable but persistent, potentially mineralizing slowly over decades.

These results demonstrate that N accumulation is not governed by a multi-factor response shaped by soil texture, soil drainage, hydrogeology, denitrification capacity, and historical management intensity. Well-drained, coarse-textured systems transmit surpluses quickly and recover rapidly when inputs fall, while finer-textured, variably drained systems act as long-term N reservoirs with slow turnover and extended lag times for water quality recovery.

The magnitude of accumulation observed at both sites exceeds that typically reported for other grasslands or catchments elsewhere (e.g., Van Meter et al., 2016; Zhou et al., 2016; Liu et al., 2022), underscoring the exceptional storage potential of temperate grass-based dairy soils. In particular, the larger and more persistent legacy N accumulation at Johnstown Castle align with recent findings that finer-textured soils with higher silt and clay content enhance N retention through multiple mechanisms; including organo-mineral bonding, NH_4^+ fixation, and reduced macropore flow which limit N mobility and prolong turnover (Geisseler et al., 2024; Yuan et al., 2025; Ilampooranan et al., 2022). Such soil–mineral interactions explain why denitrification and mineralization proceed more slowly in variably drained, clay-rich systems, resulting in higher but more stable total N pools. Together, these findings reinforce that legacy N accumulation in Irish dairy soils reflects the combined influence of soil texture, drainage, hydrogeology, and management, and highlight the importance of integrating these controls into national nutrient budgets and water quality policy frameworks.

4.5.4 Influence of historical versus current inputs

Quadrant analysis of historical and current N inputs provides further evidence of how soil drainage, texture, and hydrogeology mediate legacy accumulation and responsiveness to management. At Curtins, paddocks with both high historical and high current inputs corresponded to the largest subsoil legacy pools (>5,000 kg N ha^{-1}), while those with reduced recent inputs (the high-historical–low-current quadrant) showed signs of gradual depletion. This indicates that legacy N in well-drained, karstified soils with deep water table is more labile and responsive to current management changes, consistent with the rapid groundwater quality improvements observed at this site once fertilizer rates were reduced. The short water residence times and limited denitrification capacity in Curtins' karstic subsurface amplify this responsiveness, allowing management effects to manifest within years rather than decades.

At Johnstown Castle, most paddocks clustered in the high-historical–low-current quadrant, with large and persistent legacy stores (>7,000 kg N ha^{-1}) remaining despite substantial input reductions in recent years. These patterns reflect the combined influence of

finer soil texture (higher silt and clay content), shallow and fluctuating groundwater tables, and moderate-to-high denitrification potential, which together slow mineralization and enhance N retention in stable organic and mineral-associated forms. As a result, reductions in surface N inputs take far longer to translate into measurable decreases in subsoil legacy N or groundwater NO_3-N, producing multi-decadal recovery timelines. This finding supports the observations of Jahangir et al. (2012, 2013) and Clagnan et al. (2019), who showed that imperfectly drained soils with high denitrification capacity can simultaneously buffer leaching and prolong N residence times.

Overall, the contrast between Curtins and JC highlights the dual nature of soil drainage controls: well-drained systems transmit management change rapidly but remain highly vulnerable to losses, while variably drained systems buffer groundwater quality yet accumulate long-lived legacy pools. From a policy perspective, these site-specific lag responses suggest that uniform six-year reporting cycles under the EU WFD are insufficient to capture the true pace of recovery, especially in variably drained catchments. Adaptive, soil- and drainage-based targets are needed to align environmental monitoring with realistic recovery horizons.

4.6 Conclusions

This study demonstrates how long-term management intensity, soil properties, and hydrology interact to determine N accumulation and transport in contrasting dairy systems. At Curtins, high historical surpluses (>300 kg N ha^{-1} yr^{-1}) generated substantial but labile, mineralizable legacy pools (~3,000–6,500 kg N ha^{-1} in the top 50 cm) that rapidly leached to groundwater through permeable loam soils and karstic pathways, producing NO_3-N peaks > 50 mg L^{-1} in the early 2000s but declining below the EU drinking-water limit (11.3 mg L^{-1}) by 2024 once inputs fell. At Johnstown Castle, lower surpluses (~100–150 kg N ha^{-1} yr^{-1}) nonetheless produced larger accumulations (~4,000–11,000 kg N ha^{-1}), reflecting the influence of finer-textured soils with higher silt and clay content, shallow water tables, and moderate denitrification potential. These characteristics enhanced retention through organo-mineral binding, NH_4^+ fixation, and reduced macropore flow, buffering groundwater (mean ~ 3.6 mg L^{-1} NO_3-N) but fostering persistent, slowly mineralizing legacy stores likely to release reactive N over decades.

