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Evaluating the role of scale in the sustainability of nutrient management of pasture based dairy systems

Thesis submitted to University College Dublin in fulfilment of the requirements for the degree of Doctor of Philosophy

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Submitted to University College Dublin, May 2020

Statement of Original Authorship

I hereby certify that the submitted work is my own work, was completed while registered as a candidate for the degree of Doctor of Philosophy, and I have not obtained a degree elsewhere on the basis of the research presented in this submitted work

Philip Murphy

May 2020

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PUBLICATIONS FROM THESIS

Planned publications from this thesis

Chapter Three: **Murphy, P. M.; Murphy, P. N. C.; Wall, D. P., (in prep.)**. A spatial comparison of paddock, field and farm scale nitrogen balance and use efficiency on commercial grass-based dairy farms. Expected periodical: *Agriculture Ecosystems and Environment.*

Chapter Four: **Murphy, P. M.; Murphy, P. N. C.; Wall, D. P., (in prep.)**. A spatial comparison of field and farm scale Phosphorus and Potassium balance and use efficiency on 15 Irish pasture-based dairy farms. Expected periodical: *Nutrient Cycling in Agroecosystems.*

Chapter Five: **Murphy, P. M.; Murphy, P. N. C.; Wall, D. P., (in prep.)**. Changes in soil fertility and the factors that affect it at paddock scale on commercial grass-based dairy farms. Expected periodical: *Nutrient Cycling in Agroecosystems.*

Chapter Six**: Murphy, P. M.; Murphy, P. N. C.; Wall, D. P., (in prep.)**. Herbage nutrient status at paddock scale as affected by soil and sward management factors on grass-based dairy farms. Expected periodical: *Grass and Forage Science.*

Refereed conference publications

Murphy P.M., Murphy P.N.C. and Wall D.P. An evaluation of nutrient balances at the whole-farm and field scale on 21 Irish dairy farms, Proceedings of the 27th European Grassland Federation General Meeting Cork 17th- 21st June 2018. Volume 23 Grassland Science in Europe (2018).

Murphy P.M., Murphy P.N.C. and Wall D.P. Evaluating the role of precision nutrient management on grass-based dairy farms. Proceedings of the IFS Agronomic Conference, 7-8 December 2017, held in Cambridge, UK. – Winner of the Brian Chambers award 2017

Agreed collaboration publications

Suggested primary authors: **Murphy P.M., Howes J[1]**. Calibrating a New Zealand model to an Irish dairy system, methodology and validation. [1]Fertilizer & Lime Research Centre, Massey University, Contact: J.Howes@massey.ac.nz

Peer reviewed journal publications

Nikoloski, S., Murphy, P., Kocev, D., Džeroski, S. and Wall, D.P., 2019. Using machine learning to estimate herbage production and nutrient uptake on Irish dairy farms. Journal of dairy science, 102(11), pp.10639-10656.

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LIST OF ABBREVIATIONS

Al	Aluminium
BAL	Balance
C	Carbon
Ca	Calcium
CAP	Common Agricultural Policy
CEC	Cation Exchange Capacity
CH_4	Methane
CO_2	Carbon Dioxide
CSA	Critical Source Area
CSO	Central Statistics Office
Cu	Copper
DAFM	Department of Agriculture Food and Marine
EC	European Commission
EEC	European Economic Community
EPA	Environment Protection Agency
EPRS	European Parliamentary Research Service
ESBN	European Soil Bureau Overview
EU	European Union
FAO	Food and Agriculture Organisation
Fe	Iron
FH2020	Food Harvest 2020
FW2025	Food Wise 2025
GAP	Good Agricultural Practices for Protection of Waters
GHG	Green House Gas
GIS	Geographical Information System
GPS	Global Position System
H2PO4	Dihydrogen Phosphate
IPCC	Intergovernmental Panel on Climate Change
K	Potassium
KT	Knowledge Transfer
LOI	Loss on Ignition
LU	Livestock Unit

Mg	Magnesium
MINas	Mineral Accounting System
Mn	Manganese
N_2	Nitrogen
N_2O	Nitrous dioxide
NAP	Nitrates Action Plan
ND	Nitrates Directive
NFS	National Farm Survey
NH_3	Ammonia
NH_4	Ammonium
NO	Nitrous oxide
NTC	Nutrient Transport Continuum
NUE	Nitrogen Use efficiency
OECD	Organisation for Economic Co-operation and Development
P	Phosphorous
PRG	Perennial Ryegrass
S.I.	Statutory Instrument
SOM	Soil Organic Matter
SR	Stocking rate
ST	Soil Test
TGA	Total Grassland Area
TUAA	Total Utilisable Agricultural Area
UE	Use Efficiency
WC	White clover
WFD	Water Framework Directive
Zn	Zinc

THESIS ABSTRACT

A review of the pertinent literature suggests that there may be an over reliance on farm scale assessments alone to improve within-farm nutrient management. In this thesis, a simultaneous assessment of farm and field scale nutrient balance was investigated to help bridge the gap in knowledge between farm and field scale nutrient management and potential nutrient losses, with a view to supporting the development of a more effective within-farm nutrient budgeting strategy or tool. Primary data was gathered from commercial grass-based dairy farms in the south and south east of Ireland. Nutrient use at the farm, field and paddock scale was characterised and evaluated simultaneously, to determine the sustainability of nitrogen (N), phosphorus (P) and potassium (K) management practices. Additionally, a spatially explicit precision nutrient advice strategy (including soil pH measurement, soil test for P (STP) and K (STK) concentrations), was implemented on the selected farms. The nutrient balance (BAL) i.e. nutrient source pressure, and use efficiency (UE) indicators, were evaluated across these farms and spatial scales from 2015 to 2017, while changes in soil fertility were monitored from 2015 to 2018. As well as this, herbage nutrient concentration was assessed through spring, summer and autumn of 2015 and 2016 at the paddock scale. Mean nitrogen use efficiency (NUE), and phosphorus use efficiency (PUE), were found to be significantly lower ($p < 0.05$) at farm scale (NUE: 28% and PUE: 52%), compared to results at field scale (NUE: 57% and PUE: 100%). Mean potassium use efficiency (KUE), was found not to be significantly different between these scales (KUE: 56% and 91%, respectively). With regard to nutrient source pressure, phosphorus balance (PBAL) was found to be significantly higher ($p < 0.05$) at the farm scale than at the field scale (9 and 6 kg P ha^{-1}, respectively), while the differences in nitrogen balance (NBAL) (165 and 139 kg N ha^{-1}, respectively) and potassium balance (KBAL) (23 and 39 kg K ha^{-1}, respectively) were not found to be significant between scales. Furthermore, variability (denoted by CV: Coefficient of Variation) in UE was found to be substantially larger between individual fields and paddocks (CV, 67%, 74% and 127% for N, P, and K, respectively) than between individual farms (CV, 39%, 33% and 76% for N, P, and K, respectively). Variability in BAL, was also found to be substantially higher between individual fields and paddocks (CV, 77%, 294% and 179% for N, P, and K, respectively) than between individual farms (CV, 34%, 58% and 86% for N, P, and K, respectively). Indicators of soil fertility (soil test values and soil optimum indices distribution) and indicators of nutrient management (BAL and UE) were found to develop towards more agronomically optimum levels. For example, mean results for paddocks across these farms showed that low fertility paddocks increased their soil pH to 6.3 or medium/adequate soil levels, while the mean results for high fertility paddocks show that STP decreased to 6.7 mg l^{-1}, and STK decreased to 142.9 mg l^{-1}, closer in range to agronomic optimum soil levels. Mean field scale PBAL and KBAL surpluses decreased significantly between 2015 and 2016 also (from 7 to 4 kg P ha^{-1} and 52 to 34 kg

K ha^{-1}). Finally, it was shown that the paddock scale herbage nutrient status strongly relies upon soil nutrient supply and soil fertility management. STK level was a major factor for herbage K and Calcium (η^2 = 0.26 and 0.11, respectively) while soil test Magnesium level was the major factor for herbage Magnesium (η^2 = 0.28) and STP level was the major factor for herbage P (η^2 = 0.35). Soil pH level was the main factor for herbage Manganese and Calcium (both η^2 = 0.15). The results presented in this thesis show that the scale at which nutrient use indicators are observed (e.g. farm or field/paddock), has a significant influence on the apparent nutrient efficiency and nutrient source pressure of specialised dairy farms. This has potentially important implications for agri-environmental policy makers, nutrient regulation and nutrient monitoring systems. Moreover, the variability found in nutrient efficiency and herbage yield and quality at field and paddock scale, has implications for farmers and knowledge transfer services (i.e. decision-making support and tools). The results also indicate that precision nutrient advice at paddock scale on specialised grass-based dairy farms, has the potential to improve nutrient spatial distribution. With improved spatial distribution, it would be expected that soil fertility levels, herbage quality, and nutrient balance and nutrient use efficiency would improve over time, at field scale and ultimately farm scale. This may have sustainability benefits, both socioeconomic (farm efficiency and profitability) and environmental (reduced source pressure for nutrient loss to the wider environment), as it is expected that appropriately tailored nutrient management plans would help to maximise the benefits and efficiency, and minimise the potential loss of nutrients. Furthermore, results suggest that improved management practices might differ between subzones of these dairy farms; notably, between the grazed area of the farm (milking platform) and the silage area. On the milking platform in particular, the paddock scale (ca. 0.5 – 2.5 ha) might be a suitable scale to target improved nutrient management practices, as these are already established as spatial management units. There are limitations for farmers to adopting this strategy due to the inherent difficulties and uncertainties relating to data collection. For example, field or farm area may change between each year which will invariably affect stocking rate. The large number of variables and number of changes that they are subject to make it impractical for farmers to calculate these nutrient balances by themselves. From a research perspective however, the approach does provide insight into nutrient cycling pinch points on pasture based dairy systems i.e. the where and why key nutrient pressures occur. This information can then be translated into practical advice by knowledge transfer agencies. Data recording software, although available in different capacities, does not yet support this approach in Ireland. Optimising nutrient management at this scale is likely to help improve the quality of environmental receptors such as air and water in the longer term. Future farm scale policy and regulations (e.g. stocking rate or fertiliser limits) need to be reinforced by efforts to address within-farm spatial variability which could help to reduce environmental losses.

CHAPTER ONE: GENERAL INTRODUCTION

1.1 Nutrient cycling in the agri-environment

On a planetary scale, agriculture cannot be uncoupled from the environmental problems that are associated with food production. Agriculture is one the largest consumers of water, one of the main sources of nitrate pollution of groundwater and surface water, and one of the principal sources of ammonia pollution at that scale. It is also a major contributor to phosphate pollution of waterways (OECD, 2012) and to the release of the greenhouse gases (GHGs) methane and nitrous oxide into the atmosphere (IPCC, 2019). Intensification of existing agricultural land has led to a significant contribution to air and water pollution (EU, 2016), and in some cases reduced productivity from land due to soil and water degradation (OECD, 2015).

To sustain milk and meat production for example, the long-term management of dairy farming systems must minimise and replace the loss of nutrients. Nitrogen, phosphorus, and potassium are nutrients that are typically imported in the highest quantities to meet the production requirements of grass-based dairy farms (Dillon et al., 2018). Interactions between the soil, plant and animal, make the management of these high nutrient quantities' problematic. Nutrient losses occur during these interactions due to soil characteristics, sward management and spatial distribution factors (McCormick et al., 2009; Diacono et al., 2013; Miao et al., 2011; Webb, 1989; Webb et al., 2000; Gillingham, 1973; Van Leeuwen et al., 2019). Much of the information on these interactions is derived from component studies that typically only examine a single environmental or management factor such as reseeding (Necpálová et al., 2013 and Shalloo et al., 2011), or sward composition (Bryant et al., 2017; Totty et al., 2013; Woodward et al., 2013). In some cases, environmental factors such as nutrient losses on different soil types and gradients can be evaluated together (Houlbrooke et al., 2003; Sharpley, 1996). However, evaluation of such factors in isolation, risks a loss of knowledge and understanding on how the interactions would occur in a real dairy system. This thesis explores the theory of using commercial farms as experimental units, with the view that, it would enable the results to be insightful, and potentially more applicable than for example, plot studies. An evaluation of commercial farms may be more reflective of the actual relationships and interactions between the soil plant and animal in a grass-based dairy system.

The identification of opportunities to minimise leakages in nutrient management practices are crucial for reducing their contribution to environmental impacts. However, the capacity to enable change at farm level, through policy and advisory efforts, may be limited to a significant extent, by differences in the scale at which observations and measurements are made and perceived. This thesis focuses specifically on the role of scale in nutrient management practices typically being implemented on commercial dairy systems.

1.2 Thesis aims and objectives

The aim of this research thesis was to evaluate the role of scale in nutrient management of grazed grass dairy farms so that knowledge on strategies/best practices could be enhanced for improved production and environmental sustainability of Irish dairy farms. The research was carried out on 21 commercial dairy farms in the south and south east of Ireland. Commercial dairy farms were selected as the experimental unit so as to allow the investigation of different scales, from sub-field to farm, and to allow the characterisation of differences across scales on real farming systems. For grassland systems evaluating from the smallest area of uniform management (paddock scale) may be a fundamental step in further developing precision nutrient management techniques so that the risk of nutrient losses to the greater environment can be further reduced.

The objectives were;

- To characterise and evaluate nutrient management (nitrogen, phosphorus, potassium nutrient balance) at the farm and field scale simultaneously and specific to soil characteristics (chemical and physical), grassland management (rotational grazing and pasture cropping, herbage yield, sward species) and other *in-situ* environmental factors (soil type and gradient), for nutrient use sustainability on commercial grass-based dairy systems
- To evaluate the potential effectiveness of spatial explicit field and paddock scale precision nutrient management advice on baseline soil fertility levels and soil fertility distribution (soil pH, soil test phosphorus and potassium), and identify the primary drivers of change in soil fertility at field/paddock scale on commercial grass-based dairy systems
- To characterise the herbage quality (nutrient concentration levels: nitrogen, phosphorus, potassium, Calcium, Magnesium, Copper, Manganese, Zinc) of commercial grass-based dairy farms at paddock scale, and evaluate the relationship between herbage quality, soil fertility, soil characteristics and sward management factors at this scale

An overview of the chapter layout and primary content of this thesis is shown in Figure 1-1.

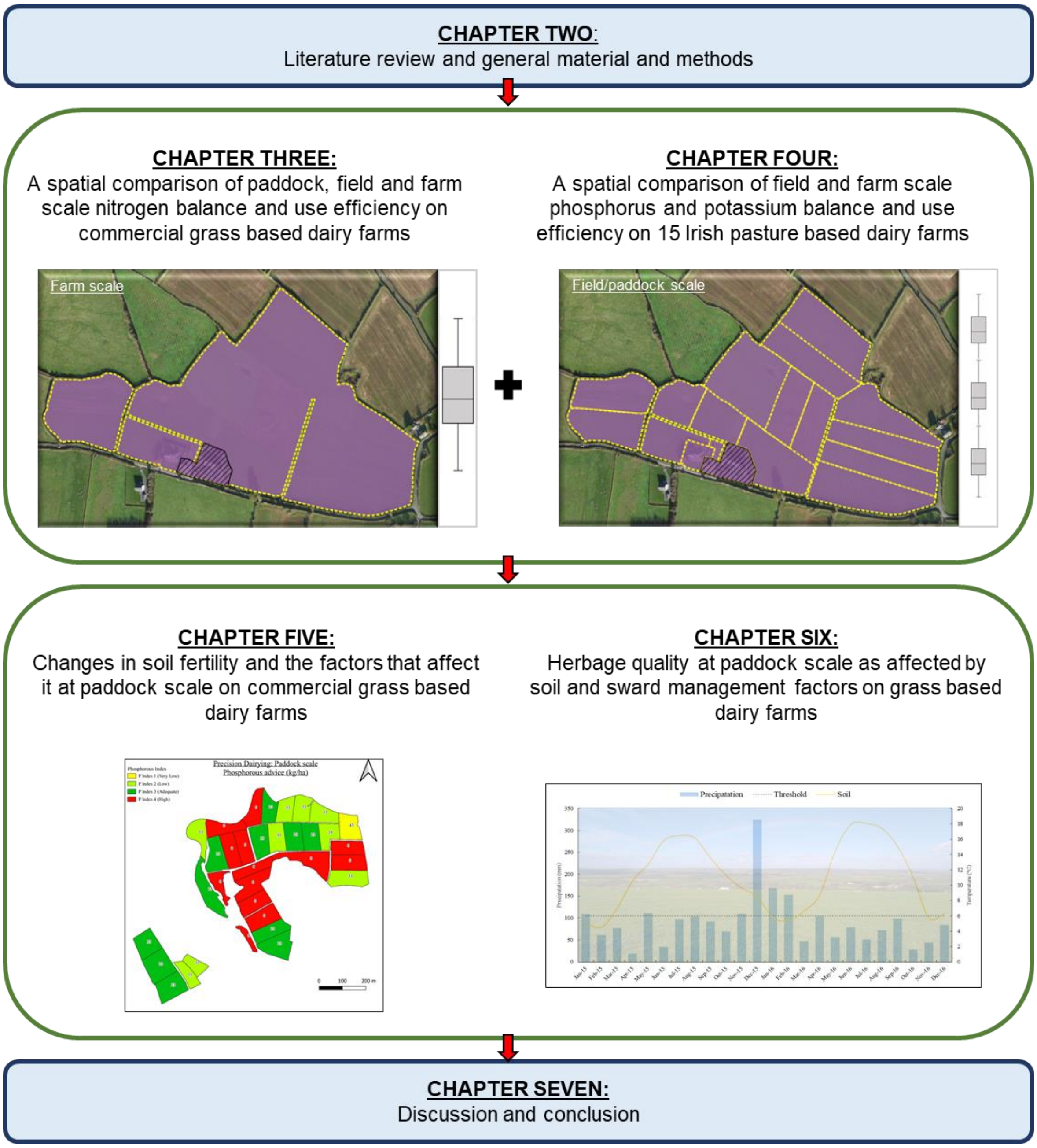

Figure 1-1. Flowchart illustrating the layout of thesis and primary content contained in each chapter thesis.

CHAPTER TWO: LITERATURE REVIEW AND GENERAL MATERIAL AND METHODS

2.1 Dairy systems

There are approximately 11 million farms in the European Union (EU) and 44 million people are employed in the entire EU food supply chain (EC, 2017a). The agri-food sector in Ireland, a member state of the EU, generated 7% of gross value added (€13.9 billion), 9.8% of Ireland's exports and 10% of national employment (including inputs, processing and marketing) in 2016 (DAFM, 2018). In the same year, there were 137,500 farms, average farm size was 32.4 ha and within this, there were approximately 15,639 Dairy farms (CSO, 2016; NFS, 2017). Distinctively, land cover statistics for the EU (Eurostat, 2018) estimate that Ireland has the largest proportion of area under grassland with 67.1%, while the EU average is just 19.5%. This is just one of the main characteristics that enables Ireland to have a substantial and economically important outdoor grazed grass dairy industry.