Collectively, these findings reveal that N accumulation is governed by multiple interacting controls including drainage class, soil texture, hydrogeology, water-table depth, and denitrification potential rather than any single factor. Curtins-type systems potentially act as

"fast responders," transmitting surpluses quickly but recovering rapidly when inputs are reduced, while JC-type systems likely function as "slow retainers," protecting groundwater in the short term but accumulating legacy N that delays long-term recovery.

From a management standpoint, well-drained sites should prioritize fertilizer and stocking-rate reductions to deliver immediate groundwater-quality gains, whereas variably drained systems require longer-term strategies that stabilize and gradually draw down soil N stores. For policy, recognizing these contrasting pathways is critical: the EU WFD's six-year reporting cycles are too short to capture recovery in slow-responding catchments. Integrating legacy N estimates, site-specific drainage classification, and extended soil and groundwater monitoring into national environmental assessments will enable realistic recovery timelines and targeted, evidence-based mitigation strategies across Ireland's dairy landscapes.

Ultimately, these contrasting site responses illustrate the need to move beyond one-size-fits-all N regulations and to align farm management, soil monitoring, and policy design with the long-term biogeochemical realities of grass-based dairy systems.

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Chapter 5

Conclusions

5.1 Knowledge Gap and Objectives

Nitrogen is central to agricultural productivity but its mismanagement drives nitrate (NO_3^-) and ammonium (NH_4^+) contamination of waters, gaseous emissions to air and overall environmental decline. In Ireland's grass-based dairy systems, high fertilizer inputs and stocking rates sustain milk yields but also generate persistent N surpluses. These surpluses contribute to nutrient losses that threaten groundwater quality, aquatic ecosystems, and climate targets, particularly in intensively farmed regions of the south and southeast. However, impacts vary regionally due to differences in soils, hydrology, and farm intensity, creating uneven pressures on water and air quality.

Ireland now stands at a crossroads; the drive for agricultural productivity must be balanced against the need to restore and protect environmental quality. Yet, policy responses remain largely reactive, often lagging behind emerging scientific understanding of N dynamics and legacy effects. This thesis addresses this gap by quantifying soil N accumulation, explaining the lag between improved N management and water quality recovery, and informing forward-thinking mitigation strategies.

While groundwater NO_3^- contamination has been widely studied, soil N accumulation remains poorly quantified, even though it represents a substantial store that can continue releasing Nr for years or decades. These soil and subsurface legacies explain much of the temporal lag between changes in management and measurable recovery in water quality. Understanding these processes is essential for developing realistic mitigation timelines and informing effective policy design.

This thesis set out to investigate how N accumulates, is stored, and moves within intensive dairy systems, linking farm management practices to both short-term and long-term environmental outcomes. Specifically, it aimed to

(i) simulate the impact of alternative management practices on N surpluses and associated losses to air and water under derogation conditions;
(ii) quantify legacy soil N accumulation across contrasting sites with well-drained (Curtins) and variably drained (Johnstown Castle) soils, assessing how these pools differ in magnitude, depth distribution, and persistence; and
(iii) estimate multi-decadal farm-scale N surpluses using fine-resolution farm management data, linking historical inputs and outputs to long-term patterns of soil accumulation.

Together, these objectives provide a framework for understanding how both short-term management actions and long-term biophysical processes influence N cycling and legacy accumulation in intensive dairy systems, thereby informing strategies for sustainable N use. The most novel contribution of this work lies in the quantification of legacy soil N, a major but previously under-measured component of agricultural N budgets. While many studies acknowledge the presence of legacy N, few have directly quantified its magnitude or persistence. This research demonstrates that these soil reserve pools represent a substantial, long-lasting contributor to N dynamics and must be explicitly considered in policy design, predictive modelling, and farm management practices. Accounting for soil legacy N is essential for realistic predictions of water quality recovery, effective mitigation planning, and the efficient use of existing soil N stores as a resource to support both environmental and economic sustainability.

5.2 Synthesis of Findings

The three research strands presented in this thesis offer complementary insights into N dynamics in Irish dairy farming systems. By combining model simulations, long-term data analysis, and targeted soil investigations, the work provides a multi-scalar perspective on how N is cycled, lost, and stored.

5.2.1 Short-term management interventions (Chapter 2)

Chapter 2 modelling showed that farm-level management can significantly reduce N losses under derogation conditions. Scenarios tested with the €riN-MDSM model included fertilizer rate reductions, substitution of calcium ammonium nitrate (CAN) with protected urea, and restrictive grazing during winter–spring.