2.1.1 The rotational grazing dairy system

The technical characteristics of a grazed grass dairy system in Ireland (seasonal milk production) from 2017 according to an economic analysis of 1,568 farms (Teagasc, 2017) are as follows; Mean herd size was 117 cows and mean farm size was 52.0 hectares (ha). Stocking rate mean was 2.7 LU/ha (1 LU = 1 mature cow). Mean grazed grass utilised (estimated as ~75% of total herbage accumulation, Creighton et al., (2011)) was 9.3 tonnes dry matter ha^{-1}. Mean number of grazing rotations per field/paddock was 8, i.e. the number of times the herd re-grazed a field/paddock during a grazing season. Mean milk yield was 5,561 l cow^{-1} or 12,623 l ha^{-1} and mean milk solids was 444 kg cow^{-1} or 1,008 kg ha^{-1}. Mean concentrate feed per cow was 1,038 kg. In terms of fertiliser import levels, Dillon et al. (2018) showed that mean inorganic fertiliser import in 2015 for dairy farms was 129, 9 and 21 kg ha^{-1} for nitrogen (N), phosphorus (P) and potassium (K), respectively.

The grass growing season, dictated by seasonal soil and weather conditions, is the main driver of system management strategies. Calving is typically in late winter or early spring (February - March) for a spring dairy system while the calving period is typically in autumn (September to October) for a winter dairy system (O'Donovan and McEvoy, 2016). In 2018, there were approximately 1800 registered (and an estimated 600-800 producers under various unregistered price incentive schemes) winter dairy systems in the Ireland (Patton and Lawless, 2019). They supplied just over 14% of national production (1.1bn litres), of which 45% (490m litres or 6% of total national production) was used for domestic consumption (Patton and Lawless, 2019). It is estimated that a further 600 - 800 farms nationally produce winter milk under various non-registered price incentive schemes. Grazed grass and supplementary concentrate feed are the main source of nutrition throughout the spring and summer months (April – October) until winter housing. Winter forage (grass silage) is used to bridge the gap in time between one growing season and the next, because soil temperatures, and other climatic factors, slow down grass growth substantially through winter months (November – January) and soil conditions are often too wet for animals to traffic. Animals are typically housed over these winter months. Weather permitting, two major harvests of silage are taken in late spring and summer (May and July) (Figure 2-3), often from areas set aside for this purpose, to meet the herd feed demand during the housing period. Additional harvests may be taken as a grass utilisation practice. This system description characterises the low-cost grazed grass seasonal milk production system found in Ireland (Shalloo et al., 2018), particularly in the southern half of the country.

2.1.2 Other dairy systems in the EU

In a European context, the main producers of milk are Germany, France, the United Kingdom (UK), the Netherlands, Poland, Italy and then Ireland. When considered together, they account for three quarters of total EU production. There are however substantial contrasts in dairy systems between EU countries. For example, in 2016, the EU average milk production per cow was 7,021 l cow^{-1} but many countries had higher yields than this; 9,504 l cow^{-1} in Denmark, followed by 8,222 l cow^{-1} in the Netherlands, 7,574 l cow^{-1} in Germany, 7,152 l cow^{-1} in the UK and 6,963 l cow^{-1} in France (EPRS, 2018). High milk yields were recorded also in Finland and Sweden (9,063 and 9,039 l cow^{-1}, respectively).

In comparison to management strategies in other countries, estimates indicate that 98% of cows in Ireland have access to pasture grazing, while only 10% in Greece and almost zero in Bulgarian have access, highlighting one such significant difference between EU farms (Van den Pol-Van Dasselaar et al., 2015). Data for the UK in 2015, indicated that 83% of milking cows, grazed outdoors for between 21 and 52 weeks of the year, with only 5% of farms not providing any grazing outdoors. The use of outdoor grazing is influenced by factors such as, soil type, topography, climate, farm infrastructure, but also factors such as, social pressure, farm economics, farmer skill level and motivation, have an important influence (EC, 2017b). Data on grazing access per country is limited overall.

2.1.3 Dairy systems outside the EU

Globally, the five largest milk producers during 2015 to 2017 was firstly the EU, with a 20% share in global production, followed by India (20%), the United States (12%), Pakistan (6%), and China (5%) (OECD-FAO, 2018). The share Oceania contributes to world milk production was only 3.8% during 2015 to 2017. Over the next decade, most of the growth is expected to come from a further increase in the dairy herd (0.2%) and yield (1.3%) (OECD-FAO, 2018).

2.1.4 Milk quota abolition in Ireland

In 1984, the EU (formerly the European Community), introduced a milk quota system across all member states, to prevent surpluses which were occurring through oversupply of milk products. However, due to a projected increase in demand for dairy products, over 30 years later, the quota policy was abolished in 2015, allowing increases in output again. In the EU, milk production increased by 5.6% leading up to and after 2015 (EU, 2018).

In Ireland, national production increased by 9.1% in 2017 (CSO, 2019b), which was predominantly influenced by increased milk price. Overall, milk output volume, has increased by 87% on dairy farms since 2014 (NFS, 2017). Between 2014 and 2017, this production expansion was achieved through a combination of increased herd size, and greater productivity per cow (yield per cow) (NFS, 2017). Nationally, two major policies were developed consecutively to achieve the milk production expansion, both of which contained environmental recommendations for air, biodiversity and water quality, to do so sustainably. FoodHarvest 2020 originally set targets of a 50% increase in dairy production output over a decade from 2010, and FoodWise2025 set out additional targets to increase the value of primary production over a decade from 2015 (Food Harvest 2020, 2010; FoodWise 2025, 2015). A report, evaluating the potential impact of the FoodHarvest 2010 production expansion plans was carried out (Farrelly et al., 2014), and included recommendations such as the following: “improved on farm nutrient management planning and knowledge transfer” (water quality), “adoption of best practices for increased livestock numbers” (air quality) and “adoption of best technology for animal breeding and outputs” (GHG’s). The 50% increase in dairy production was met well in advance of 2020 according to the technical analysis of the dairy expansion post-quota (NFS, 2017).

2.2 Policy surrounding the reduction of nutrient losses to the environment

2.2.1 Pedoclimate of Ireland in relation to nutrient loss risk

Nutrient leakages from dairy farm systems, are not consistent or constant, as they are subject to many different environmental factors which alter the quantity and rate, at which they are potentially lost. Therefore, environmental policy and efforts to minimise leakages, aim to be specific to the characteristics of the region or country they are implemented in. In 2018, a scale of 1:250,000 national soil map of Ireland was developed, on the recommendations of the European Soil Bureau Network Technical Working Group (ESBN) as described by Creamer & O’Sullivan, (2018). The final digital version of the soil map, contains information on over 450 soil series, including spatial and quantitative data on soil type and properties (Creamer et al., 2015). Consequently, it highlights the extent of soil spatial variability, across the whole country, which is a fundamental aspect to addressing nutrient loss. At the EU scale, Ireland falls within the Atlantic Central environmental zone, which is deemed the main agricultural zone (or sometimes the “dairy belt”) of the EU (EU, 2011). Where Ireland is situated in this environmental zone, weather conditions like temperature, is influenced heavily by the Atlantic Ocean, and also, height above sea level. Mean annual temperatures generally range between 9°C and 10°C (Met Éireann, 2019).

Higher temperatures may occur inland during the summer, with mean seasonal max ranging between 18°C and 20°C, while winter temperatures don't typically go below 0°C (Met Éireann, 2019). High ground on the western half of the country receives the highest rainfall, and generally decreases towards the northeast. The average annual rainfall is approximately 1,230 millimetres (mm) across the entire country. Seasonally, spring and summer receive an average of 260 mm, and Autumn and Winter receive an average of approximately 350 mm (Met Éireann, 2019). Overall, the high rainfall rates in Ireland, is one of the main drivers of both leaching (downward movement of rainfall/soil water through soil profile), and gleying (due to restricted water movement in the soil profile) (Creamer and O'Sullivan, 2018) i.e. nutrient flow pathways. As a result, poorly drained soils are the main risk of surface runoff (includes overland flow, run-off and shallow later flow), and well drained soils are the main risk of leaching (EU, 2011). Within the Atlantic Central environmental zone, Ireland was classified as "medium" risk potential for surface runoff and classified as "high" risk potential for leaching (EU, 2011). Although this represents country scale, it is at least indicative of the efforts required to manage nutrient loss risk within the Irish pedo-climate characteristics.

2.2.2 Environmental monitoring and policy in the EU and Ireland

There is a major challenge to sustaining food production relative to demand, and additionally, reducing environmental risk associated with that food production. The EU Common Agricultural Policy (CAP), aims to support farmers, and improve agricultural productivity, but also help tackle climate change and the sustainable management of natural resources (EC, 2016). There are a number of agricultural policies that aim to reduce losses from farms, in particular to waterways, but also to the air and GHG's to the atmosphere. These policies typically affect the total nutrient allowance at the farm scale and a limited number of within farm components.

2.2.3 Water quality policy and monitoring

Ireland is bound to a number of EU environmental policies related to agriculture, for example, the water framework directive (WFD). The EU Water Directive 2000/60/EC (EU, 2000) is a policy framework aimed at achieving good quality water across all of Europe. Individual member states are given the capacity, to match the implementation of the requirements of the directive, to the characteristics of their country. The EU Nitrates Directive (ND) (EEC, 1991) which falls under the umbrella of the WFD, is more specifically targeted at improving or maintaining water quality in relation to agricultural management of land and farm facilities. The ND aims are to protect water quality from pollution from agricultural sources, and to promote good farming practice. Under the ND, each member state, is required to prepare a national Nitrates Action Plan (NAP), which outlines rules for nutrient management, and application of organic manures and inorganic fertilisers. Any

NAP, must set a limit on the amount of livestock manure applied to the land each year, set periods when land spreading is prohibited due to risk of losses, and set capacity levels for the storage of livestock manure. Irelands first NAP was implemented in 2006 (S.I. No. 31 of 2014, EU, 2014). Member states are required to monitor compliance with the WFD and ND, and where member states have any rules, they will be subject to financial penalties. At farm scale, non-compliance with the rules set out in the NAP will result in financial penalties. Member States are required to review NAPs at least every four years. Ireland's fourth NAP came into operation in 2017 and will be reviewed in 2021. The NAP restricts the use of N and P fertilisers at farm scale, relative to cropping rates and soil test results, and defines maximum allowable limits for N and P, in an effort to reduce nutrient loss risk to water. In brief, the other main measures for Irish farm owners include the following: a maximum limit on the organic nitrogen loading per ha of 170 kg N ha^{-1} as standard, coming from grazing livestock and, or the application of livestock manure and slurry, however, a successful application for a nitrate's derogation, enables a farmer to increase this limit, of up to a maximum of 250 kg N ha^{-1} while implementing further control measures than the standard ones; a closed spreading period during the winter months when inorganic fertiliser, organic manure, and slurry application is prohibited (the period of time differs according to the location of the farm in the country but there are three main zones, A) 16 weeks (15th September – 12th January) – south and south east of Ireland, B) 18 weeks (15th September-15th January) – midlands and west of Ireland), C) 20 to 22 weeks (15th September- 31st January); and a farmer must have sufficient leak-proof, and structurally sound storage capacity for organic manures to satisfy these closed spreading periods.

In relation to environmental monitoring in Ireland, the latest Environmental Protection Agency (EPA) water quality indicators report (EPA, 2018b) reported that 53% of surface water bodies assessed, are in satisfactory ecological health or the equivalent to being "Good" or "High" ecological status. The WFD aims to achieve "Good status" for all waters (surface, ground and coastal waters) by December 2015 – now December 2021). However, the remaining surface water bodies (47%) are in moderate, poor, or bad ecological status, and the number of high-status waterbodies has declined also. It was also shown, that coastal waters have the highest proportion of water bodies in good or high ecological status (80%), lakes (51%) and estuaries (38%) while 92% of groundwater bodies were found to be in "Good" status (EPA, 2018b). In terms of change in water quality since the last full assessment in 2010-2015, there was an overall net decline in 117 surface water bodies or a 4.4% decrease. According to the EPA, the main issue associated with the impact on water quality was nutrient pollution (nitrogen and phosphorus). Agriculture and wastewater treatment were the main sources of pressure associated with nutrient pollution (EPA, 2018b). As grazed grass livestock systems make up 60% of the farms in Ireland and dairying contributes to 18% of this (NFS, 2017), the aforementioned milk production expansion, may have implications for further environmental losses, and so emphasises the importance of future nutrient management strategies.

2.2.4 Air quality and GHG policy and monitoring

For air quality, Directive 2008/50/EC (EU, 2008) establishes targets for improving human health and environmental quality up to 2020. It also specifies ways of assessing these and sets specific standards. Ammonia emissions, which affect air quality, must be reduced to below the National Emission Ceiling target of 160,000 tonnes. The agricultural sector accounts for practically all ammonia emissions in Ireland (EPA, 2018a). Ireland's ammonia emissions have gradually increased since 2011, and in 2016 they were 0.6% above this target (CSO, 2019a). The major source of ammonia on grazed grass dairy systems, comes from the ammonia volatilisation and soil microbial activity processes, that occur after land application of organic manures, urea N fertilisers and from dung and urine patches (Lockyer et al., 1989; Fischer et al., 2016, Maire et al., 2020, Hyde et al., 2006; Klein and Van Logtestijn, 1994). For nitrogen oxide emissions, there has been a decrease from an average of 175,800 tonnes in 1990-1994, to 107,300 tonnes in 2016, but this is still 65% above the National Emissions Ceiling for 2010 (65,000 tonnes), and 2015 (104,400 tonnes) (CSO, 2019a). GHG emission reduction targets, as decided under the Paris Agreement and amending Regulation (EU) No 525/2013, are binding to each member state and governed by Regulation (EU) 2018/842. Methane (CH4) emissions, a bi-product from microbial fermentation in the rumen of ruminant animals (Johnson et al., 1993), are substantial in Ireland also, due to the scale of livestock systems. In 2017, emissions of CH4 were 14.2 million tonne carbon dioxide equivalents (CO^2eq), indicating a decrease of 4.9 per cent on the 1990 level of 14.1 million tonne CO^2eq (Duffy et al., 2019).

According to the EPA (2019), total GHG emissions from agriculture are projected to increase by 1.4% over the period 2018 – 2020 to 20.5 Mt CO^2eq, which is based on animal numbers, crop areas and nitrogen fertiliser application to soils. Dairy cow numbers are projected to increase by 7% between 2018 and 2020 and 11% between 2020 and 2030 and also, fertiliser nitrogen use is projected to increase by 5% between 2018 and 2020 and 6% between 2020 and 2030 (EPA, 2019). Total N_2O emissions in Ireland are mainly due to N fertiliser use and emissions from animal organic manures. Fluxes in N_2O from agricultural soils are one of the most uncertain emission sources and difficult to manage due to substantial spatial and temporal variability (Krol et al., 2016). Carbon dioxide emissions from agriculture are typically much smaller compared to N_20, <5% (Lanigan et al., 2018).

Despite comprehensive environmental policy, the impact of nutrient losses, according to current monitoring of environmental indicators (water, air and GHG levels), is showing deterioration in the short and potentially the long-term trends also. Nutrient cycling and the losses associated with these losses are complex, and monitoring and managing each and every aspect of the nutrient source and especially the loss pathway is not possible. However, focussing attention on manageable aspects of nutrient cycling on agricultural soils and landscapes, before an impact can occur, may help to improve nutrient management and reduce nutrient losses to the wider environment (Lanyon and Beegle, 1989; Buysse et al., 2005; Monaghan et al., 2008).

2.3 Nutrient cycling and losses in grazed grass dairy systems

2.3.1 Nutrient cycling, transformations, and flows

Nitrogen, phosphorus, and potassium are the nutrients that are typically imported in the highest quantities to meet the production requirements of grass-based dairy farms (Dillon et al., 2018). Sulphur (S) is another essential nutrient and is needed to maintain optimum yield, protein content and high nitrogen use efficiency (Brown et al., 2000). Literature typically focuses on N, or P but S import is important especially in Europe due to the absence of background atmospheric deposition of S in the last 40 years (Mathot et al., 2009) and increased export from high yields from grassland systems (Stevens and Watson, 1986). Once applied to the soil, nutrient flows, transformations and cycles of N, P and K, are impacted differently by 1) pedoclimatic and landscape factors 2) seasonal (temporal) patterns and 3) spatial scale (McCormick et al., 2009; Diacono et al., 2013; Miao et al., 2011; Webb, 1989; Webb et al., 2000; Gillingham, 1973; Van Leeuwen et al., 2019). The consideration of these factors, when implementing nutrient management strategies on dairy farms, could potentially minimise the risk of actual losses (Goulding et al., 2008; Gourley et al., 2007; Mounsey et al., 1998).

Manageable or controllable aspects of nutrient cycles may include, for example, inorganic fertiliser imports, supplementary or concentrate feedstuff use, and organic manure distribution (Mounsey et al., 1998). One of the major challenges in improving nutrient management on farms, is to make management decisions, or advice i.e. "controllable" factors, match the "non-controllable" factors, such as soil type and weather which can exacerbate nutrient loss, spatially and temporally (Grizzetti et al., 2007; Gourley et al., 2007).

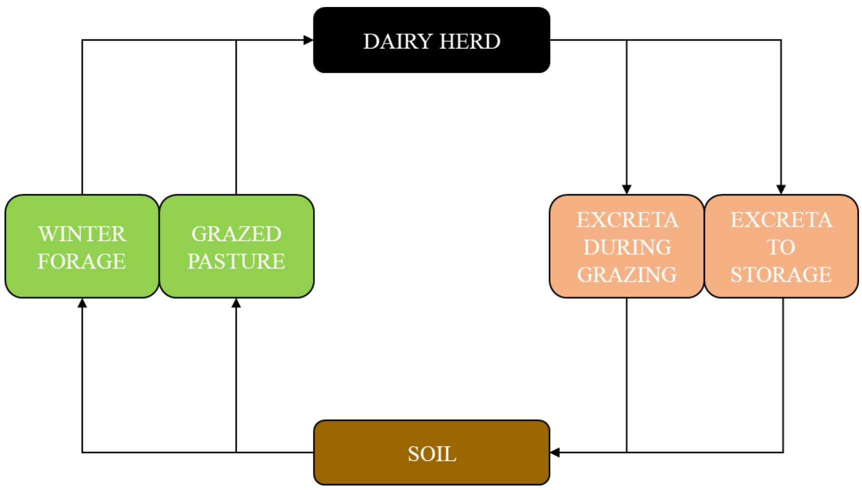

Figure 2-1. Animal (e.g. dairy herd)–Manure (deposited or collected)–Soil–Plant (grazed or harvested) nutrient cycle, without spatial boundaries delineated. Adapted from Oenema et al. (2015).