Restrictive grazing was the most effective measure, cutting NO_3^- leaching by 38–44 % and bringing losses close to the 30 kg N ha^{-1} water quality target. However, it also increased

slurry storage needs, ammonia emissions (NH_3), and potential welfare concerns, requiring complementary strategies such as improved housing and slurry management.

Fertilizer adjustments also yielded clear benefits. Reducing inorganic N from 250 to 200 kg N ha^{-1} cut NO_3^- leaching by 27 % and NH_3 volatilization by 31 %, while replacing CAN with protected urea lowered NH_3 losses by 5–6 %. The greatest reductions occurred when these measures were combined with moderate stocking rates (170–230 kg N ha^{-1}).

These results highlight both opportunities and trade-offs. Restrictive grazing provides major water quality benefits but must be managed alongside emissions and infrastructure challenges. Fertilizer reductions and substitutions add further gains, but no single measure is sufficient. Balanced outcomes will require integrated packages that combine restrictive grazing, lower fertilizer inputs, and protected urea—protecting both air and water quality while sustaining farm viability.

Going forward, this type of scenario analysis can guide future systems trials on Irish research farms to test which combinations deliver the best environmental and economic outcomes. For policymakers, this work demonstrates how modelling can inform the design of evidence-based regulations and targeted support schemes, identifying where specific practices (e.g., protected urea use or grazing restrictions) will have the greatest impact. For the dairy industry, it provides a direction of travel—toward systems that optimize fertilizer efficiency and reduce environmental losses while safeguarding farm profitability.

5.2.2 Multi-decadal nitrogen surpluses and legacy accumulation (Chapter 3)

At Curtins (well drained site), long-term records and soil analysis confirmed that persistent surpluses from high fertilizer and manure inputs built large legacy N pools with measurable effects on groundwater. Over 24-year study period, annual fertilizer N inputs ranged from approximately 200 to 350 kg N ha^{-1}, while manure and slurry applications added a further 60 to 120 kg N ha^{-1}. Grass N uptake varied between 350 and 480 kg N ha^{-1} yr^{-1}, leading to paddock-level N surpluses that frequently exceeded 200 kg N ha^{-1} yr^{-1}, driving cumulative imbalances stored in soils. A total of 75 soil cores were collected across 15 paddocks managed under varying N input management, and total N accumulation in the upper 50 cm ranged from approximately 4,000 to 5,500 kg N ha^{-1}. This confirms that long-term management intensity was a major driver of subsurface N storage and highlights the persistence of legacy N in well-drained dairy systems.

Groundwater monitoring revealed a delayed but persistent response to changes in N management. NO_3-N loads declined from over 70 kg N ha^{-1} in the early 2000s to just under 30 kg N ha^{-1} by 2024, but concentrations have since plateaued rather than continuing to fall. This stabilization occurred despite further reductions in fertilizer inputs and farm surpluses, indicating that perhaps groundwater NO_3-N is now sustained by the gradual release of N accumulated in subsoils during previous decades. The persistence of high total and potentially mineralizable N in deeper soil layers, together with limited denitrification capacity in well-drained soils, supports the interpretation of a legacy N signal rather than ongoing surface inputs.

Soil cores revealed significant subsoil accumulation, particularly between 25–50 cm depth, with total N exceeding 5,000 kg N ha^{-1} in high-input paddocks. Quadrant analysis showed that paddocks with both historically and currently high surpluses held the largest stores, but even reduced-input paddocks retained substantial legacy pools.

These findings highlight why water quality recovery lags behind farm practice changes. Once accumulated, subsoil N remains available for leaching and mineralization for decades, constraining the speed of recovery and potentially creating the impression of policy failure. Reducing current surpluses is necessary to prevent further accumulation, but legacy pools will continue to drive NO_3^- losses. Long-term groundwater monitoring, deep soil profiling, and in-field mineralization studies are essential to track and manage these legacies.

5.2.3 Soil drainage class and nitrogen dynamics (Chapter 4)

The comparison of Curtins (well-drained, low denitrification) and Johnstown Castle (variably drained, variable denitrification) highlights how drainage class and hydrological setting shape N storage and recovery. While accumulation largely reflects historical surpluses, whether this N is retained or transmitted to groundwater is strongly mediated by drainage, denitrification, water table depth, and management.