A simplified Animal–Manure–Soil–Plant nutrient cycle for a dairy system (Figure 2-1), shows the four main components or "nutrient stocks" of a cycle – dairy herd, excreta, soil and pasture/forage. **Dairy herd**: nutrients (e.g. N, P and K) are consumed by the dairy herd through **grazed pasture**, supplement feedstuff and **winter forage** (Oenema et al., 2003). **Excreta**: In grazed grass systems, the animals spend, on average 236 days (depending on the region) (NFS, 2017) outside grazing and so a major proportion of excreta are returned to the soil during this period. Otherwise, excreta are collected in storage tanks during the winter housing period and re-distributed to the fields and paddocks later in the year. **Soil**: In the soil, a variety of microbes make nutrient additions via excreta, inorganic or organic fertilisers become available for plant uptake (Adesemoye and Kloepper, 2009). The cycle starts again when the plant is consumed as pasture or forage. Nutrient losses vary depending on the nutrient in question, and the associated nutrient surplus in the cycle. A more detailed nutrient cycle for N, P and K is shown in Figure 2-2 which shows these differences in cycling and susceptibility to different losses.

The following section describes in more detail the major pools and transformation pathways involved in the cycling and losses of N, P and K in grass-based dairy systems. The paragraphs below are numbered and correspond to the numbers in Figure 2-2 (e.g. paragraph 2.3.3.1 [2.a] Fixation (N_2 to NH_3), is connected to, 2.a in the N cycle illustration (Figure 2-2)).

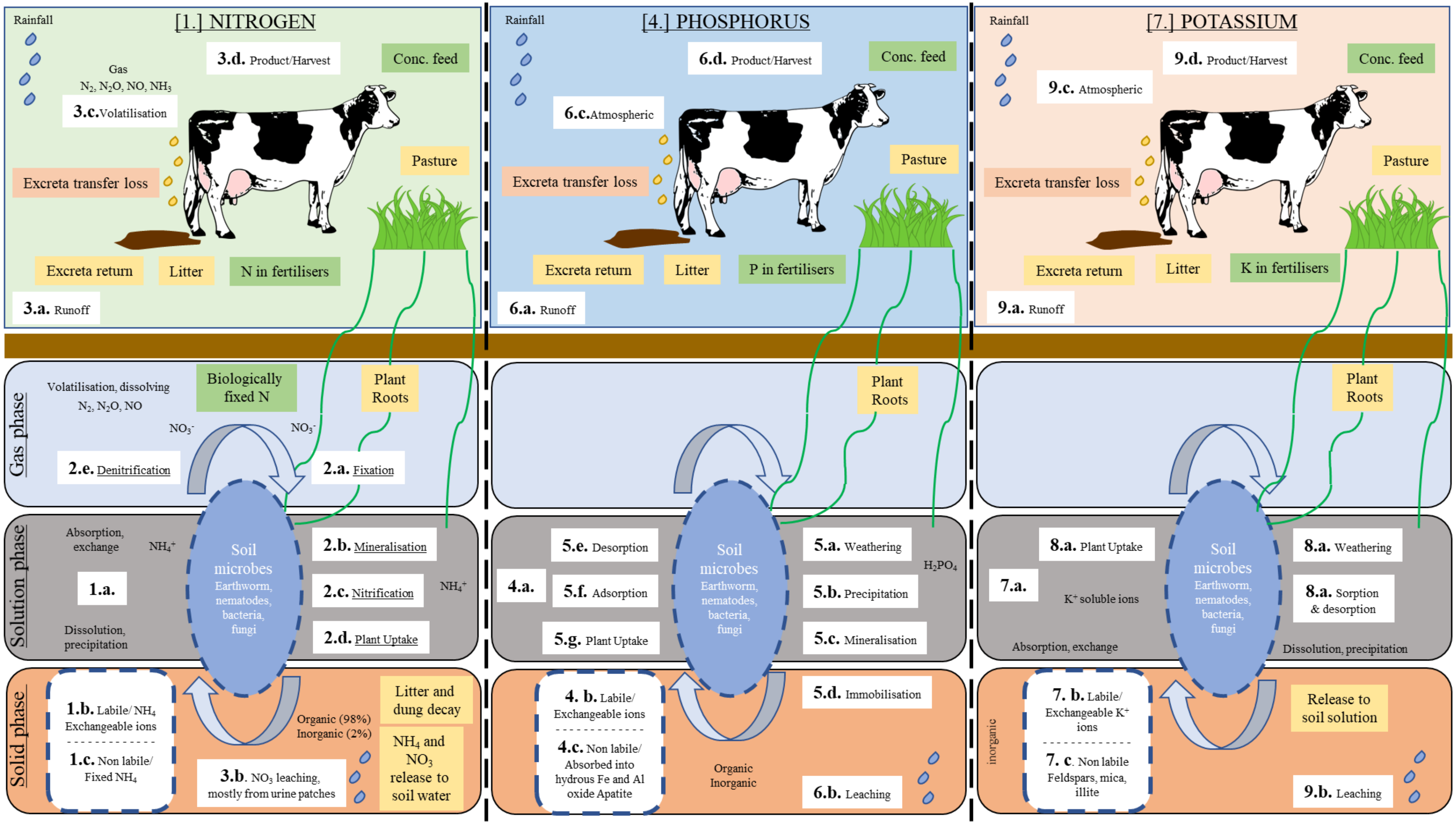

Figure 2-2. Diagrammatic version of nutrient cycles for N, P and K. Reproduced and adapted from, FLRC, 2016, and also, OECD, 2007, Oenema et al., 2015, Gourley et al., 2007.

2.3.2 [1.] Nitrogen forms in the soil of grazed grass dairy systems

N in the atmosphere, or in the soil, can go through many complicated chemical and biological changes, be combined into living and non-living material, and return back to the soil or air in a continuing cycle.

2.3.2.1 [1.a] Soil solution (NH_3 and NH_4)

To become biologically available, N_2 must be converted to reactive N through various transformations and processes. Inorganic forms in the soil include NH_3 and $NH_{4,}$ inorganic oxidised forms are NO, NO_2, HNO_3, N_2O, NO_3 and organic forms include urea, amines and proteins. In the soil solution phase of the soil (Figure 2-2), N is plant available as NO_3 and NH_4 ions but this only makes up approximately 2-3% of the soil solution (Brady and Weil, 2008).

2.3.2.2 [1.b]. Exchangeable (labile NH_4)

Ammonium ions are positively charged, and therefore, sorbed to negatively charged clay particles, and soil organic matter. The positive charge limits ammonium N from being washed out of the soil (or leached) by rainfall. However, in warm, moist and aerobic soils with neutral pH, very little exchangeable NH_4 accumulates, because nitrification (biological conversion of NH_4 to NO_3) continually removes NH_4 from the soil solution (Figure 2-2) (Haynes, 2012). In contrast, the negatively charged nitrate ion is not held by soil particles and is very soluble in soil solution, and is easily lost from the soil system by leaching (OECD, 2007). Eurostat, (2013) As a result, the solution phase is quickly depleted by plant uptake and leaching. NO_3 concentrations, in warm and moist cultivated soils are normally higher than NH_4 concentrations. In these soils, the main source of N taken up by plants is NO_3. This is because the rate of nitrification (oxidation of NH_4 to NO_3) rapid in warm or well drained soils (Haynes, 2012). In cold wet soils or poorly drained soil, nitrification is typically slower. Wetness restricts the air-filled pore space in the soil and slows oxygen diffusion rates, and nitrification becomes slower (Stepniewski et al., 1994).

2.3.2.3 [1.c.] Non exchangeable (non-labile NH_4)

In higher organic matter top soil layers, fixed NH_4 normally accounts for less than 5% of the total N (Scherer, 1993). In subsoils, where organic matter is lower, fixed NH_4 may make up to 50% of the N present (Steffens and Sparks, 1997).

2.3.3 [2.] Nitrogen transformations/processes

2.3.3.1 [2.a.] Fixation (N_2 to NH_3)

Virtually all N present in soils is derived from either biological or chemical fixation of N_2 from the atmosphere (78% N_2) (Fisher and Newton, 2002). N_2 is an inert compound and biologically unavailable to most organisms. Of the total soil N, almost 98% of it is present in water-insoluble/organic form (Cameron et al., 2013). For example, ammonium ions can be held unavailable in the interlayer spaces of illitic clay minerals (such as vermiculite and chlorite).

However, rhizobium bacteria on the root nodules of legume plants (e.g. white clover (*Trifolium repens*) can fix up to 150 kg N ha $year^{-1}$ on Irish soils (Humphreys and Lawless, 2006). Fixation can vary depending on soil fertility and plant species (Peoples and Baldock, 2001). On high abundance clover-based pasture which has not been disturbed by cultivation, the supply of soil N can range from 0-100 kg N ha year (Curtin et al., 2019). It has been shown that as organic N levels in the soil increase, the rate of N biological N fixation decreases (the population of legumes decreases in natural succession to non-legumes) until a steady state equilibrium is reached (Curtin et al., 2019).

Synthetic fertiliser N is produced via a non-biological fixation process; the Haber Bosch process (Smil, 2004). This chemically fixed N (N_2 to NH_3) is then further processed to make inorganic fertiliser N for soil application. According to Fowler et al. (2013), in 2010, the fixation of atmospheric N through Haber Bosch (120 Tg N yr−1) was double that of the natural terrestrial sources of N (63 Tg N yr−1).

2.3.3.2 [2.b.] Mineralisation/ammonification (NH_3 to NH_4)

The natural decomposition of protein in plant litter and animal excreta (Figure 2-2) by soil bacteria and fungi adds N to the soil via mineralisation/ammonification processes. A small proportion of N (NH_3 and $NO^-{}_3$) is added to the soil via dry deposition and rainfall also (2-22 kg N ha^{-1}, Doyle et al., 2014).

2.3.3.3 [2.c]. Nitrification (NH_4 to NO_2 and NO_2 to NO_3)

NH_4 can be transformed into nitrite (NO_2) or nitrate (NO_3^-) via nitrification by soil microbes. Nitrification requires the presence of oxygen, so this process can only happen in oxygen rich environments like circulating or flowing waters and the very surface layers of soils and sediment. NO_2 tends to bind to soil particles because they are oppositely charged, while NO_3^- tends not to bind to soil as they are the same charge and so losses occur in this form of N (Schulten and Schnitzer, 1997).

2.3.3.4 [2.d.] Plant N uptake ($NH_4^+$and NO_3^-)

Ammonium (NH_4^+) and Nitrate (NO_3^-) ions are the plant available forms of N in the soil that non-leguminous plant roots can absorb. In aerobic soils, where nitrification is favoured, the NO_3^- supply to plant roots is mostly satisfied by the transpiration flow of soil water solution (Haynes, 2012). In wet, cold soils or acidic soils, where nitrification is not favoured, ammonification still occurs but NH_4^+ is the dominant form of N in the soil solution. Supply to the plant root is mostly by diffusion of NH_4^+ from exchange sites or organic matter decomposition.

In the rhizosphere, large quantities of N are required by plants and the form of N taken up by plants is an important factor for determining the uptake of other ions by plant roots (Brady and Weil, 2008). This is because there must be charge balance across the root soil interface when plants take up nutrients. If taking up an excess of positive charge (i.e. all plant N taken up as $NH4^+$) a balancing cation must be detached from the roots. This cation is normally H^+, which acidifies the soil outside the root zone (Fisher and Newton, 2002). If taking up an excess of negative charge (i.e. all plant N taken up as NO_3^+) a balancing anion must be detached from the roots. This anion is normally OH which makes the soil outside the root alkaline (Fisher and Newton, 2002). This is an important factor that contributes to soil pH changes and the requirement levels of liming on grazed grass dairy systems.

2.3.3.5 [2.e.] Denitrification (NO_3^- and NO_2^- to N_2 and N_2O)

Through denitrification, oxidised forms of N such as nitrite (NO_2^-) and nitrate (NO_3^-) are converted to dinitrogen (N_2) and to a lesser extent, nitrous oxide (N_2O). Denitrification is an anaerobic process that is carried out by denitrifying bacteria.

2.3.4 [3.] Nitrogen losses

2.3.4.1 [3.a.] Erosion/runoff

Nitrogen loss via erosion and runoff, are low on pasture soils depending on the amount of cultivation, for example, in comparison to tillage soils (Douglas et al., 1998; Schilling and Spooner, 2006). However, the loss of N to surface water, with sufficient phosphorus (P), can still lead to eutrophication and increased nitrate levels (Chambers et al., 2001; Eickhout et al., 2006).

2.3.4.2 [3.b.] Leaching

As nitrate is not adsorbed by positively charged soil surfaces losses of nitrate occur in the drainage of the soil solution during and after rainfall, or via artificial field drainage (Hanly et al., 2017). Factors that determine the extent of nitrate leaching are the use of N fertiliser, the accumulation of nitrate in the soil, temperature and frequency of rainfall (Jarvis, 2000). Accumulated soil nitrate is derived from the mineralisation of soil organic N, from additions of nitrate fertiliser, hydrolysis and nitrification of urea, and ammonia fertiliser and urine patches in grazed pastures (Cameron et al., 2013). When animals deposit nutrients to the soil via excreta (urine and manure “patches”), it results in an uncontrolled mosaic of low to very high nutrient status zones within the pasture (Maire et al., 2020). The percentage of N in urine increases, as dietary N increases while remaining fairly constant in faeces (Marini and Van Amburgh, 2005). The high concentration of mineral N found under urine patches, is more N than plants within the spot can utilise, and may lead to higher leaching losses than would otherwise occur (Silva et al., 2000; Stout et al., 1997), this is particularly the case for grazed grass livestock systems.

2.3.4.3 [3.c.] Atmospheric

According to Sommer and Jensen, (1994), agriculture accounts for up to 50% of the volatilised ammonia on the planet. It is estimated that the potential risk of ammonia volatilisation from urea fertiliser can range from 0% to 65% of the N applied, but depends on soil and climatic conditions (Martens and Bremner, 1989; Bussink and Oenema, 1998), highlighting the importance of inorganic N application strategies on grazed grass systems. Volatilisation occurs mostly when there is a combination of soil pH above 7 and free NH_4 ions (Haynes, 2012; Peoples and Baldock, 2001). This commonly occurs in urine, manure and after urea application. High cation exchange capacity (CEC) and low soil pH will reduce N loss via volatilisation; as will high soil moisture contents, which will allow rapid dispersion of the high concentrations of urea added in urine spots and urea granules (Brady and Weil, 2008). Grazed swards may lose up to 1kg N ha^{-1} day^{-1} (Stout et al., 2000) from urine spots when high animal stocking rates are present.

Denitrification is the only N transformation that removes N from the ecosystems and it roughly balances the amount of N fixed by the N biological fixation process (Fisher and Newton, 2002). The rate of denitrification varies with soil conditions. In heavy clay soils it can be up to 0.6 kg N day, as has been observed in autumn and spring in temperate climate regions (Mosier et al., 2002; Sexstone et al., 1985).

2.3.4.4 [3.d.] Grazing/Harvesting/Products

When considered from the field or paddock perspective, significant nutrient quantities are removed in pasture consumption and forage harvests. A dairy cow can consume up to 0.79 kg N per day (Latinga et al., 1987) on 1kg concentrate feed and 16 kg DM forage per day. Another way of looking at it, firstly, a sufficient nitrogen concentration for mature leaves in a pasture is considered to be 30 g kg^{-1} (Wall and Plunkett, 2016). Then, using the mean grazed grass utilisation value from earlier (9.3 tonne DM ha^{-1}) it can be estimated that 279 kg N ha^{-1} yr^{-1} could be removed in grazed pasture. The top performing (in terms of total herbage accumulation) dairy farms in Ireland produced an average of 18 tonne DM ha^{-1} (or around 13.5 t DM ha^{-1} utilised), in 2017 of pasture and forage (O'Donovan and Egan, 2019), This suggests a potential for approximately 405 kg N ha^{-1} to be removed from the soil in a year depending on growing conditions and management. To take this estimation one step further, with a milk nitrogen use efficiency value of a lactating cow at 22% (Foskolos and Moorby, 2018), although it can range from 23 to 31% (Deenen, 1994), it suggests that approximately 316 kg N ha^{-1} is partitioned into animal product and excreta which is then returned to the soil, predominantly as urine (Deenen, 1994), each year.

2.3.5 [4.] Phosphorus forms in the soil of grazed grass dairy systems

2.3.5.1 [4.a.] Soil solution

There is typically a very small pool of solution P. For example, in unfertilised soils from northern Australia (rain fed grass region), the concentration is typically below 2 µM, while even in highly fertilised soils, the P concentration is typically below 20 µM (Menzies et al., 1994). P dissolved in soil solution is usually in orthophosphate form, but small amounts of organic P may exist as well. Plants will only take up P in the orthophosphate form (Brady and Weil, 2008). As the pool is small, a growing crop would quickly deplete the P in the soluble P pool, and so the pool has to be continuously replenished.

2.3.5.2 [4.b.] Active/exchangeable/labile

The active/exchangeable pool is the main source of available P for crops (OECD, 2007) (Figure 2-2). It contains P in solid form which is relatively easily released to the soil solution As plants take up phosphate from the solution P pool, the concentration of phosphate solution is decreased and some phosphate from the active P pool is released (Vaclav, 2000). The ability of the active P pool, to replenish the soil solution P pool in a soil, is where soil fertility is important with respect to phosphate fertiliser import rates and timing on grazed grass dairy systems. The active P pool contains inorganic phosphate that is adsorbed to small particles in the soil. It also contains phosphate that reacted with elements such has calcium and aluminium to form semi soluble solids, and organic P, which is easily mineralised (Brady and Weil, 2008). Adsorbed phosphate ions are held on active sites on the surfaces of soil particles. The amount of phosphate adsorbed by soil increases when the amount of phosphate in solution increases and vice versa (Brady and Weil, 2008). Soil particles can act either as a source or a sink of phosphate to the surrounding environment depending on conditions.

2.3.5.3 [4.c.] Fixed/non exchangeable/non-labile

The fixed pool contains inorganic phosphate compounds that are very insoluble, and organic compounds that are resistant to mineralisation by microorganisms in the soil. Phosphate in this pool may remain in soils for years without being made available to plants and may have very little impact on the fertility of the soil (Horrocks and Vallentine, 1999). Some slow conversion between the fixed P pool and the active P pool does occur in some soils (Brady and Weil, 2008).

2.3.6 [5.] Phosphorous transformations/processes

2.3.6.1 [5.a.] Weathering

Soils naturally contain P rich minerals, which are weathered over long periods of time and slowly made available to plants (Vaclav, 2000). The reaction increases the availability of P to plants.

2.3.6.2 [5.b.] Precipitation

Phosphate precipitation is a very slow process in which P reacts with another substance to form a solid mineral (Brady and Weil, 2008). As phosphate is removed from the soil solution this reaction decreases the availability of P to plants. Soil pH levels indicate how certain elements (iron, aluminium and calcium) interact with P in the soil, and it is this interaction that affects P availability (Brady and Weil, 2008). When the soil pH is less than 5, the iron and aluminium concentrations are very high and react quickly with P (Brady and Weil, 2008). This creates iron and aluminium phosphate minerals. When the soil pH is more than 7, calcium concentrations are very high and P fixation with calcium occurs (Odutola Oshunsanya, 2019). P fixation occurs to some degree for all soils, even for pH levels from 6 to 7 (Brady and Weil, 2008). This pH range is where P availability is typically at its highest in mineral soils and fixation due to iron aluminium and calcium is at its lowest (Oshunsanya, 2019).

2.3.6.3 [5.c.] Mineralisation

The release of inorganic P from organic by mineralisation is caused by microorganisms breaking down complex compounds (Brady and Weil, 2008). The activity of microorganisms is influenced greatly by soil temperature and soil moisture (Spohn and Kuzyakov, 2013). The process is most rapid when soils are warm and moist but well drained. The reaction increases the availability of P to plants.

2.3.6.4 [5.d.] Immobilisation

The reverse process of mineralisation, immobilisation refers to the tie up of plant available P by soil minerals and microbes that use phosphorus for their own nutritional need (Bunemann et al., 2010). This reaction reduces the availability of P to plants. The availability of P is primarily dependent upon the pH of the soil (Brady and Weil, 2008). Microbes may compete with plants for P, if the decomposing organic materials are high in carbon and low in N and P (Adesemoye and Kloepper, 2009). Mineralisation and immobilisation occur simultaneously in soil. If the P content of the organic material is high enough to fulfil the requirements of the microbial population, then mineralisation will be the dominant process (Adesemoye and Kloepper, 2009).

2.3.6.5 [5.e.] Adsorption

Adsorption is the chemical binding of plant available P to soil particles, which makes it unavailable to plants, this is a quicker process compared to desorption (Bunemann et al., 2010). Adsorption also differs from precipitation: adsorption is reversible chemical binding of P to soil particles while precipitation involves a more permanent change in the chemical properties of the P as it is removed from the soil solution (Eurostat, 2013). Since phosphate is an anion, particles that generate an anion exchange capacity (aluminium and iron oxides) will form strong bonds with phosphate. Additionally, in calcareous soils adsorption may occur as phosphates sorb to impurities, such as aluminium and iron hydroxides, or displace carbonates in calcium carbonate materials (Jones and Oburger, 2001).

The soil mineral types therefore affect P adsorption, e.g. volcanic soils and highly weathered soils have high P adsorption capacities (Fay, 2007). Also, as the amount of clay increases in the soil, the P sorption capacity increases as well (Brady and Weil, 2008). This is because clay particles have a high proportion of surface area for which phosphate sorption can take place. At low pH, soils have greater amounts of aluminium in the soil solution, which forms very strong bonds with phosphate. Generally, P sorption increases as temperature increases (Brady and Weil, 2008). Organic matter can reduce the adsorption capacity and therefore increase the P availability for plants also (Brady and Weil, 2008). For example, organic anions can displace sorbed phosphate, as it can become coated with aluminium and iron oxides, which reduces P sorption (Brady and Weil, 2008). Waterlogged or poorly drained soil, reduces P sorption, by increasing the solubility of phosphates that are bound to aluminium and iron oxides and amorphous materials (Daly et al., 2015).

2.3.6.6 [5.f.] Desorption

Desorption is the release of adsorbed P from its bound state into the soil solution (Eurostat, 2013). Other anions such as silicates, carbonates, sulphates, arsenate, molybdate compete with phosphate for a position on the anion exchange site (Brady and Weil, 2008). As a result, these anions can cause displacement, or desorption, of phosphate from the soil exchange sites. Desorption causes phosphate availability in the soil solution to increase (Bunemann et al., 2010). Phosphate sorption and desorption control the soil solution P concentration in most soils. This is an important factor in determining how much P fertiliser will be available to the plant, and how much P fertiliser is required to raise soil test P to target levels of grazed pastures. The soils capacity to retain P, can be determined by assessing the anion retention capacity of the soil, which is an indication of the ability of soils to adsorb P (Doran and Parkin, 1994). The equilibrium condition between the amount of P adsorbed on a soils surface, and the P in the soil solution, determines the ability of a soil to immediately replenish solution P, which has been taken up by the roots (Brady and Weil, 2008).

2.3.6.7 [5.g.] Uptake

Phosphate must be in the soil solution before it can be adsorbed by plant roots, however, the concentration in solution is normally very low and inadequate to sustain plant growth (Eurostat, 2013). In pasture soils, growth of roots and micro-organisms convert inorganic (Pi) to organic P (Po) which is immobilises plant available P (Fay, 2007). The amounts of Po in soils vary enormously from soil to soil (Fay, 2007). The amount of Po in a soil results directly from the difference in organic P addition and organic P decomposition. Processes that lead to increases in soil organic matter also lead to increases in soil Po (immobilisation) (i.e. high plant growth rate and slow litter or dung decay; high anion retention may slow organic matter decay). Phosphate fertiliser is converted to Po in productive environments (e.g. grassland where Po input as roots shoots and animal excreta decay is high) (Fay, 2007).

As a plant root takes up orthophosphate ions, the already low solution of P concentration, drops. Replacement of P must be to replenish the concentration at the root surface or the root must grow into an area in the soil with a higher P concentration (Vance, 2001). Movement of P can take place either by the soil solution moving (mass flow), during periods of net water uptake by the plant roots (as transpiration occurs during daylight hours), or by P ions diffusing (diffusion) from a high concentration near the soil surface to the low concentration on the root surface (Brady and Weil, 2008). Travel time for P to reach roots in high sorbing (relatively less P ions in solution – more adsorbed to soil colloids) soils, will be longer than in low P sorbing soil (relatively more P ions in soil solution – less adsorbed on soil colloids). Therefore, fertiliser P must be supplied to one soil type more than the other, to increase the soil solution P concentration and allow P to move faster to the plant root.

Liming is one of the important practices that has often been shown to influence P mobilisation in soils. Liming has been shown to reduce the amount of P fertiliser required to achieve a given yield on grazed pastures (Ryant et al., 2016 and Wall et al., 2018). The reduction of P requirement results directly from an increased solubilisation of soil P and its subsequent uptake, and or, indirectly from an increase in P uptake due to reduced Aluminium and Manganese toxicity (Spohn and Kuzyakov, 2013). Furthermore, an additional lime-sparing effect of P is owed to calcium in lime stabilising cell membranes in the absence of sufficient P and preventing the leakage of water soluble carbohydrates and nutrients as shown by Bailey, (1991).

2.3.7 [6.] Phosphorous losses

2.3.7.1 [6.a] Erosion/runoff

Surface runoff and erosion of topsoil, by wind and water, removes organic and inorganic P dissolved in the runoff water and on fine soil particles (particulate P) (Daniel et al., 1998). Eroded soil represents the major transfer mechanism of P from terrestrial to aquatic systems (McConnell et al., 2016). The proportions of dissolved and particulate P, may vary depending on landscape, soil type, their management and climate (Correll, 1998). Poor land management can accelerate this transfer (Cherry et al., 2008; Ekholm et al., 2005; McDowell et al., 2003).

2.3.7.2 [6.b.] Leaching

Due to P being strongly adsorbed by soils, little P leaching occurs, however it is expected to be a greater risk on arable systems (Djodic et al., 2004). For pastoral systems however, pathways from artificially drained fields are a major contributor to losses (Hanly et al., 2017; Houlbrooke et al., 2003). In some cases, P can be lost from highly stocked catchments with well drained soils also (Jordan and Shortle, 2017).

2.3.7.3 [6.c.] Atmospheric

The atmosphere does not play a significant role in the movement of phosphorus, because phosphorus and phosphorus-based compounds are usually solids.

2.3.7.4 [6.d.] Grazing/Harvesting/Products

A sufficient phosphorus concentration in mature leaves of pasture is 3 g kg^{-1} DM. Therefore, a farm growing 9.3 t DM ha^{-1} or 13.5 t DM ha^{-1} could remove approximately 27.9 kg P ha^{-1} and 40.5 kg P ha^{-1}, respectively, for pasture and forage production. According to Arriaga et al. (2009) for dairy cattle in the USA (note the diet would be different to a predominantly grass-based dairy cattle system), the mean P use efficiency for a lactating cow was 32% (standard deviation 4.5), which suggests that up to 24.54 kg P ha^{-1} would be returned to pasture in dung and urine or captured in animal physiology each year.

2.3.8 [7.] Potassium in soils of grazed grass dairy systems

2.3.8.1 [7.a.] Soil solution

Potassium cations (K+) dissolved in the soil solution account for 0.1-0.2% of total soil potassium (Sharpley et al., 1988). Plant roots take up K as the K+ ion from the soil solution. Potassium is taken up by plants in large quantities (Zörb et al., 2014). K+ ions in the soil water which are in dynamic equilibrium with other forms of K.

2.3.8.2 [7.b.] Active/exchangeable

Exchangeable K constitutes 1-2% of soil K, as ions held on to negatively charged cation exchange sites, on soil clay minerals and organic matter surfaces (Wakeel et al., 2013). Exchangeable K must exchange with solution cations before it can be taken up by plant roots. Exchangeable K includes K+ ions adsorbed and released on clay and organic colloids (Wakeel et al., 2013).

2.3.8.3 [7.c.] Fixed/non exchangeable

Fixed K makes up 1-10% of soil K+ ions held in the interlayer spaces of the 2:1 layer clay minerals, which are mica weathering products (e.g. hydrous micas illite – vermiculite) (Wakeel et al., 2013). The clay expands with larger hydrated cations (H+, Ca2+) occupying the interlayer space before fixed K can slowly exchange with solution and exchangeable K and become plant available (Kayser and Isselstein, 2005). Un-weathered mineral K constitutes 90-98% of soil K still present in soil parent material such as feldspars and micas (Zörb et al., 2014). The slow release of K from ongoing weathering of these nutrients will replenish the supplies of exchangeable and solution K, provided that losses of K from the soil solution pool, by plant uptake and leaching, are not faster than the rate of weathering (Sharpley et al., 1988).

2.3.9 [8.] Potassium transformations/processes

2.3.9.1 [8.a.] Weathering

The release of mineral K is limited by the rates of weathering, which in cool, dry climates can be very slow (Zörb et al., 2014). Feldspar weathering will replenish a K-depleted soil, but the time frame can be on the order of decades or even centuries (Zörb et al., 2014). Micas range from relatively resistant muscovite to more easily weathered biotite (Zörb et al., 2014). Depletion of solution K by plants or leaching can induce further mica weathering to maintain an equilibrium with the soil solution (Zörb et al., 2014).

2.3.9.2 [8.b.] Sorption and desorption

Clay fixation immobilizes K in a nonexchangeable form, making it temporarily unavailable for crop uptake (Brady and Weil, 2008). With high K concentrations and dry conditions, the clay layers shrink together and make K unavailable to plant roots (Wakeel et al., 2013).

Potassium fixation and release by clay is closely related to soil pH and CEC (Brady and Weil, 2008).

As the pH increases, H+ ions are removed from cation exchange sites between clay particles. These negatively charged sites are more available for binding K+, so the soil solution K decreases and the amount of nonexchangeable K stored in the soil increases (Zörb et al., 2014). Following the same reasoning, soils with more cation exchange sites (i.e., higher CEC) will attract more K+ than a soil with fewer cation exchange sites, such as sandy soils (Wakeel et al., 2013).

Soils with high K-fixing capacities have been found to show little response to large amounts of K fertiliser because the surplus of available potassium quickly binds to clays (Wakeel et al., 2013). K fixation can be beneficial by reducing leaching losses of K and storing a reservoir of K for future crops (Zörb et al., 2014). A sandy soil and a clay loam soil may have the same initial soil test K values, but their response to crop uptake of K will probably be very different because sands have low CEC's and therefore a lower K supplying power than clay loams (Wakeel et al., 2013).

2.3.9.3 [8.c.] Uptake

The rate at which a crop takes up K is dependent in the concentration of K+ ions in the soil water film that bathes the root (Sharpley et al., 1988). Additionally, the concentration of the root surface is limited by the rate at which K+ ions will diffuse towards the zone of low K+ concentration at the root surface or in the rhizosphere soil (Jones, 1998). The size of the exchangeable K pool is a better measure of K amounts available to plants than the amounts of soil solution K (Zörb et al., 2014). However, measures of exchangeable K can be unreliable because K can also be released from fixed K and other soil mineral K Soils with greater amounts of 2:1 layer silicate clays and organic matter (fine textured soils) have higher CEC and hold more K in the exchangeable form (Zörb et al., 2014). In such soils, as plant growth or leaching depletes soil K, a greater reserve of exchangeable K can be release to maintain the solution K concentration in a range suitable for plant growth (Zörb et al., 2014). Sandy or coarse texture soils with low CEC lack this ability, vice versa in soils with low CEC, more of the added fertiliser K remains in soil solution and easily take up by plants or leached. (Wakeel et al., 2013).

2.3.10 [9.] Potassium losses

2.3.10.1 [9.a.] Erosion/runoff

Erosion by wind and water removes soil from the top of the profile where plant available K is usually highest, due to plant recycling of the nutrient (Brady and Weil, 2008). Therefore, minimizing erosion losses will help maximize the amount of K available for crops

2.3.10.2 [9.b.] Leaching

Annual leaching loss of K from the soils in a temperate region under agricultural production (receiving only a moderate rate of K fertiliser) is usually about 25 to 50 kg K/ha (Kayser and Isselstein, 2005).

Soluble potassium K is prone to leaching from soils. Leaching losses of potassium are generally small except under high rainfall (1500mm) on coarse textured soils derived from sand or pumice, or on peats (Sharpley et al., 1988). In these situations, leaching losses can be high. The amount of K leached, depends on the stocking rate and the amount of K being ingested, because up to 70% of the excreted K appears in urine, resulting in potential accelerated leaching of K from urine patches (Kayser and Isselstein, 2005). K release from soil depends largely on the soil type, some soils can supply K at a sufficient rate that little or no fertiliser additions are needed, other soils supply K at a much slower rate relative to the amounts required for annual plant growth (Sharpley et al., 1988). K ions are attracted to cation exchange sites with less bonding strength than other common soil cations Ca^{2+} and Mg^{2+} and with less strength than anionic H2PO4 (Sharpley et al., 1988). Amounts of K lost by leaching in water are significant. Frequent applications of Ca2+ in fertiliser and lime materials will displace K^+ form CEC sites and accelerate leaching losses (Kayser and Isselstein, 2005). Whereas plants require large amounts of K, animals do not, and therefore excrete the unwanted K in dung and urine. Large deposits of K in cow urine patches can create hotspots of K leaching (Kayser and Isselstein, 2005).

2.3.10.3 [9.c.] Atmospheric

Similar to P, there is no significant role of the atmosphere of K in the cycle because of the nature of the element as a solid.

2.3.10.4 [9.d.] Grazing/Harvesting/Products

Plants take up very large amounts of K. When most or all of aboveground biomass is removed in a harvest, the soil loss of K can be substantial. Annual losses (off-take) of K through harvesting can be as great as 400 kg K/ha and are common if the plant is a legume and is cut several times for hay (Kayser and Isselstein, 2005). From our example, if pasture utilisation was 13.5 t DM ha^{-1}, and herbage K concentration was 45 g kg^{-1}, approximately 608 kg K ha^{-1} would be taken up from the soil before being portioned by the animal and returned.

2.3.11 Summary

As shown, there are many pathways for losses for N, P and K on grazed grass systems. Increased dairy production outputs which are now expected of grass-based systems across Europe, can also be expected increase nutrient source pressure on these pathways. Added to this risk, is that nutrients are also lost if used too early or too late in the growing season.

2.4 Temporal (seasonality) aspect of nutrient cycling

Nutrient transformations and flows do not happen at a uniform rate during grass growing season. Seasonal patterns of temperature and precipitation, are very important in terms of nutrient availability and potential transport during a nutrient cycle. As mentioned, soil temperature directly influences microbial activity (Naher et al., 2019), which is the fundamental cog of the nutrient cycle for N P and K. Therefore, it is very important that the controllable aspects of nutrient management, such as the rate and timing of imports, is fitted to the times in the year when losses and connectivity to loss pathways, are at the lowest risks. Early spring and late autumn, for example, have the lowest evapotranspiration and highest precipitation and recharge, and therefore high percolation rates, which means, leaching and runoff are extremely likely pathways for nutrient loss in these seasons (Cai and Ofterdinger, 2016; Fenton et al., 2011). On the contrary, evapotranspiration is highest and precipitation and recharge is lowest in summer, and therefore percolation/leaching and runoff are much lower in this season (Cai and Ofterdinger, 2016; Fenton et al., 2011). Therefore, the early and the late sides of the growing season (January to March and September to October) are riskiest for nutrient imports loss (similar to NAP fertiliser spreading prohibition periods) [*see section 2.2.3*], compared to the middle of the growing season (April to August). This seasonality is illustrated hypothetically in Figure 2-3. A typical growing season lasts between March and September, and so the nutrient imports, should match growth rates relatively closely. For example, Mihailescu et al. (2014) showed the N imports were highest between March and June, for a sample of commercial Irish dairy farms, with a mean of 40 kg N ha^{-1}, and was predominantly as inorganic fertiliser. It was also shown that P imports were highest between April and June with a mean of 2.83 kg P ha^{-1}, and was also predominantly as inorganic fertiliser (Mihailescu et al., 2015). Overall, this gives a basic framework, of the nutrient management strategy in relation to seasonality, for grass-based dairy systems in Ireland.