The comparative analysis of the Curtins and Johnstown Castle sites demonstrates how soil drainage, texture, and hydrological setting govern both the storage and fate of legacy N in intensive dairy systems. At the well-drained Curtins site, historically high surpluses exceeding 300 kg N ha^{-1} yr^{-1} led to the formation of substantial legacy pools of 3,000–6,500 kg N ha^{-1} within the top 50 cm. These values sit at the upper end of those reported for temperate agroecosystems: Van Meter et al. (2016) estimated cumulative N retention of ~1,400 kg N ha^{-1}

over 50 years in U.S. basins, Zhou et al. (2016) observed 450–2,150 kg N ha^{-1} in 0–4 m profiles of semi-humid croplands in China, and Liu et al. (2022) reported up to 7,100 kg N ha^{-1} within 10 m in intensively fertilized orchard systems. The magnitude of accumulation at Curtins highlights the exceptional capacity of well-drained grassland soils to retain reactive N (Nr) after decades of high inputs. Subsequent mineralization and leaching of this stored N produced pronounced groundwater NO_3-N peaks (>50 mg L^{-1}) in the early 2000s, but concentrations declined rapidly once fertilizer use decreased, falling below the EU maximum allowable concentration (11.3 mg L^{-1}) by 2024. This pattern confirms that well-drained systems are highly vulnerable to leaching yet can respond relatively quickly when management improves.

At Johnstown Castle, lower surpluses (~100–150 kg N ha^{-1} yr^{-1}) still generated larger accumulations of 4,000–11,000 kg N ha^{-1} in the top 50 cm; considerably higher than legacy N stocks reported in most other agricultural systems. Comparable studies have documented accumulations ranging from ~450 kg N ha^{-1} in semi-humid croplands (Zhou et al., 2016) to ~1,400 kg N ha^{-1} over 50 years in U.S. basins (Van Meter et al., 2016), and 6,000–7,000 kg N ha^{-1} across 0–10 m in intensively managed orchards (Liu et al., 2022; Nakagawa et al., 2021). One likely contributor to these exceptionally large legacy pools at JC is its finer soil texture, with relatively high silt and clay contents that enhance N retention through multiple mechanisms: binding of organic N to mineral surfaces, increased cation-exchange capacity, ammonium fixation, and reduced macropore flow. Studies have shown that total soil N tends to rise with clay content even as mineralizable fractions decline (Geisseler et al., 2024), and that clay minerals mediate N transformations and long-term storage (Yuan et al., 2025). Ilampooranan et al. (2022) similarly highlight that such conditions in intensive agricultural soils promote persistent organic N accumulation and delay water-quality recovery. These properties reduce N mobility and help explain the long-term retention observed at JC, where imperfect drainage, shallow groundwater, and moderate denitrification restrict NO_3-N transport (mean ~3.6 mg L^{-1}) but promote slow mineralization of stored N over decades. Consequently, JC-type systems trade short-term water-quality protection for prolonged legacy persistence.

Simple uptake estimates suggest Curtins' legacy (~5,000 kg N ha^{-1}) could sustain grass for ~12 years without fertilizer, or ~5 years assuming 40 % availability, while JC's legacy (~7,000 kg N ha^{-1}) could sustain growth for ~17 years (~7 years at 40 %). These figures highlight both the buffering capacity of legacy N and its potential use as a transitional nutrient source.

The study showed that site-specific factors such as soil texture, denitrification (inherent natural attenuation capacity of the soil, subsoil, bedrock at the sites, water table), soil drainage class, artificial drainage systems and management strongly influence accumulation. Overall, Curtins-type systems act as "fast responders", transmitting surpluses quickly but recovering rapidly once inputs decline, while JC-type systems are "slow retainers," buffering groundwater but prolonging recovery.

For management, these findings imply that well-drained sites should prioritize reducing fertilizer and stocking rate inputs to deliver relatively immediate improvements in groundwater quality, whereas variably drained systems require longer-term strategies that not only limit new inputs but also stabilize and gradually draw down existing soil N stores. For implementation under the EU Nitrates Directive (ND) and Water Framework Directives (WFD), this distinction is critical. Current policy cycles and reporting frameworks typically operate on short 6-year intervals, yet the persistence of legacy N in soils and groundwater means that measurable water-quality improvements in many variably drained catchments will unfold over much longer timescales. Consequently, short-term reporting may underestimate progress or misrepresent delayed recovery as policy failure. To address this, WFD implementation in Ireland should incorporate differentiated timelines and indicators that reflect site-specific hydrogeological conditions, explicitly recognizing the lag effects of legacy N. Integrating soil- and drainage-based classification into monitoring and target setting would enable more realistic expectations of recovery, better alignment of agri-environmental supports, and ultimately more credible progress toward WFD objectives.

5.2.4 Overall synthesis

The three strands of this thesis converge on a central insight: sustainable N management in intensive dairy systems requires both immediate interventions and action on long-term legacies, with outcomes strongly conditioned by soil and site characteristics.

Short-term modelling confirmed that measures such as restrictive grazing, reduced fertilizer rates, and protected urea can substantially cut N surpluses and leaching, though always with trade-offs in emissions, infrastructure, or costs. Multi-decadal analysis showed that historical surpluses have built large legacy pools that continue to influence groundwater even after inputs decline, potentially explaining the plateau in recent water quality improvements. The comparative site analysis indicates that N accumulation and release are governed by the interaction of multiple factors. At the well-drained Curtins site, low

denitrification potential combined with rapid percolation and limited water retention led to the faster transmission of NO_3-N to groundwater and a more immediate response to management change once inputs declined. In contrast, at Johnstown Castle; where imperfect drainage coincides with higher clay content, a shallower water table, and greater denitrification potential; N mineralization is suppressed and legacy pools accumulate more extensively within the soil profile making this site a slower responder to management change, as stored N is released gradually over time. Together, these findings highlight that N accumulation and recovery dynamics reflect the combined influence of soil drainage, texture, hydrology, and denitrification potential, rather than a single controlling factor.

Taken together, the findings show that reducing current surpluses is essential to prevent further accumulation, but water quality recovery will be gradual and uneven due to legacy effects and site-specific conditions. One-size-fits-all approaches are inadequate. Instead, management and policy must combine short-term efficiency measures with long-term strategies for legacy drawdown, tailored to drainage class and hydrological setting. Recognizing this temporal disconnect is crucial: farmers need confidence that their efforts will be valued despite delayed responses, policymakers must set realistic expectations, and scientists must continue refining tools that capture both current and legacy N dynamics.

5.3 Original Contributions of the Thesis

This thesis makes several novel contributions to the understanding of N dynamics in grass-based dairy systems, with implications for science, policy, and practice:

- **Scenario analysis of mitigation strategies in derogation systems.** Using the €riN-MDSM model, this thesis provided one of the first systematic evaluations of restrictive grazing, fertilizer and slurry reductions, and substitution of CAN with protected urea. The work highlighted both the mitigation potential and the scale of baseline losses in intensive systems under derogation, setting the stage for legacy N investigations.
- **Quantification of legacy soil nitrogen across contrasting drainage and hydrogeological settings.** A major innovation of this thesis is the direct quantification of legacy soil N pools across both well-drained and variably drained systems; an area previously underexplored in temperate dairy regions. By combining deep soil coring (75 cores across 15 paddocks- Curtins; 45 cores across 9 paddock-Johnstown Castle), multi-decadal management data, and groundwater monitoring, the research demonstrates that soil N accumulation in the upper 50 cm can reach 3,000–11,000 kg

N ha^{-1}, depending on site conditions. The findings show that while drainage strongly mediates N transport and release, accumulation is a multi-factor response governed by soil texture, hydrogeology, water table depth, and denitrification potential. This represents one of the first empirical estimates to capture the magnitude and variability of legacy soil N in Irish grassland systems.

- **Integrating soil texture and drainage into conceptual models of nitrogen legacies.** The comparative analysis between Curtins and Johnstown Castle established a new conceptual framework linking drainage class, denitrification potential, soil texture, and hydrology to legacy N dynamics. Well-drained systems with low denitrification potential act as "fast transmitters" of N, vulnerable to leaching but responsive to improved management; while variably drained, finer-textured systems function as "slow retainers," where higher denitrification potential and limited percolation suppress mineralization and promote long-term storage. This refined framework advances existing conceptual models by showing that N legacies emerge from interacting physical and biogeochemical processes.
- **Reframing legacy nitrogen as both risk and resource.** Legacy pools were shown to sustain grass growth for 5–7 years under reduced fertilizer use, suggesting a pathway to lower inputs while maintaining production. This positions legacy N as a potential economic and environmental win–win for farmers and policymakers. It supports the design of adaptive management approaches that gradually draw down soil N stores while maintaining productivity and also reducing N losses.
- **Implications for modelling, monitoring, and policy design.** The thesis highlights how legacy N processes must be explicitly represented in farm-scale models, water quality assessments, and regulatory frameworks. It shows that current policy reporting cycles (e.g., the EU WFD's 6-year horizon) are too short to capture recovery in areas with substantial legacy pools, risking misinterpretation of delayed progress as policy failure. The integration of soil- and drainage-based indicators into both modelling and monitoring frameworks would allow more realistic recovery timelines, targeted mitigation, and credible evaluation of environmental outcomes. Future policy should adopt a dual-monitoring approach that combines water quality data with direct soil-based assessments of legacy N. Regular soil coring and regional estimation across different soils and drainage classes would provide the evidence needed to set realistic environmental targets, refine farm N management, and calibrate predictive models.