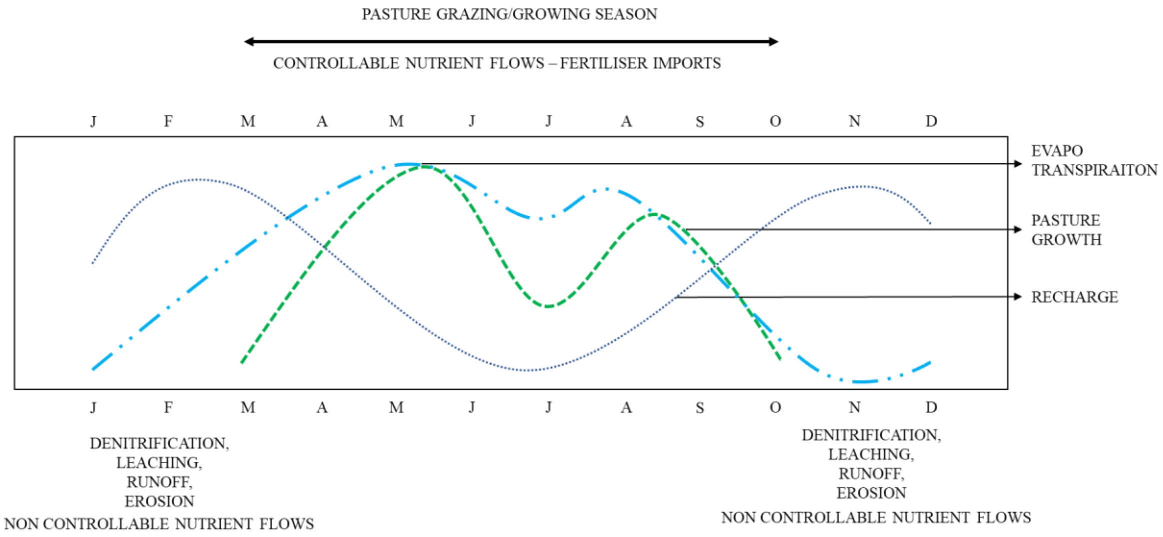

Figure 2-3. Illustration of seasonality showing differences in evapotranspiration, pasture growth and recharge levels and loss risk during different times of the year. Adapted from Magdoff et al., 1997.

2.5 The nutrient transfer continuum conceptual framework

A conceptual framework called the nutrient transfer continuum (NTC) (Figure 2-4) is an effective reference, when describing the connectivity between the various controllable and non-controllable factors in nutrient cycling. In short, the source of nutrient flow (imports), and depending on spatial delineation (scale), may start at farm scale, be distributed at field scale, and be affected by physical and chemical properties in the soil, before being transported sporadically via temporal/seasonality conditions to the environmental receptor that is being monitored by policy makers (for example waterways, air or atmosphere) where an impact is determined. This nutrient "journey" is the nutrient transfer continuum conceptual framework developed by Haygarth et al., (2005a) and previously explored by Lemunyon and Gilbert (1993), and also known as a nutrient cascade (Smith et al., 2013).

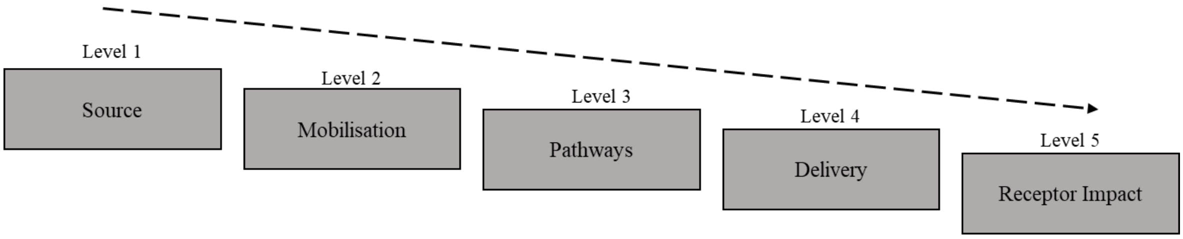

Figure 2-4. Nutrient transfer continuum (NTC) conceptual framework showing each level from source to receptor impact. Adapted from Haygarth et al., (2005a) and Murphy et al., (2015).

This concept has advantages in providing insight into the reality that multidisciplinary understanding is needed to fully appreciate the transport of nutrients from source to receptor (Figure 2-4). Haygarth et al., (2005a) suggested that research is predominantly tied to a single scale through component analysis or "mechanistic" studies, which serve well within a specific context, but inaccuracies may occur when such studies are scaled up or down to represent large or diverse scales. This indicates that there is a need for multi scaled approaches to reduce the risk of inaccuracies. Haygarth et al., (2005) for example, highlighted this for soil P and P export, and concluded that it was not possible to disaggregate between soil P concentration and P export at large scales (catchments), as there was no clear relationship between P import via fertilisers and the resulting P transferred to the receptor scale (waterbody).

Furthermore, Wall et al., (2011), used the nutrient transfer continuum concept to evaluate the effectiveness of the EU NAP. The study outlined the importance of, not only budgeting at the source end of the continuum, but also the inclusion of the linkages via pathways and delivery to the receptor, that a policy may be monitoring and basing regulation on. Additionally, Murphy et al., (2015) indicated that source level reduction of nutrients as implemented by the NAP, may have resulted in the intended reductions of source level pressure. However, the impact of that reduction was not clearly detected at the receptor level, at least within the short term (up to 3 years). As a result, the source level was a better observable unit than the receptor level, in terms of monitoring changes over time and monitoring responses to policy or regulations. Both studies also outline that distribution of nutrients was poor at source level, and that budgeting strategies at this level of the continuum needed to include, mobilisation factors such as soil type and chemical and physical factors, as well as landscape factors, in an effort to increase precision in nutrient distribution. Farm scale and field scale are both situated at the source level of the continuum, and if monitored together, in a multi scaled approach, it may improve the insight gained from observations of nutrient flows, for better decision making.

Nutrient budgeting at two different spatial scales simultaneously requires scale specific information, however. Nutrient budgeting as a methodology is typically used to observe or monitor nutrient flows for individual experimental units, or at a single scale, typically in isolation [*see section 2.7*].

2.6 Nutrient use indicators and spatial aspects of nutrient cycling

Nutrient budgets or balances of the agri-environment, are typically constructed either (i) to increase the understanding of nutrient cycling in a given system, (ii) as performance indicators and for awareness raising in nutrient management and environmental policy, or (iii) as regulating policy instruments, to enforce a certain nutrient management policy in practice, or a combination of these (OECD, 1999b). Oenema et al., (2003) distinguished three main types of nutrient budgets; farm-scale (or “farm gate”), soil surface scale, and soil system scale budgets. A farm scale budget is the most common measure of environmental pressure, and commonly used as an environmental performance indicator. A soil surface budget is appropriate for estimating the net loading of nutrients on the soil for a given spatial location. Soil system budgets account for nutrient imports and exports, recycling of nutrients within the system, nutrient loss pathways and changes in soil nutrient pools; it is the most detailed budget and provides detailed information for nutrient management.

Nutrient budget/balance (Equation 2-1) and nutrient use efficiency (Equation 2-2) are calculated as follows:

$$Balance\ (kg\ ha) = \frac{(Nutrient\ imports - \ exports\ (kg))}{Spatial\ area\ (ha)}$$ **Equation 2-1.**

$$Efficieny\ (\%) = \frac{Nutrient\ exports\ (kg)}{Nutrient\ imports\ (kg)} X\ \frac{100}{1}$$ **Equation 2-2.**

Ultimately, nutrient use indicators are used to benchmark environmental performance (OECD, 1999b) by policy makers and other stakeholders as an approach to decrease nutrient losses from agriculture. However, the spatial extent or boundary of the nutrient cycling system being observed has an impact on the accuracy of what is being measured and influences what data has to be assumed or estimated in the budget. Leip et al., (2011) highlighted this for farm and catchment scale, and Aarts et al., (2000) highlighted it for farm, field, and soil system scales. Nutrient indicators at soil or field scale require more detailed data at a much higher spatial resolution; field scale manure application and fodder nutrition data, for example. The methodology for a soil balance incorporates all nutrients that are imported via fertilisers and exported from the soil via harvested products (Eurostat, 2013).

Each of these scales differs in spatial resolution, and as a result, methodological requirements. Figure 2-5 shows the nutrient cycles when two spatial boundaries are selected for observation simultaneously, farm and field scale. The farm scale boundary is indicated by the solid line, if farm scale was observed in isolation, everything inside this boundary line would be invisible, i.e. the within farm nutrient flows (field boundary dashed line) would be unknown. This highlights the need for more than one scale when observing indicators of nutrient budgeting strategies or balances.

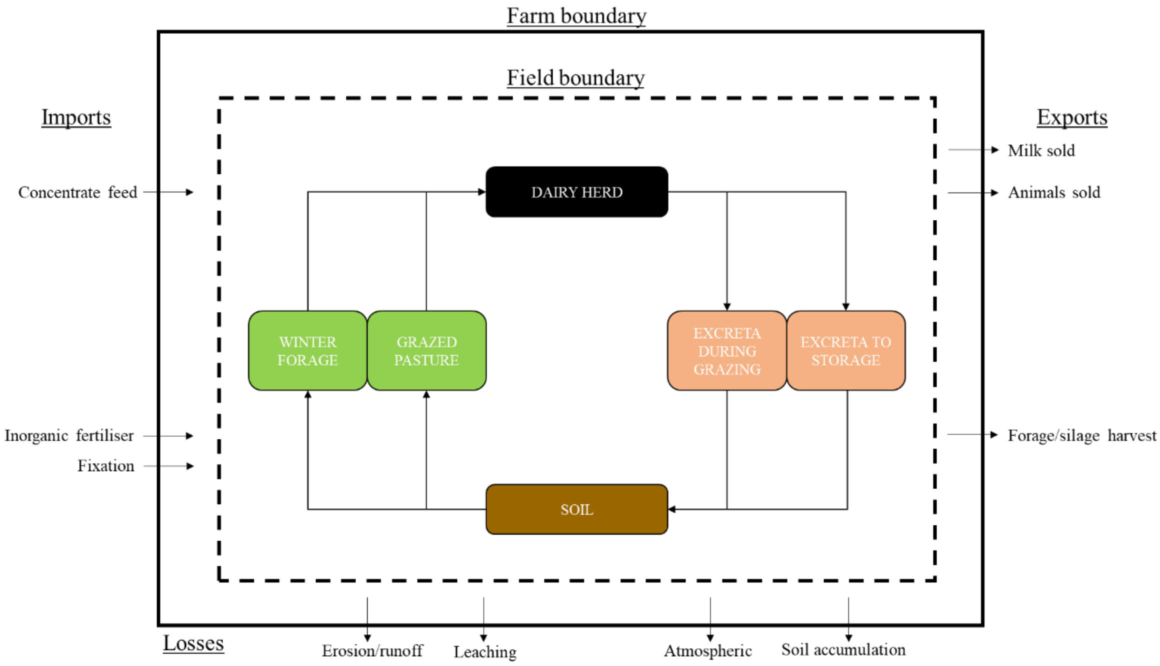

Figure 2-5. Animal–Manure–Soil–Plant nutrient cycle with farm and field scale spatial boundaries delineated and including nutrient import, export and loss variables. Adapted from Oenema et al., (2015).

Ultimately, it might be expected that indicators of sustainability at the farm scale will be strongly influenced by actual nutrient management at the field scale. And yet, data on nutrient management and sustainability indicators at the field scale are typically not included in studies (Buckley et al., 2016a; Einarsson et al., 2018; Mounsey et al., 1998; Soberon et al., 2015).

2.7 Nutrient budgeting scales

There are different capacities to accurately estimate nutrient losses at different nutrient budgeting scales due to different spatial and temporal limitations of each scale, all of which can be viewed within the context of the NTC. There are potentially important advantages and disadvantages at each scale that are summarised in the following sections.

2.7.1 Plot scale studies

These component studies and/or approaches of addressing nutrient losses are limited because they typically only examine a single environmental or management factor. For example, plot scale studies typically study a single component of management such as reseeding (Haygarth et al., 2005; Necpálová et al., 2013 and Shalloo et al., 2011) grassland cropping strategies (Hanly et al., 2017; Eriksen et al., 2015) or sward composition (Bryant et al., 2017; Totty et al., 2013; Woodward et al., 2013). Environmental factors are sometimes explored together however, such as nutrient losses on different soil types and gradients (Houlbrooke et al., 2003; Sharpley, 1996). In these cases, the management or environmental factor is often “controlled” or simulated.

Plot scale studies are limited spatially compared to the scale of actual losses in a field. Plot areas vary between 0.3 and 0.5 ha, with multiple replications of one or more experiments and a control, or are based in lab using samples from the field. This leads to the necessity to “scale up” the results. Averaging or scaling can be carried out by simply developing the calculations needed to represent a whole field or farm, but this allows uncertainties and a loss of information to occur (Mu et al., 2017; Oenema et al., 2015; Oenema et al., 2003). Due to a lack of knowledge of the impact and of interactions and additional environmental or management factors at the new larger or smaller scales, the applicability of the results to commercial farms is restricted.

Plot scale studies can also be limited temporally. Depending on the intensity of the monitoring, studies can be limited to a single growing season, or winter season drainage period for example, and can’t or don’t replicate the procedure for another season. Generally, a longer study period is needed for plot scale studies, to either explore affects over a longer term, or to explore the added influence of other environmental or management factors, or both. Exceptions however, are where plot scale studies are used to explore soil fertility over the long term (> 20 years) (Johnston, 1997; Blake et al., 2000; Paul et al., 2019).

The advantages of plot scale component studies however, are that they can often replicate or simulate multiple parts of the nutrient transfer continuum, which farm and field scale studies cannot. Measuring nitrogen and phosphorus losses from artificially drained plots on a research farm for example (Houlbrooke et al., 2003), in theory represents all aspects of the continuum; source, mobilisation, pathway, delivery and at least the potential nutrient total to reach the receptor. Other studies aim to address the changes at the source level only (Woodward et al., 2013; Bryant et al., 2018). Plot studies are also used to generate typical or benchmark values, and can be used to validate farm models (Huhtanen et al., 2011; Nousiainen et al., 2011; Potter, 2006). Farm model studies have to make assumptions, and these plot studies can help to improve the accuracy of those assumptions. Overall, studies at plot scale have a high capacity to capture and represent actual losses, but are limited temporally and sometimes spatially. This is due to the difficulty in scaling up the results accurately, which negatively affects the applicability of the results.

2.7.2 Farm scale studies

There may be an over reliance on fam scale nutrient balance studies and approaches to optimise nutrient use, when in fact, it does not budget nutrient flows at field scale or the spatial and temporal aspects of environmental and management factors. In terms of the nutrient continuum, farm scale studies are based entirely at the source level. This is effective for detecting changes in nutrient management, before impacts are observed at the receptor level, as shown by Murphy et al., (2015) and in agreement with (Cherry et al., 2008 and Sharpley et al., 2009). This does not capture actual losses from the farm which, in reality, are expected to occur, and can be better managed at, the field scale or smaller. However, there are both advantages and disadvantages of farm scale only studies and approaches.

The farm scale works well as a guideline to improving nutrient management practices (Nevens et al., 2006). Lanyon and Beegle (1989) showed that the correction of farm scale nutrient deficits were needed to manage high stocking rate densities and soil deficiencies. It has since been shown that the use of information from farm scale nutrient balances has been effective in reducing nutrient surpluses on livestock farms (e.g. Hanegraaf and den Boer, 2003; Swensson, 2003). As a result, farm-gate balances are considered a useful and reliable indicator of nutrient use efficiency and the potential environmental impact risk of nutrient use (Öborn et al., 2003).

Nutrient balances are so widely used they have become incorporated within national action programs to reduce nutrient losses from agricultural sources. The Netherlands has used N and P nutrient budgeting systems since the 1980s, firstly as a voluntary management tool for dairy farmers. The approach was then developed into a tool for policy makers. MINas (MINerals Accounting System) now plays a significant role in the Dutch implementation of the Nitrates Directive (Hanegraaf and den Boer, 2003; Schröder et al., 2003).

The generally high availability of farm level data and information makes it easier to communicate the agri-environmental performance of farms to farmers, advisors and policy makers at this scale. This however, may contribute to a high expectation of farm scale nutrient balances to reduce actual nutrient losses (Öborn et al., 2003), which are not clear at farm scale alone, as shown by the NTC (Haygarth, et al., 2005). Dairy systems have substantial internal nutrient flows, and accounting for the processes that determine nutrient flow, may greatly affect total nutrient use efficiency, and consequently, nutrient losses. Farm scale alone contributes little in recognition and quantification of internal nutrient flows, sometimes leaving them out entirely (Watson et al., 2003). There is a need for linking farm scale nutrient budgets to internal nutrient sources and flows.

Additionally, Mulier et al., (2003) highlighted major limitations that made it difficult to accurately calculate farm scale balances. These included large variation between actual and reported nutrient composition of concentrate feed, uncertainties in the estimates of the quantity and nutrient composition of organic manure, and in the assessment of changes in stocking rates on the farm, and the accuracy of the data supplied by farmers. Furthermore, these uncertainties are rarely quantified or even estimated (Oenema et al., 2003 and Swensson, 2003). One potential way of addressing this, is to ensure standardised and uniform approaches are used or to include multiple scales when nutrient budgeting. Goodlass et al., (2003) identified more than 55 nutrient budgeting systems across EU member states and found that almost none of them used confidence intervals or variation coefficients. This highlights the necessity to use farm scale budgeting approaches carefully. Schröder et al., (2003) and Öborn et al., (2003) suggested increased availability and use of reference values, and/or further subdivisions of farm characteristics, for example, comparison of dairy farms within sub-categories of milk production per ha, rather than a blanket comparison of dairy systems. This may likely make evaluation of results and agri-environmental targets more refined. A recent example of this is Thomas et al., (2020), whom establishes nationally representative farm scale benchmarks under different farm characteristics, including by soil group and production intensity, in an Irish context.

There is only a weak relationship between farm scale surpluses and environmental impacts (Öborn et al., 2003). The use of nutrient balances for example, by policy makers, to meet environmental targets for nutrient management in agriculture, makes the assumption that there is a relationship, direct or otherwise, between nutrient surpluses and their environmental impact. The assumption that follows, is that a reduction in total nutrient imports will result in a matching reduction in losses, and therefore also the impact on the wider environment (Öborn et al., 2003). Nutrient balances and surpluses actually provide a measure of the total nutrient loss potential, not the actual losses (Öborn et al., 2003). As shown by the NTC concept, there are any number of factors that could impact the actual losses from the farm. Although a general relationship exists, there is less evidence to support specific relationships between balances or imports of N (Bechmann et al., 1998) and P (Edwards et al., 2000) and actual nutrient losses. Similarly, N balances and even long-term balances over a number of years have limited strength in predicting nitrate concentrations in waterways because of the intervening influence of soil and crop processes

(Öborn et al., 2003). In the case of N, soil nitrate concentrations are not based entirely on accumulated soil N, but on the short-term temporal balance between N mineralisation, immobilisation and crop demand (Withers and Lord, 2002).