Integrating soil legacy N monitoring into national programs would improve policy credibility and ensure mitigation measures reflect the true recovery potential of Irish agroecosystems.

Collectively, these contributions establish this thesis as a foundational step in quantifying and interpreting legacy soil N in intensive dairy systems. By linking multi-decadal management to soil accumulation across contrasting drainage classes, this work establishes a new framework for understanding short- and long-term N processes and provides evidence to guide predictive modelling and policy design.

5.4 Limitations

While this thesis provides new insights into N dynamics in dairy systems, several limitations must be acknowledged.

- **Modelling limitations (Chapter 2).** The €riN-MDSM model was valuable for scenario analysis but has several structural and operational constraints. It operates on an annual time step, limiting its ability to capture short-term interactions between rainfall, soil moisture, and N transformations. Built in Excel, the model is computationally heavy and unstable at scale, restricting its scalability and integration with other tools. To realize its full potential, it should be migrated to a coded version that can interface with the Soil Moisture Deficit (SMD) model and the Agri-Forecast platform. Such development would enable dynamic simulations of fertilizer timing, weather events, and management decisions, allowing it to evolve into a robust on-farm and policy decision-support tool.
- **Empirical site limitations (Chapters 3 and 4).** Soil and groundwater investigations were based on a small number of research sites without any replicated natural cores, limiting statistical generalizability. Reconstruction of multi-decadal farm management also required backfilling of missing data and assumptions about drainage, fertilizer application, and slurry distribution, which introduce uncertainty into surplus estimates and their link to groundwater outcomes.
- **Soil coring and analytical constraints.** Soil sampling was limited to 1 m depth, meaning legacy N stored below this horizon was not captured. Future work should extend coring to greater depths and across sites with contrasting management, drainage, and hydrogeological settings to develop a nationally representative understanding of

legacy N accumulation. Such data would provide critical evidence for regional and national N budgeting and improve integration of legacy N into global nutrient models and policy frameworks. Laboratory-based analyses may have altered mineralizable fractions compared with in-situ dynamics, potentially under- or overestimating available pools. To address this, future studies should incorporate in-field mineralization assessments and explore spectroscopy-based approaches to rapidly estimate total and mineralizable N across larger sample sets. If successfully calibrated using existing datasets, spectroscopy could substantially improve throughput and spatial coverage of soil legacy N monitoring.

- **Temporal and climatic scope.** While multi-decadal groundwater monitoring captured lagged responses, disentangling short- and long-term effects remains difficult. This thesis did not extend to groundwater modelling, which is essential for linking soil N legacies to NO_3-N transport pathways and predicting recovery timelines at catchment and regional scales. Future work should integrate soil legacy data with mechanistic groundwater models to simulate N movement through the unsaturated and saturated zones, enabling more accurate forecasts of water quality responses. Similarly, upscaling from site to landscape level will require combining farm datasets, soil surveys, and hydrological mapping to quantify legacy N patterns across Ireland. Climate change through shifts in rainfall, temperature, and extreme events will further influence N cycling and mobilization, reinforcing the need for coupled soil–groundwater–climate modelling frameworks to inform adaptive policy and management.

Despite these limitations, this thesis advances methods for integrating multi-decadal management data with soil N accumulation. Many of these constraints directly inform priorities for future research, including model refinement, greater replication, deeper coring, and closer integration of soil–groundwater–climate processes.

5.5 Implications for Science, Policy, and Industry

The findings of this thesis carry important implications across research, policy, and practice, as described below.

- **Scientific implications**

  This research advances understanding of the nitrogen cascade in agricultural systems by empirically quantifying legacy N pools and linking them to multi-decadal surpluses

and water quality. It demonstrates how past management exerts long-term influence on current N cycling and shows that drainage class is one of several interacting factors—alongside texture, hydrogeology, and management which together shape the expression of these legacies. Methodologically, the integration of farm budgets, soil coring, and groundwater monitoring provides a transferable framework for studying N accumulation and release. Laboratory incubations and spectroscopy-based analyses also highlight the potential to upscale this work using national soil spectral libraries, enabling rapid estimation of total and mineralizable N across regions. Together, these advances strengthen the evidence base for integrating soil legacy N into national monitoring, modelling, and budgeting frameworks.