The inclusion of mobilisation and temporal factors at farm scale is needed. There are fundamental differences between N and P cycles in the soil, and in their capacity to move through the landscape, as shown already [*see section 2.3*]. At the source level, reductions in N and P surpluses alone may not be sufficient. Studies have already highlighted the importance of soil erosion and physical transfer of P with soil particles to waterways (Sharpley et al., 1995; Sharpley and Smith, 1990). Many other studies stressed the importance of fertiliser imports (Schröder et al., 2007, Ledgard et al., 1999; Egan et al., 2018; Schröder et al., 2010), soil P concentrations (Heckrath et al., 1995), and organic manure inputs (Graetz and Nair, 1995; Lanyon, 1994). The importance of organic manure management at the field scale due to its nutrient content variability (O'Bric, 1991; Berry, P. et al., 2012; Berry, P. B. et al., 2012) and especially in relation to increased stocking rates the application rate and timing (seasonality) is extremely important to prevent losses to vulnerable receptors, air or water.

Assessing farm and field scale information simultaneously, could be a more effective strategy to address the mobilisation and seasonal/temporal aspects of internal nutrient flows. Greater consideration for the temporal aspect of nutrient loss for example, may be necessary to fully understand linkages between specific sources of nutrients and some environmental impacts. Accurate field scale source apportionment, and targeting of management practices, to reduce nutrient loss during particularly hydrologically sensitive times of the growing season, may be more cost effective than blanket reductions in nutrient use (Öborn et al., 2003). Some efforts have been made to address this. Edwards et al., (2003) described more targeted management options to address nutrient enrichment in an estuary of a designated nitrate vulnerable zone in north east Scotland. Also, under another nitrate vulnerable zone in England, measures to reduce the surplus have been combined with improved management techniques such as better timing of manure inputs (Lord et al., 1999). In terms of addressing mobilisation factors, for example, management of land cover and manure with regard to soil erosion and run-off rates are likely to have a short-term measurable impact on reducing P loss (Withers and Lord, 2002; Edwards et al., 2000). Overall, farm and field scale approaches should be tailored to allow synchronisation of nutrient supply and nutrient demand of plant temporally and spatially.

If data were more available, and methodologies standardised, disaggregating farm nutrient balances into different production units (Bengtsson et al., 2003) or animal housings (Gustafson et al., 2003) may help to assess the agri-environmental performance of the different units. Aarts et al., (2000) highlighted the importance of the assessment of a farm's different subunits, which identified weaknesses in existing nutrient management strategies. Again, however, with increased level of detail, there is potentially increased levels of uncertainties and incorrect assumptions if left unchecked (Watson and Atkinson, 1999; Oenema et al., 2003). Subunits would need to be held to the same high characteristic and data availability standards as farm scale budgets should be (Öborn et al., 2003).

In any monitoring and policy evaluation approach, it appears to be extremely important to provide nutrient data across the continuum from source to delivery and impact and at a manageable scale (e.g., (Wilcock et al., 2013) to ensure that the interpretation of water quality responses to policy are fully informed Murphy et al., (2015). Where data are collected at a scale not conducive to the specific goal, it is likely to lack in the resolution needed to clarify the issues of that goal (Harris and Heathwaite, 2005; Kirchner, 2003). Murphy et al., (2015) in a study that looked at P mitigation measures on Irish dairy farms, suggested that, even if sufficient P was being imported at the farm scale, in many cases, this P was not distributed adequately at field level to where it was required, even with soil P test values available to farmers. This highlighted the potential for further improvement in P management, to match P application to P requirement at the field scale. It was also highlighted in this study, that there was strong requirement for P management of advice and support at the field scale, not just farm scale.

To summarise, the capacity of the farm scale to represent, or indicate, potential nutrient losses is high. However, with increased intensity, environmental pressures increasing and a number of weaknesses in the methodology of farm scale alone, there is a need for higher resolution management, potentially through multiscale approaches. Simultaneous farm and field scale assessment may still not be sufficient to address actual nutrient losses, but it may start to allow the incorporation of, for example, mobilisation and temporal factors, which are very important to reducing surpluses and losses in a spatially explicit way. This use of field scale information would however, also carry with it, the risk of uncertainties and wrong assumptions, that farm scale studies are also vulnerable too, unless strict standardised methods are used and standard values made available too. Although not limited spatially, due to the methodology representing a whole farm rather than a proportion of it, the combined approach may still be limited temporally. It is typical to use annual data for calculating balances, but the combined approach may allow the start of addressing internal nutrient flows within year, such as between seasons or between high mobilisation risk periods.

This combined approach also eliminates the risks associated with scaling between farm and field scale budgets, however scaling up to catchment levels would likely still not be accurate or appropriate. The field scale data might allow a better distribution strategy to be developed however, which may help target unnecessarily high surpluses, for example on critical source areas or fields with high soil fertility, which might be expected to help reduce the losses to the wider environment.

2.7.3 Farm and field scale studies

Where research has been carried out on farm and field scale simultaneously, there are still some important limitations. Aarts et al., (2000) carried a detailed analysis of N use at farm, field and whole system scale of a research farm and found a small difference between farm and field. This, however, was attributed to N stocks on the farmyard and N losses from livestock buildings and manure storage systems, which highlights the need to also consider these factors when budgeting N use at the source level. It was then shown that the further partitioning of N surplus at a soil system scale provided insight into soil N stocks and values on denitrification, leaching, runoff and unaccounted N. This degree of N surplus partitioning however was only achieved through the use of a research farm as the experimental unit and a modelling tool (MINas). Therefore, partitioning N surplus from field scale may be limited to the type of farm being analysed, and the availability and experience of using modelling software.

Another limitation is the use of research farms as experimental units. Such farms are likely to have greater economic and knowledge transfer resources than a typical commercial dairy farm and may also have a legacy of operating with different nutrient strategies, sometimes outside of the nutrient regulations (at plot scales) of commercial farms for experimental purposes. Experimental farms simply have the facilities to monitor field imports and exports more accurately than commercial farms which enables them to make precise management decisions at the field level (Oenema et al., 2015; Van Leeuwen et al., 2019). Additionally, there is the risk of paying too much attention to elements that appear less relevant in practice or that insufficient attention is paid to issues that prove important later (Van Keulen et al., 2000).

Other studies that explored combined farm and field scale assessment (Gourley et al., 2012a; Dungait et al., 2012; Withers et al., 2001; Aarons et al., 2015), have distinguished subunits by crop type (for example grassland or arable). However, in the case of grassland, these studies did not distinguish between subunits within grassland. This may be important for providing insight into nutrient cycling differences between areas of the farm in relation to management practices. Furthermore, where soil fertility levels are measured in a study, some studies take a sub sample of fields to represent field scale. This brings with it the risk of loss of information by averaging and also potentially limits the insight into spatial distribution at the field scale, when not all fields are represented.

Furthermore, knowledge of the distribution of soil fertility levels, is needed to refine N and P surplus management. Soils with low soil fertility rely upon a nutrient surplus to build soil fertility (Horrocks and Vallentine, 1999; Wall et al., 2015; Buckley et al., 2016b; McDonald et al., 2019). In some cases, soil types with high N mineralisation potential, and with a high soil P test value, can sustain negative N and P balances and still produce sufficient yields (Jones, et al., 2013). In these cases, a blanket nutrient reduction at source level could make building or maintaining soil fertility levels difficult. Furthermore, budgets tend to focus on N and P, but K has a very important role in nutrient cycling for animal and crop production and can significantly affect utilisation of N and P in grassland systems (Kayser and Isselstein, 2005). For example, Bučiene et al., (2003), carried out farm and field scale K balances for a livestock system and illustrated the important differences between external and internal nutrient flows for K.

To summarise, spatial limitation farm and field scale in nutrient budgeting approaches, is dependent on the choice to represent all fields on a farm, or a subsample of fields. Temporally, most nutrient budgeting approaches still rely on annual totals, and temporal data within season is typically absent. There is the advantage of partitioning farm scale surpluses, and with this, a gain in spatial distribution insight which may contribute to improving overall distribution and identifying areas that pose the highest risks in terms of nutrient losses potential. Further partitioning of field scale surpluses is limited to modelling tools and availability and resources which, however, may not be available, for example, on commercial farms.

2.7.4 Sub field scale/precision nutrient management studies

According to Hedley (2015), precision agriculture focusses on improving nutrient use efficiency at the appropriate scale and requires the following: 1) appropriate decision support systems such as digital prescription maps, 2) computer equipment capable of varying application rates to different scales. A reliance on technology for precision agriculture is also highlighted by Cook et al., (2000), who states that precision agriculture benefits from the co-development, accessibility and affordability of several technologies, including global positioning systems (GPS), geographic information system (GIS), increasingly smaller computer components, automatic control, in-field and remote sensing, mobile computing, advanced information processing, and telecommunications. Furthermore, remote sensing technologies can be used to delineate hydrologically sensitive areas, and use them to determine Critical Source Areas (CSA's) (Shore et al., 2014; Thomas et al., 2017) which then enables high resolution and precision in nutrient management, provided the required technology is available to do so. Many of these technologies have been predominantly developed for tillage systems (Zhang et al., 2002), and so their applicability to grazed system can be reduced. Despite this, for grazed grass systems, Hedley, (2015), showed that aerial variable rate application, unmanned aerial vehicles (for landscape mapping), targeted dairy effluent irrigation and even animal tracking technologies are potentially eligible options for improving nutrient management import rates and distribution. Furthermore, Higgins et al., (2019) showed that grass growth monitoring, yield monitoring and soil sensor technologies are available for grazed grass systems also. However, despite the increased availability of the precision agriculture option, there is still a lack of widespread uptake on specialised dairy systems, mainly due to the need for more research or costs (Higgins et al., 2019). The cost of precision agriculture technological equipment, farmer income levels and access to skills or training required for the digital tools were also highlighted as major constraints to the uptake of precision agriculture approaches by farmers (Kritikos, 2017; Zhang et al., 2002).

The concepts of precision agriculture, improving nutrient use efficiency at the appropriate scale, and, the use of delineated zones, are very applicable to grazed grass systems however. For example, the paddock system, a component of good grazing infrastructure is already adopted by Irish grass-based dairy system and is established as a strategy to improve herd performance and grass utilisation (Kavanagh, 2016a). The implementation of the paddock system is similar to the concept of precision agriculture because it delineates the whole farm into smaller more management areas or zones. Paddocks, the smallest unit of uniform management on specialised dairy systems, may be a valuable link between farm and field scale. During the paddock planning process, factors such soil type and slope are broadly taken into account when designing the paddock layout, although, the main focus is on distance from parlour (Kavanagh, 2016a). Therefore, on grazed grass systems, the paddock scale may, in effect, represent the concept of a precision agriculture approach, without the cost of expensive technology. The strategy in which paddocks are grazed, such as strip grazing within paddocks, is an added complexity to this concept link however. The issue is that nutrient management on Irish grassland utilises “blanket” applications across the entire farm area during the growing season, which can make record keeping easier (Lalor and Humphreys, 2016), but increases nutrient loss risks in high rainfall periods and is less targeted towards plant nutrient uptake rates (Brockman, 1974). Blanket application of nutrients across several fields don’t take consideration of spatial variation in nutrients and can be counter production for yield and environmental targets (McCormick et al., 2009), in practice it should constrained to specific times of the year only (McNamara, 2019), if not avoided altogether.

2.7.5 Modelling studies

Farm modelling studies are not majorly limited spatially or temporally, depending on the technology, methodology and goal of the study, and as a result modelling studies can examine multiple aspects of the NTC, or have the capacity to scrutinise an individual aspect rigorously.

As a result, modelling studies have assessed the source level budgeting (Schils et al., 2007; Buysse et al., 2005; Adenuga et al., 2018b; Kuipers and Mandersloot, 1999; Khalil et al., 2019), environmental factors (Whitehead et al., 2007; Veltman et al., 2018; Adenuga et al., 2018a) and management factors (Nousiainen et al., 2011; Huhtanen et al., 2011; Monaghan et al., 2008) to a high degree. On top of this, modelling studies have the advantage of estimating losses, where measuring them is too difficult or costly (Khalil et al., 2019; Bragina et al., 2019), and even aspects outside of farm scale for example, response time to market signals (Yang et al., 2020). The other advantage is the capacity to model measures or scenarios that may have a significant impact on nutrient losses without the cost and risk of error of actually implementing them (Basso et al., 2015). Additionally, modelling studies allow multiple repetitions and very large sample sizes (Veltman et al., 2018). In effect, the user is in control of the spatial and temporal limitation of the study, rather than the actual experimental unit being explored in real time controlling the limitations.

The limitations of farm modelling studies however, are in the data availability/requirements, quality/uncertainties of data, and risk of being subjective in model inputs or using best case assumptions. Using data for modelling requires a number of important standards, which may not always be met. For modelling pastoral systems, for example, an assumption might be made that the N soil pool is zero at the start of monitoring (Ledgard et al., 1999). Models are further limited by the amount, and quality, of the data available at the time. Simple or basic models are often added to other simple models and so on, leading to substantial complexity and a requirement for superior knowledge of the all the limitations of individual models (Cichota et al., 2010). As a result, parameterisation of model inputs is required. Modelling approaches will always need experimental datasets to keep checks and balances in place for outputs to be valid (Shalloo et al., 2004b). The need for additional model development as well as the need for more and better-quality data was emphasised by (Holzworth et al., 2015; Jones et al., 2017; Boote et al., 2013; Basso et al., 2001; Basso et al., 2015). Extensive validation processes are essential if models are to be used to predict actual, not potential loss, however few models have this capacity (Cherry et al., 2008). Source level budgeting with models that can utilise land use and livestock numbers data can estimate nutrient applications from national datasets. However, the transport processes of the continuum are not always considered, resulting in the need for back calculations for estimations (Cherry et al., 2008).

To summarise, farm modelling studies have a high capacity to represent actual losses due to their flexibility in scale, temporal and spatial capabilities. However, models are severely limited by data availability, and data quality, as well as complexity issues, but this depends on the goal.

2.8 Conclusion

A review of the pertinent literature shows that source level budgeting is sufficient for observing, for example, changes in policy relating to nutrient use. Studies reviewed indicate that there is need for a better nutrient budgeting tool at the source level, that address the spatial distribution of nutrients within farms. Additionally, it was also found that addressing this issue of spatial distribution may require using a multi-scale approach, rather than a mechanistic or component level study. This is because, there are limitations with using one scale only. For example, farm scale studies don't take into account important within farm (i.e. field scale) spatial, temporal and mobilisation factors. There may also be an over-reliance on farm scale information as a basis for improving nutrient management at field scale. Plot studies provide important information on nutrient responses, agronomic sufficiency and environmental optimum nutrient thresholds, however these results can have limited applicability to commercial farms because of high levels of variability in soils, environmental and management, as well as the associated risks of inaccuracies relating to scaling up of the results. Precision technology is well established at this stage, but it is still limited by economics, and is not entirely applicable to livestock systems yet either. Modelling studies enable important measures and scenarios to be tested, but they still require validation with quantitative data, and can be limited by assumptions in data, which lead to uncertainties in results and conclusions. Farm or field scale combined studies also pose the risk of uncertainties due to the nature of the data requirements and extrapolation to other farms and regions. However, the simultaneous evaluation of farm and field scale nutrient cycling may reduce uncertainties. This is because bridging the knowledge gap through establishing the linkages between these scales, may allow a better insight into the nutrient loss risk potential, as nutrients flow between the farm boundary and the individual field boundaries. There are few studies that utilise commercial farms as experimental units, or delineate subunits within them, which provide an opportunity to understand the actual variability within and between end users of the information and may help focus the applicability. Finally, in the absence of actions that address the issue of scale in relation to nutrient management, it can be expected that water quality will continue to decline, air quality will deteriorate and GHG emissions will rise. Therefore, new knowledge needs to be established in relation to the requirements and recommendations for appropriate nutrient management between scales, so that farmers are supported in their management decisions, that advisory services can guide best practice on farms and that policy makers can effectively develop legislation, all of which benefits the agri-environmental sustainability of dairy systems. Such information would not be limited to dairy systems only but to other livestock systems internationally.

2.9 General material and methods

The general material and methods section contains a list of the data and variables that were measured, collected, or surveyed for this thesis. This section does not explain the analysis or equations used in each chapter but instead serves as a compendium of all the data and variables that were collected. This section should therefore be read prior to, or in conjunction with, each subsequent experimental chapter.

Table 2-1 groups the field and farm and regional scale variables under three main categories: Soil (S), Environment (E), and Management (M) factors. Table 2-2 lists the variables used to calculate nutrient cycling indicators and defines what each nutrient import and export variable represents in this thesis. Table 2-2 also includes the details of two other calculated variables: change in soil test and herbage nutrient uptake. Table 2-3 lists the nutrient values used for concentrate feed, milk sold, and animals sold. Table 2-4 explains the weighting procedure used to weight the field scale nutrient cycling indicators. Finally, Appendix 1B – Coefficient values, should be referred to for information on stocking rate nutrient value (Table 10-1) and organic manure nutrient value (Table 10-2).

Table 2-1 List of farm, field, and regional scale variables, the frequency of which these data were measured/collected/surveyed, and the methodology/analysis that was carried out to capture the data

Variable	Measured/collected/surveyed frequency	Methodology/analysis
	Soil factors (S)	
Soil chemical properties (mg l^{-1}): Aluminium (Al), calcium (Ca), copper (Cu), iron (Fe), magnesium (Mg), manganese (Mn)	2015 soil samples	Lab analysis – Mehlich II (Mehlich, 1984).
Soil organic matter (%): Soil organic matter (SOM)	2015 soil samples	Lab analysis – Loss on Ignition (S.I. No. 605 of 2017, EU 2017).
	Environmental factors (E)	
Soil drainage class: Well, Moderately, Poorly	2015 and 2016 – multiple farm visits	Topographic features, denoted by contour lines, were hand drawn onto paddock scale farm maps in 2015 to identify areas of homogenous characteristics (flat, sloping, undulating). Seasonal observations of soil surface drainage conditions (waterlogging and/or excessively dry areas of paddocks) were also hand drawn. These delineated areas were then surveyed by auger and soil texture analysis. One auger was carried out per area unless there were significant landscape differences. Results of the soil texturing were then grouped into three drainage categories. (Creamer et al., 2015).

Variable	Measured/collected/surveyed frequency	Methodology/analysis
Gradient class: Level, Gradient	2015 and 2016 – multiple farm visits	Field analysis – Areas where topographic slope was greater than 10° or flat where were grouped into two separate categories (Roberts and Watkins, 2014).
Management factors (M)		
Field nutrient import ($kg\ ha^{-1}$): Nitrogen, phosphorous, potassium, lime	2015 to 2017 – annual farm visit (calendar year: January to December)	A nutrient record book was collected annually at the end of each calendar year (January to December) for 2015, 2016 and 2017. Organic manure and inorganic fertiliser nutrient data was collected as rate, date, and field/paddock ID. Follow up phone calls to participating farmers were made quarterly to ensure all details were being entered adequately. Where participants were already using online software packages for recording farm information, this data was used with permission instead of a nutrient record book. All data were collected at field and paddock scale. An anonymised database was made to store all nutrient import information.
Soil fertility ($mg\ l^{-1}$): Soil test pH, soil test phosphorous (STP), soil test potassium (STK), and soil test magnesium (STMg)	2015, 2016, 2017 and 2018 – annual farm visit (calendar year: January to December)	Field/paddock scale soil fertility testing was carried out according to S.I. No. 605 of 2017 (EU, 2017). Lab analysis to determine soil nutrient content was based on Morgan's extractant methodology (Peech and English, 1944).
Sward species composition (%): *Lolium perenne,* Perennial rye grass (PRG) *Holcus lanatus,* Yorkshire Fog (YF) *Poa annua,* Annual Meadow Grass (AMG) *Trifolium repens,* White Clover (WC) *Stellaria media,* Common Chickweed (CC) *Rumex obtusifolius,* Broad leaf Dock (BLD) *Taraxacum officinale,* Dandelion (D) *Carduus crispus,* Welted Thistle (WT) *Urtica dioica,* Nettle (N)	2017 – single farm visit in spring (April and May)	A 0.5 m X 0.5 m quadrat with string subdivisions (see Appendix 1A) was systematically placed three times in a diagonal line moving down the long axis of each field and paddock. A percentage proportion was recorded for each plant species by visual estimation using the string subdivisions, an adaption of the DAFOR methodology for herbage surveys (Kent et al., 2001). Individual herbage species were grouped into different proportions based on the average area proportion occupied by the species in each of the three quadrat recordings e.g. 0-33% (Low), 34-66% (Medium), 67-100% (High).

Variable	Measured/collected/surveyed frequency	Methodology/analysis
Geranium mole, Doves foot Cranes Bill (DFCB) *Ranunculus repens,* Creeping Buttercup (CB) *Cardamine hirusta,* Hairy Bittercress (HB) Lamium perpereum, Red Dead Nettle (RDN) *Capsella bursa-pastoris,* Shepherds Purse (SP) *Plantago lanceolata,* Ribwort Plantain (RP) *Trifolium pratense,* Red clover (RC) *Cerastium fontanum,* Common Mouse ear (CME)		
Sward characteristics: Sward cover (kg DM ha^{-1}) dry matter (%), and nutrient concentration (%)	2015 and 2016 – Farm visits in spring (March and April), summer (June and July) and autumn (September and October)	A 0.5 m X 0.5 m quadrat was semi-randomly placed three consecutive times in a diagonal line, moving down the long axis of each paddock, to get a representative sample. Three individual samples were cut from each paddock, using a battery powered grass shears (Bosch ASB 10.8 V), at a height of 40-45 mm from the soil surface, to simulate grazing height residual (O'Donovan and McEvoy, 2016). Individual samples were wet weighed first and then pooled, to make a 100 g sub sample to represent the whole paddock. The sub sample was dried at 40°C for 48 hrs to determine the sub sample dry weight. Dry matter content (DM%) was calculated as dry weight as a proportion of the 100 g sub sample. Herbage cover (kg DM ha^{-1}) per paddock was calculated using the DM% and the average weight of the individual herbage samples. Lab analysis - The dried herbage sub sample was milled through a 0.2 mm sieve and analysed for nutrient concentration using a LECO Truspec and Elemental Macro Cube instrument (LECO Corporation, 2020) for herbage N, and HNO3 digestion followed by ICP-OES for the remaining elements P,K, Ca, Mg, Cu, Mn, Zn (LECO Corporation, 2020).
Cropping rate: Grazing Only, Grazing + Harvest, 2 Cut Harvest	2015 to 2017 – annual farm visit (calendar year: January to December)	Information was collected from farmers by personal communication or by farm information recording software where available.

Variable	Measured/collected/surveyed frequency	Methodology/analysis
Reseeding interval: < 5 years, 5 to 10 years, > 10 years	2015 to 2017 – annual farm visit (calendar year: January to December)	Information was collected from farmers by personal communication or by farm information recording software where available. (Where a specific date of reseeding could not be provided, a range in time was used instead e.g. more than five years or more than ten years since reseeded).
Pasture production: Yield (kg DM ha yr^{-1}), No. of grazing's (count), No. of harvests (count)	2015 to 2017 – annual farm visit (calendar year: January to December)	Participating farmers recorded pasture production weekly by grass measuring. Grass measuring was carried out by cut and weigh, plate meter or "eye ball" estimation. Professional training and cut and weigh equipment (battery shears, quadrat and weigh balance) was provided to first time grass measuring farmers. Follow up phone calls to participating farmers were made monthly to ensure all details were being entered adequately.
Subzone: Dairy, Non Dairy, Silage	2015 to 2017 – annual farm visit (calendar year: January to December)	Information was collected from farmers by personal communication or by farm information recording software where available.
	Region and farm scale variables	
Regional weather: Rainfall (mm), solar radiation (J cm^2), air temperature (ºC), degree days	2015 to 2017 – annual farm visit (calendar year: January to December)	Met Éireann (Met Éireann, 2019) and local registered rainfall gauges.
Farm area and stocking rate: Organic manure stocking rate (kg org N ha yr^{-1}), livestock unit stocking rate (LU ha yr^{-1})	2015 to 2017 – annual (calendar year: January to December)	Nitrogen and phosphorus monthly and annual livestock organic manure statements were collected with participant permission from Irish Cattle Breeders Federation (ICBF) (ICBF, 2019). Farm level nitrogen, phosphorous and potassium fertiliser advice and farm holding maximum allowances as well as stocking rate and farm area was collected with participant permission from fertiliser plans/nutrient management plans generated by local advisors each year. Organic manure stocking rate was based on regulations (S.I. No. 605 of 2017, EU, 2017), see Appendix 1B.
Milk and animal production: Milk sold (L), milk solids (including fat and protein), animals sold (type, age and count)	2015 to 2017 – annual (calendar year: January to December)	Data on total milk sold and milk solids as well as data on animal numbers (herd size, animals sold, animal type, age and weight) was collected annually with participant permission from the ICBF database. Information was collected from farmers by personal communication or by farm information recording software where available.
Farm concentrate feed imported: Concentrate feed total (tonnes), concentrate feed crude protein level (CP%)	2015 to 2017 – annual (calendar year: January to December)	Information was collected as farm co-op fertiliser purchased statements/receipts from farmers by personal communication or by farm information recording software where available.

Variable	Measured/collected/surveyed frequency	Methodology/analysis
Farm fertiliser imported (tonnes): Nitrogen, phosphorus, potassium, lime	2015 to 2017 – annual (calendar year: January to December)	Information was collected as farm co-op fertiliser purchased statements/receipts from farmers by personal communication or by farm information recording software where available.

Table 2-2 List of calculated variables used as nutrient cycling indicators at farm and field scale and list of definitions for each nutrient import and export variable and well as other calculated variables

Variable	Methodology/Analysis
Nutrient cycling	
Nutrient balance (kg ha^{-1}): Nitrogen balance (NBAL), phosphorus balance (PBAL), potassium balance (KBAL)	Total nutrient import subtracted from total nutrient export per farm/ field area.
Nutrient use efficiency (%): Nitrogen use efficiency (NUE), phosphorus use efficiency (PUE), potassium use efficiency (KUE)	Total nutrient export as a proportion of total nutrient import.
Nutrient import	
Inorganic fertiliser (kg ha^{-1}): Bag fertiliser	Where inorganic fertiliser is synthetic fertiliser purchased and imported and spread per farm and per field area. Nutrient content/value was assumed to be 100% available.
Organic manure (kg ha^{-1}): Organic manure collected during housing period and farmyard/dungstead manure (cattle)	Where organic manure represents manure that was stored for approximately 16 weeks during the fertiliser spreading prohibition period and applied to fields during the 32-week growing period. It also includes the organic manure applied as farmyard or dungstead manure. Each form of organic manure type was assigned a different nutrient value according to Wall and Plunkett, 2016. Differences in nutrient value depending on the season organic manure was applied in were also accounted for – see Appendix 1B. Organic manure import total was therefore the combination of stored organic manure and cattle dungstead manure. Organic manure import total excludes dung and urine excreted by animals during the approximately 32-week grazing season. Organic manure was assumed to be spread using splash plate or dung spreader machinery.
Concentrate feed (kg ha^{-1}): Concentrate mixes containing protein	Where concentrate feed is the total feed purchased by the farmer and used for dairy herd (kg cow^{-1}). Nutrient value was based on the assumption that crude protein was 16%. Nutrient value was different for N P and K (Table 2-3) and total import depending on the tonnage of concentrate feed imported at the end of each calendar year. Nutrient import via concentrate feed was only counted for 32 weeks to avoid double counting in the nutrient value for organic manure collected during the 16-week storage period.
Nutrient export	
Milk sold (kg ha^{-1}): Total litres collected by milk processor	Where milk sold is the total milk sold off the farm at the end of the calendar year. Nutrient value in milk was based on individual farm average crude protein levels – see Table 2-3.

Animals sold (kg ha^{-1}): Total animals sold	Where animals sold is the total number, type and actual weight of animals sold at the end of the calendar year. Nutrient value was based on typical values used by McDonald, 2011 – see Table 2-3.
Harvest (silage) (kg ha^{-1}): Silage yield and herbage nutrient uptake	Where silage harvest is the total yield and average herbage nutrient concentration i.e. herbage nutrient uptake, at time of harvest. Average nutrient concentration of three herbage samples from spring summer and autumn in 2015 and 2016 were weighted by the total silage yield at the end of each calendar year. Average for each season was used to generate a value for 2017 nutrient concentration. Note that annual grazing yield was counted separately to silage yield as nutrients are not considered to be removed from the system in grazing scenarios. Nutrient export via silage harvests were counted for any field or subzone where the event took place and were not restricted to silage subzone fields only.
Other calculated variables	
Change in soil test (mg l^{-1}). Change in soil test pH, phosphorus, and potassium. "Change in" was denoted by Δ symbol	Difference between one year and the preceding year (2015 to 2018) for each soil fertility component.
Herbage nutrient uptake (kg ha^{-1}): Grazing yield and herbage nutrient uptake	Herbage nutrient uptake was based on total annual grazing and harvest yield per paddock recorded weekly by the farmer and the average herbage nutrient concentration from three herbage samples taken in spring, summer and autumn of 2015 and 2016. Average for each season was used to generate a value for 2017 nutrient concentration. Herbage nutrient concentration was analysed in the lab a LECO Truspec and Elemental Macro Cube instrument (LECO Corporation, 2020) for herbage N, and HNO3 digestion followed by ICP-OES for the remaining elements P, K, Ca, Mg, Cu, Mn, Zn (LECO Corporation, 2020).

Table 2-3 Nutrient coefficient values used for concentrate feed, milk sold, and animals sold

Variable	Collected[4] as	Nitrogen	Phosphorus	Potassium	Unit
Concentrate feed[1]	Tonnes	26	5	10	Kg tonne^{-1}
Milk sold[2]	Litres	0.0059	0.0009	0.0015	Kg l^{-1}
Animals sold[3]	Kilograms	0.024	0.010	0.002	Kg kg^{-1}

(1). Assuming a protein content of 16%, N value for concentrate feed was calculated using the total annual quantity multiplied by its crude protein and divided by 6.25 (McDonald, 2011). Phosphorus value was calculated as 0.5 kg phosphorus in respect of each 100 kg according to S.I. No. 605 of 2017 (EU, 2017). Potassium value was calculated based on the assumption that 184 g K day^{-1} from concentrate feed (to meet a requirement of 1.2% K kg DM^{-1}) was sufficient to meet animal dietary K needs (McDonald, 2011).
(2). Nitrogen in milk was calculated using the total annual quantity and multiplied by individual farms annual average crude protein value divided by 6.38 (ARC, 1994). Phosphorus and K values were calculated based on mineral composition values for milk as shown by McDonald, 2011.
(3). Nitrogen, P and K values were calculated by multiplying the total annual livestock weight in animals sold by the mineral composition values for animal products as shown by McDonald, 2011.
(4). Every calendar year 2015 to 2017.

Table 2-4 Field/paddock scale apportionment procedure used to weight concentrate feed, milk sold, and animals sold nutrient values at field scale

Variable	Methodology
	Paddock apportionment procedure
<u>Weighted values (kg ha^{-1}):</u> Farm scale milk sold, animals sold, and concentrate feed	Farm scale information on total nutrient value in milk sold, animals sold and concentrate feed were weighted by individual paddock weightings. Individual paddock weightings were calculated as: No. of grazing's/defoliations (count) weighted by paddock area (hectare) <u>Example:</u> • Dairy Paddock 1 = 8 grazing's multiplied by 1.66 ha = 13.28 weighting. • 29 paddock weightings sum total = 408.10 Each individual paddock weighting then had a proportion value relative to the sum of the subzone weightings. <u>Example:</u> • Dairy fields – 273.08 weighting • Non-Dairy fields – 86.32 weighting • Silage fields – 48.80 weighting • Total paddock weightings = 408.10 • Dairy Paddock 1 as a proportion of dairy subzone total weighting = 13.28 as a proportion of 273.08 = 0.05 (or 5%). These weighting values were then used to proportionally distribute the total farm nutrient values of milk sold, animals sold and concentrate feed to field scale for field specific nutrient cycling calculations.

CHAPTER THREE: A SPATIAL COMPARISON OF PADDOCK, FIELD AND FARM SCALE NITROGEN BALANCE AND USE EFFICIENCY ON COMMERCIAL GRASS BASED DAIRY FARMS

Preface: This paper has been prepared for the submission to the journal *Agriculture Ecosystems and Environment*. Formatting and reference style have been edited for consistency throughout the thesis. Figure and table captions have been assigned with a chapter prefix. Acknowledgements have been removed.

Keywords: Nitrogen cycling, Nitrogen indicators, Scale, Farm, Field, Paddock, Subzone

3.1 Abstract

Conventional agriculture is associated with imbalances in nitrogen (N) flows and N losses to the wider environment that can cause harmful impacts. This study compares the sustainability of N use (nitrogen balance and nitrogen use efficiency), between farm and field/paddock scale, on commercial grazed grass dairy farms in Ireland. The pastoral N cycle was evaluated in detail by subdividing the grassland area of each farm into three management subzones (Dairy and Non-Dairy (grazed) and Silage (cut for conservation)). A total of 15 farms (mean grassland area 69.3 ha), and 446 fields/paddocks (mean area 1.4 ha) were monitored and evaluated over three years (2015 to 2017). Nitrogen balance (NBAL) and nitrogen use efficiency (NUE) were determined at farm and field/paddock scale from recorded N inputs and outputs in inorganic fertiliser, organic fertiliser, feeds, milk, animals, and grass silage. While farm scale NBAL (mean 165 kg N ha^{-1}, CV 34%) and NUE (mean 28%, CV 39%), did reflect the average overall N source pressures (mean 139 kg N ha^{-1}, CV 77%) and use efficiencies (mean 57%, CV 67%) found at field scale, they did not accurately reflect or capture the variation in N flows, N source pressures, and likely resultant N losses to the environment, that occur at the individual paddock and field scale. Silage fields had the highest mean NUE (86%) and lowest N surplus (27 kg N ha^{-1}), followed by Dairy (57% and 144 kg N ha^{-1}) and Non-Dairy subzones (37% and 150 kg N ha^{-1}). Field scale environmental (gradient and soil drainage), and management (soil fertility (pH, P and K), sward species composition, cropping and reseeding interval) factors, contributed significantly to variation in NBAL and NUE between fields. Results suggest that there may be significant potential to improve N efficiency and reduce N source pressure, and losses to the wider environment, through management practices targeted at the field/paddock scale. Moreover, results of this study suggest that these management practices, might differ between subzones of these dairy farms; notably, between the grazed area of the farm (milking platform) and the silage area. On the milking platform, in particular, the paddock scale (ca. 0.5 – 2.5 ha) might be a suitable scale to target improved N management practices, as these are already established as spatially explicit management units. Farm scale policy or regulations (e.g. stocking rate limits, or farm level fertiliser restrictions), do not enforce or encourage the need to address within farm spatial variability. This has important implications for policies that target improved N management and reduced N losses to the environment.

3.2 Introduction

Nitrogen (N) is a key driver of plant productivity and plant protein content and has been fundamental to the significant yield increases associated with the “Green Revolution”. As N fertiliser became more available and cost effective, it allowed yields to increase and food production to expand. The global N cycle has significantly changed as a result of anthropogenic activity and will continue to change into the future (Eickhout et al., 2006). The modification of the N cycle for agriculture, causes imbalances, which can lead to N emissions to the wider environment in harmful amounts. This includes losses of ammonia (NH_3), nitrous oxide (N_2O), nitrate (NO_3). Ammonia contributes to acidification, eutrophication and atmospheric pollution. Nitrous oxide (N_2O), is a potent greenhouse gas that contributes to the depletion of the o-zone layer. Nitrate (NO_3) can leach from soils and pollute ground and surface waters and cause eutrophication. These losses of N from the soil-plant system, are negatively impacting air and water quality, as well as contributing to climate change, at a global scale (Cameron et al., 2013). Air pollution remains a serious concern for climate and human health (OECD, 2015), and contamination of water from agriculture costs billions of dollars annually (OECD, 2012).

To better manage N losses from agriculture, farmers, policy makers, and other stakeholders, may use N management sustainability indicators to benchmark environmental performance (OECD, 1999a). The rate of fertiliser N can be used as an indicator of the amount of N leaching, however Schröder et al., (2010) reported that surplus N balance (beyond plant N requirement) was a better predictor ($r^2 = 0.86$) compared to total amount applied ($r^2 = 0.11$).

Spatially, farm scale represents the Total Utilisable Agricultural Area (TUAA) of an individual farm. This typically means the farm is treated as a single homogenous unit, and the boundary of the system being examined, is the farmers ownership boundary. Nitrogen balance, as an indicator at farm scale, is useful because the majority of the data tend to be relatively easily available (such as sales, trade and production statistics), and can be built upon to represent national scale (Eurostat, 2013). The farm scale indicator is very versatile, and as a result, is used for many functions such as in agricultural monitoring and evaluation programs (Buckley et al., 2016a; Cela et al., 2014; Venturraa et al., 2008), forecasting impacts of nutrient management changes (Oenema et al., 2009), and policy making (Cook, 2018; Scoones and Toulmin, 1998). Catchment scale approaches, provided spatially relevant data are available, tend to be scaled up to represent country or even used to model global scale information (Bouwman et al., 2009; Grizzetti et al., 2007; OECD, 1999a) and can be used to inform policy makers.