- **Policy implications**

  The findings are directly relevant to the EU ND and WFD. Chapter 2 showed that targeted management—such as restrictive grazing, protected urea, and improved slurry handling can reduce N losses without compromising productivity. However, the persistence of legacy N means that current WFD reporting cycles, typically six years, are too short to capture recovery, especially in variably drained areas where delayed release from soil stores masks policy progress. Recognizing legacy dynamics within WFD reporting will help Ireland set realistic recovery timelines and avoid penalizing areas where slow improvement reflects biophysical constraints rather than policy failure. Differences in drainage class, hydrogeology, and management intensity further indicate that one-size-fits-all rules are ineffective; instead, measures must be site-specific. Expanding soil and groundwater monitoring across regions would improve detection of legacy N effects and provide the evidence base needed to design realistic environmental targets and adaptive nutrient management policies.

- **Industry implications**

  For farmers, legacy N represents both a challenge and an opportunity. Large soil N pools will continue to influence water quality for decades, but they can also serve as a nutrient source to sustain grass growth under reduced fertilizer inputs. By strategically drawing on these reserves, farmers can lower costs and reduce losses, achieving both economic and environmental gains. For policymakers and industry leaders, recognizing this dual role opens the door to measures that explicitly incentivize the gradual drawdown of legacy N while preventing further accumulation. Programs of Measures

(PoMs) must therefore be tailored to different farm types, soil drainage classes, and legacy loads rather than applying uniform requirements. Sustaining farmer motivation is also critical: clear communication about lag times, coupled with evidence-based advisory support, will help maintain trust in environmental programs even when progress appears slow. Building this understanding within the industry is essential to align farm-level decisions with long-term water quality and sustainability goals.

5.6 Future Research Directions

The findings and limitations of this thesis point to several priorities for future research on N management, legacy accumulation, and water quality outcomes in grass-based dairy systems:

- **Refine modelling tools.** The €riN-MDSM model should be developed to capture weather variability, urine composition, and interactions between fertilizer/slurry timing and soil conditions. Integration with groundwater transport models would better represent lag times and natural attenuation, improving water quality predictions under mitigation scenarios. Future modelling should assess combined mitigation strategies under realistic economic and regulatory conditions, including reduced derogation limits, increased slurry storage, and the adoption of clover and multi-species swards.
- **Expand replication across sites.** Moving beyond case studies to multi-site, replicated trials is essential for statistical robustness. Including a wider range of management practices and drainage classes would make legacy N estimates more representative at national scale.
- **Improve reconstruction of management histories.** Future work should strengthen multi-decadal N budgets by linking national survey data (e.g., Teagasc National Farm Survey) with detailed field-scale records, reducing reliance on assumptions and backfilling.
- **Extend soil coring and develop in-situ monitoring.** Deeper sampling beyond 1 m is needed to capture the full extent of legacy accumulation, particularly in well-drained soils. In-situ measurements of mineralization and N availability would reduce biases from laboratory processing and refine estimates of active legacy pools.
- **Integrate climate change considerations.** Shifts in rainfall, temperature, and extreme events will alter N cycling and legacy mobilization. Long-term experiments combining legacy monitoring with climate scenarios are needed to inform adaptation strategies.

Together, these priorities would strengthen the scientific, practical, and policy relevance of future N management research.

5.7 Closing Reflection

This thesis has demonstrated that N management in intensive dairy systems is governed by both present-day practices and the long shadow of historical surpluses. Modelling confirmed that short-term interventions such as restrictive grazing, reduced fertilizer applications, and the adoption of protected urea can substantially reduce losses to water and air, though these gains come with trade-offs that must be managed through improved infrastructure and integrated nutrient planning.

Long-term analyses revealed that decades of surpluses have created substantial legacy soil N pools in both well-drained and variably drained systems, explaining why groundwater NO_3^- improvements have plateaued despite recent input reductions. Comparative site studies further showed that hydrology, soil texture, drainage, denitrification potential, and management interact to shape how N is stored, mineralized, and released, demonstrating that accumulation reflects the interaction of multiple system factors.

A central breakthrough of this research is the quantification of legacy soil N—an under-measured but critical component of agricultural N budgets. These findings establish that legacy N is not only a constraint on recovery but also a potential resource: by strategically drawing down these soil reserves, farmers can reduce fertilizer dependence while sustaining grass growth, achieving both economic and environmental benefits. Policymakers and regulators can use this knowledge to design realistic mitigation timelines, refine model predictions, and target soil-based monitoring to better capture legacy effects across regions.

Ultimately, sustainability in dairy systems requires patience, persistence, and coordination across science, policy, and practice. Farmers must continue improving N efficiency; policymakers must set evidence-based, time-sensitive goals; and scientists must further refine models and expand soil and groundwater monitoring to track legacy dynamics. Together, these efforts provide a pathway for aligning agricultural productivity with long-term water quality and climate objectives, ensuring that the dairy sector remains both economically viable and environmentally resilient.

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Appendix

Table A. 1 Borehole information for Curtins Moorepark Dairy Research Farm.

(Cross Reference- Chapter 3, Figure 3-1; Chapter 4, Figure 4-1)

Well ID	Easting	Northing	Total Depth (m)	Surface Elevation (m AOD)	Mean WT Depth (m)	Top Screen Elevation (m AOD)	Bottom Screen Elevation (m AOD)	Piezometer (P)/ Open uncased Well (O)
1	180877.4	100677.8	35	55.24	9.67	27.24	23.24	P
2	181241.1	100731.1	39	54.21	9.58	22.21	18.21	P
3	181413.4	100869.7	59.5	53.59	9.64			P
4	181289.8	100953.2	43.5	54.45	8.88			O
5	181200.4	101030.5	33	52.17	8.57	26.17	23.17	P
7	181492.6	101077	35	54.26	15	25.26	22.26	P
8	181528.1	101176.4	49.5	55.13	10.94			P
9	181262.7	101428.1	34.5	52.43	6.6	23.00	19.00	P
10	181488.2	101441.1	34.5	56.01	4.02	27.51	24.51	P
11	181429.4	101464.8	39.9	55.51				O
12	181427.2	100823.7	46	54.21				O

Abbreviations: m AOD = meters above ordnance datum.

Table A. 2 Piezometer at Johnstown Castle Dairy Research Farm

(Cross Reference- Chapter 4; Figure 4-2, Figure 4-3)

Well ID	Easting	Northing	Total depth (m)	Surface Elevation (m AOD)	Mean WT Depth (m)	Screen Elevation (m AOD)	Top of screen (m AOD))
1	117347	302242.7	3.97	63.8	60.26	59.83	60.83
2	117300	302578.5	8.19	63.12	55.35	54.93	55.93
3	117146.2	302621.8					
4	117166.3	302669.6	6.47	58.77	53.3	57.43	58.43
5	116871.7	302722.8	2.95	53.42	52.27	52.3	53.3
6	116907.5	302811.9	4.48	53.58	52.68	50.47	51.47
7	116782.6	302826.8	3.04	49	48.14	49.1	50.1
8	116617.4	302674.5	7.74	48.2	43.58	45.96	46.96
9	117172.1	302211.5	6.92	60.44	55.77	40.46	41.46
10	117084.7	302323.2	8.62	59.89	54.35	53.52	54.52
11	117229.2	302349.2	3.59	57.82	56.15	51.27	52.27
12	117501.3	302553.6	3.59	61.91	60.34	54.23	55.23

13	117077.1	302707.7	4.96	57.2	54.79	58.32	59.32
14	117255.4	302473.1	5.55	60.65	56.36	52.24	53.24
15	116994.7	302567	5.47	55.91	52.2	55.1	56.1
16	116744	302738	5.86	48.5	43.46	50.44	51.44
17	116560.9	302810	7.2	44.08	42.26	42.64	43.64
18	116628.6	302950.3	4.24	45.44	44.46	36.88	37.88
19	116681.5	302581.3	2.64	47.03	45.13	41.2	42.2
20	116677.8	302576.6	4.43	47.15	45.03	44.39	45.39
21	116505	302682.6	7.19	44.73	39.19	42.72	43.72
22	116540.1	302287.8	4.03	47.51	44.46	37.54	38.54
23	116263.1	302499.3	5.88	43.34	39.23	43.48	44.48
24	116071.2	302471.2	5.19	34.66	33.02	37.46	38.46
25	116359.2	302787.2	5.84	39.16	35.68	29.47	30.47
26	116327	302694.7	4.65	36.28	33.02	33.32	34.32
27	117357.1	302174.1	6.96				
28	117439.5	302412.6	6.94	59.75	58.38	57.04	58.04
29	117084.1	302457.9	6.56	54.03	52.62	52.81	53.81
30	117084.9	302461.5	12	54.03	52.41	47.47	48.47
31	117084.9	302461.2	16.8	54.03	52.26	42.03	43.03
32	116209.3	302682.9	5.95	34.09	31.8	37.23	38.23
33	116213.9	302683.2	11.9	34.09	31.74	28.136	29.136
34	116212.4	302683.7	16.8	34.09	31.96	22.186	23.186
35	116726.9	302588.1	4.96				
36	116711.1	302574.7					
37	116706.4	302560.9	4.91				
38	116925.3	302643.8	37	59	53.39	40.99	41.99

Abbreviations: m AOD = meters above ordnance datum.