Nitrogen indicators at soil or field scale require more detailed data at a much higher spatial resolution, for example, field scale manure application and fodder nutrition data. The methodology for a soil balance incorporates all N that is imported via fertilisers, and exported from the soil, via harvested products or crop residues (Eurostat, 2013). The boundary of the system being examined in this case, could be a field or sub field delineated area. Such N data are not typically readily available or easy to capture. However, it is at the field scale and smaller that much of the N in the agri-environment is managed, and that N cycling and losses to the environment are determined. It follows that efforts to improve N management, to minimise losses to the wider environment, need to operate at this scale. Ultimately, it might be expected, that indicators of sustainability at the farm scale, will be strongly influenced by actual N management at the field scale. And yet, data on N management and sustainability indicators at the field scale, are typically not included in such studies (Buckley et al., 2016a; Einarsson et al., 2018; Mounsey et al., 1998; Soberon et al., 2015). Indicators such as N balance and use efficiency at the field scale are often averaged, or modelled, from the farm scale (Carey et al., 2017; Topp et al., 2007; Van Leeuwen et al., 2019). As a result, there is a very significant knowledge gap around what is occurring inside the “farm gate” i.e. at the field scale. Understanding factors which affect N cycling at the field, or sub field scale, may better enable researchers, policy makers, and practitioners to be more spatially precise in implementing measures, or generating advice that would reduce N losses.

In a grazed grass farm system, in particular, fields may differ greatly in their management due to different animal stocking rates (SR), and rates of fertiliser being applied, as well as differences in the number of defoliations. This might be expected to lead to significant variability between fields, and at a sub field scale (Aarons et al., 2015).

The field scale, therefore, may capture variability in N cycling and losses, that occur within the farm gate. Previous studies have tended to categorise all grass fields as “pasture”, and fail to distinguish between the different N flows that pasture fields with different management strategies are subject to. Previous field scale studies have attempted to capture biophysical effects on N use indicators, but they often don’t carry out extensive sampling regimes (Aarons et al., 2015; Gibbons et al., 2014; Nimmo et al., 2013), or are not fully representative of commercial scale farm scenarios (Leip et al., 2011; Ryan et al., 2012). It has been shown that higher spatial resolution of N management can be achieved by precision agriculture approaches, using sub field scale information generated from a range of sensor and GPS technologies, to delineate management zones, and deliver improved fertiliser distribution and, ultimately, efficiency (Higgins et al., 2019; Zhang et al., 2002). Similarly, Critical Source Areas (CSA’s) (Shore et al., 2014; Thomas et al., 2017) can enable high resolution and precision in nutrient management. However, cost of precision agriculture technological equipment, farmer income levels and access to skills or training required for the digital tools, means that there are still major constraints to the uptake of this approach by farmers

(Kritikos, 2017) and, in Ireland, nutrient management is still often characterised by “blanket” approaches to large areas, or even the whole area, of a farm (McNamara, 2019).

Agriculture in the European Union (EU) member states operates under the EU Nitrates Directive (91/676//EEC) (EEC, 1991); a framework that sets out minimum basic agricultural practices to prevent agricultural Nitrogen (and other nutrients) entering ground and surface waters in harmful amounts. Ireland is designated as a whole territory nitrate vulnerable zone, whereas some member states only designate specific areas within their country. Spatially, it is notable that most of these measures are generally set at, and, are assessed, or monitored at, the farm scale. The regulations do outline at least that nutrient management should be carried out on a field by field basis, and should be sensitive to spatial and temporal aspects of soil and weather conditions in order to minimise risks of nutrient loss to water. Soil testing is required to an area no greater than 5 ha, at most every 5 years. Nitrates Action Plans and regulations have been in place since 2006 but water quality has been generally either static or slightly decreasing (EPA, 2018b) since their implementation, indicating that harmful levels of N are still potentially being lost from agricultural soil plant systems.

This paper aims to investigate N cycling within and between farm and fields, on grass-based dairy farms, and discuss any implications arising within the relevant contexts of the nutrient transfer continuum (Wall et al., 2011). Using N balance and use efficiency as indicators of the sustainability of N management, the aim of the study was to 1) compare sustainability of N management at farm scale to sustainability at field scale, and 2) investigate the biophysical and management factors that drive differences in N balance and use efficiency at the field scale. It was hypothesised that 1) N balance and use efficiency would differ significantly between farm and field scale, and that 2) N balance and use efficiency would vary significantly at the field scale and would be affected by several biophysical and management factors.

3.3 Materials and methods

3.3.1 Farm selection and farm characterisation

Farms were selected from the south and south east regions of Ireland, in counties Cork, Tipperary, and Wexford (Figure 3-1). These regions are typically associated with long annual growing seasons (~242 days), and a high proportion of well drained soils (NFS, 2017). Average annual rainfall for these regions ranged from 967 to 1,208 mm (2015 to 2017). Average air temperature ranged from 9.7 to 10.3 °C (2015 to 2017). Personal communication with agricultural advisors was used to identify dairy farmers willing to participate in the study. A number of farmers were already part of the Irish Agricultural Catchments Programme (ACP) (Fealy et al., 2010; Jordan and Shortle, 2013), which analyses the impact of Nitrates Action Programme (NAP) measures. A total of 21 dairy farms were selected for this study, but only 15 participated fully (provided sufficient data to carry out the full suite of analysis for this study). All participants were specialised dairy farms, with dairy cows contributing more than 3/4 of standard agricultural output, by value (CSO, 2016).

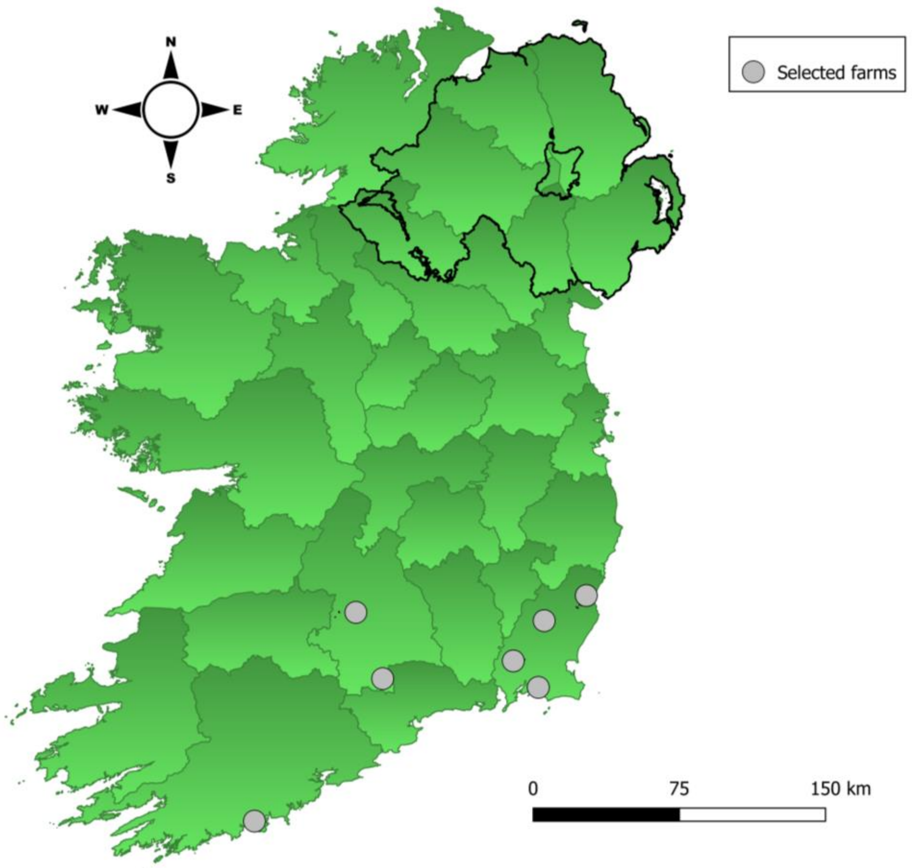

Figure 3-1. Map of Ireland showing the distribution of selected farms across the south and south east. Note: More than one farm represented by each dot, as they were located close together.

Farm characteristics for the duration of the study period are summarised in Table 3-1. Average TUAA (Total Utilisable Agricultural Area) was 53.1 ha (includes total grassland and crop area), which was slightly below the national average of 56.0 ha for dairy enterprises in Ireland (NFS, 2017). Average stocking rate was 2.9 LU ha^{-1} (1 livestock unit is equivalent to one adult cow producing 3,000 kg milk annually, (Eurostat, 2013), which was higher than the national average of the previous three years: 1.75 LU ha^{-1} in 2012 to 2014 (Butler et al., 2017). Average milk solid production (fat plus protein), was 806 kg ha^{-1} which was above the national average (649 kg ha^{-1}) for the 2012 to 2014 period. Concentrate feed imported onto the farm was, on average, lower on these farms (847 kg cow ^{1}) compared to the national average (926 kg cow ^{1}) (Butler et al., 2017). Ten out of the fifteen farms in this study were on average above the 170 kg organic N ha^{-1} restriction on stocking rate (SR), and were therefore, in nitrates derogation conditions. Some farms crossed over the derogation threshold during the study period. Ten farms were on well drained soil (Sandy loam), three farms on moderately drained soil (Clay loam), and two farms on poorly drained soil (Clay). Meteorological data (rainfall and air temperature) was gathered from meteorological stations (Met Éireann, 2019) in the region. Where registered rainfall gauges were available on farms, the data were used to represent rainfall for that local area. The farmers were first approached in Winter 2014, and data collection continued for three growing seasons; 2015, 2016 and 2017.

Table 3-1. Mean values and standard deviation for total grassland area, crop area, rainfall, air temperature, total utilisable agricultural area stocking rate, milk solids, and concentrate feed per cow and soil texture class for 15 Irish dairy farms between 2015 and 2017

Farm ID	**TGA (ha)**	**Crop (ha)**	**Rainfall (mm yr $^{-1}$)**	**Air temp. (°C)**	**Soil texture Class**	**TUAA SR (LU ha^{-1})**	**TUAA SR (kg Org N ha^{-1})**	**MS (kg ha^{-1})**	**Conc. (kg LU $^{-1}$)**
1	63.8	0.0	1,007	10.2	Sandy loam	2.5	209	813	674
2	35.0	0.0	1,208	10.1	Sandy loam	2.5	211	687	757
3	96.6	0.0	1,032	10.1	Sandy loam	2.2	191	631	298
4	44.7	2.3	1,032	10.1	Clay loam	2.6	225	1,185	866
5	58.3	0.0	967	9.7	Clay	1.7	146	624	1,106
6	34.9	1.8	1,007	10.2	Sandy loam	2.7	233	821	753
7	38.0	0.0	1,032	10.1	Sandy loam	2.0	170	797	938
8	105.3	8.4	1,208	10.1	Sandy loam	2.9	247	1,041	850
9	103.0	4.6	996	10.3	Sandy loam	2.8	238	1,128	1,128
10	83.0	0.0	967	9.7	Clay loam	2.1	177	734	1,027
11	130.1	18.2	1,007	10.2	Sandy loam	2.0	174	607	236
12	60.0	0.0	967	9.7	Clay	2.0	166	534	1,315
13	32.0	0.0	1,208	10.1	Sandy loam	2.6	219	1,089	867
14	73.3	40.4	992	9.8	Sandy loam	1.3	109	915	1,124
15	82.0	0.0	967	9.7	Clay loam	1.8	149	487	733
Mean	69.3	5.0	1,040	10		2.2	191	806	845
S.D.	30.3	11.0	90	0.2		0.5	39	222	295

TGA, Total Grassland Area; Temp, temperature; TUAA, Total Utilisable Agricultural Area; Sandy loam, Well drained soil; Clay loam; Moderately drained soil, Clay; Poorly drained soil; SR, Stocking Rate; LU, Livestock Unit; Org, organic; MS, Milk Solids; Conc., Concentrate feed; S.D., Standard Deviation.

3.3.2 Farm and field scale nitrogen balance and use efficiency

At farm scale or "farm gate", a nitrogen balance (NBAL) and use efficiency (NUE) methodology (Eurostat, 2013) was used to account for the main N flows at the farm scale that are under the direct control of the farmer. A soil surface methodology (OECD, 2007), was used to capture the main N flows for field scale balance and NUE. Details of imports and exports for both scales are explained in Table 3-2. Nitrogen balance (kg N ha^{-1}), was calculated as:

$$Nitrogen\ Balance\ (kg\ ha) = \frac{(Nitrogen\ imports - \ exports\ (kg))}{Spatial\ area\ (ha)} \qquad \textbf{Equation 3-1.}$$

$$Nitrogen\ Use\ Efficieny\ (\%) = \frac{Nitrogen\ exports\ (kg)}{Nitrogen\ imports\ (kg)} X\ \frac{100}{1} \qquad \textbf{Equation 3-2.}$$

Calculations represent pastoral/grassland zones only (Total Farm Area subtract cereal crop area). Crop area N flow data were excluded to keep the results relevant to the pastoral N cycle. Farm scale was represented by the nutrient balance calculations carried out on the Total Grassland Area (TGA) of the farm. Field scale was represented by N balance calculations carried out on individual fields/paddocks. All data were annual (calendar year) totals.

Table 3-2. Import variables (inorganic fertiliser, concentrate feed and organic manure) and export variables (milk sold, animals sold and silage) used to calculate nitrogen balance and use efficiency for farm and field (Dairy, Non-Dairy and Silage) scale

Import (kg N ha^{-1})		**Farm**	**Export (kg N ha^{-1})**	
Inorganic fertiliser	(+)		Milk sold	(-)
Concentrate feed	(+)		Animals sold	(-)
		Field		
Inorganic fertiliser	(+)		Milk sold	(-)
Organic manure	(+)	Dairy Paddocks	Animals sold	(-)
Concentrate feed	(+)		Silage	(-)
Inorganic fertiliser	(+)			
Organic manure	(+)	Non-Dairy Fields	Animals sold	(-)
			Silage	(-)
Inorganic fertiliser	(+)			
Organic manure	(+)	Silage Fields		
			Silage	(-)

In Table 3-2, inorganic fertiliser represents synthetic fertiliser (manufactured to contain specific minerals and chemicals) that is purchased and imported onto farms through the farm gate at farm scale or onto individual fields at field scale. Data on inorganic fertiliser was recorded by participants (purchase receipts) throughout each year. Nutrient values for synthetic fertiliser was considered to be 100% available to the plant. Concentrate feed represents the nutrient import value calculated from its protein content and total tonnage purchased which were recorded by participants each year. Nutrient import via concentrate feed was only counted for 36 weeks of each year to prevent double counting of nutrients in organic manure collected during the animal housing period. Nutrients imported by feeding animals concentrate feed during the NAP 16-week housing period (S.I. No. 31 of 2014, EU, 2014) were counted in the typical values used for nutrient content of organic manures. Nutrient availability values were tailored according to application rate and season (based on date) in accordance Wall and Plunkett, 2016. Organic manure application data was recorded by participants throughout each year. Organic manure collected during the housing period was imported at field scale mechanically throughout the growing season. Organic manure represents animal excreta collected during the housing period only and excludes animal excreta deposited in-situ during the outdoor grazing season. The assumption was made that nutrients in excreta deposited during the grazing season were in constant cycle throughout the farm system. Milk sold represents the nutrients exported off farm via milk collection by processors. Data on milk sold was collected by the farmer annually through the Irish Cattle Breeder Federation database (ICBF) (ICBF, 2019). Nutrient value was based on total quantity sold and tailored by the farm's average protein level. Animals sold represents the nutrients exported in the sale or movement of animal's permanently off farm. Nutrient value was based on typical values used by McDonald, 2011. Nutrient values were further tailored by the use of animal age and weight (kg) in each individual case, which was collected by the farmer each calendar year through the ICBF database. Silage represents the nutrients exported via silage harvest events in a given year. Silage harvest events are not specific to fields in the silage subzone, and so silage harvests represent the action which may occur on any subzone. Nutrient value was based on the total silage yield and herbage nutrient concentration of an individual paddock or field. Silage yield data was recorded by the farmer on a weekly base and collated at the end of the calendar year. Herbage nutrient concentration data was based on the average of three herbage samples taken from each paddock in spring, summer, and autumn on each farm in 2015 and 2016.

Field scale nutrient import values via concentrate feed, and export via milk sold and animals sold were proportionally weighted by each paddocks number of grazing defoliations and spatial area (ha). Grazing defoliations were a proxy for the number of visits each paddock experienced annually. Where a paddock had high or low defoliations and/or a large or small spatial area, nutrient load and nutrient export were weighted towards those differences. This weighting procedure does not apply to inorganic fertiliser, organic manure or silage harvest events.

3.3.3 Subzone characterisation

The TGA was divided into multiple zones for each farm. Total Grassland Area averaged 69.3 ha over the three years (Table 3-1). This grassland area was subdivided into Dairy, Non-Dairy, and Silage subzones, within which, the individual fields and paddocks were monitored.

Paddocks were the smallest individual unit with uniform management on these farms. Paddocks were most commonly used in the Dairy subzone, but sometimes also in the Non-Dairy subzone. Paddocks were not used in the Silage subzone. Paddocks are smaller sub divisions of fields, and are typically rotationally grazed through the grass growing season. This small paddock size allows for greater control of grazing by the farmer. On dairy farms, these paddocks are typically within walking distance for the dairy cows of the milking parlour, via a network of farm lanes, and this area of the farm is referred to as the “milking platform”.

ArcGIS® software (ArcGIS, 2016), was used to map individual paddock, field and farm boundaries. The boundary of a field is commonly a hedgerow, while the boundary of a paddock is commonly a single strand electric fence. Bing maps aerial photography was used as a basemap (Bing™ Maps, 2005). Paddocks and field boundaries were traced digitally, and later a ground truth was carried out. A number of farms were already digitally mapped to paddock scale commercially by Grasstec (GrasstecGroup). These maps were then imported into ArcGIS. Paddock boundaries were updated where changes to paddock or field layout had occurred during the study. Geometric areas were calculated in hectares by ArcGIS. An example of farm spatial layout is shown in Figure 3-2.

The three subzones were characterised as follows:

- **Dairy subzone:** Fields or paddocks carrying dairy cows fully throughout each growing season. Dairy paddocks were the most numerous individual management unit of the three grassland subzones, and were commonly closest in distance, to the farm infrastructure (milking parlour and winter housing). On average, there were 27 Dairy paddocks per farm, with an average area of 1.4 ha. Dairy subzone fields made up, on average, 57% of the TGA of these participating farms.
- **Non-Dairy subzone:** Fields and paddocks that carried Non-Dairy livestock. These fields were used for pasture grazing, but for the purpose of calf rearing, or a beef enterprise. There were, on average, 7 Non-Dairy fields on each farm, with an average area of 3 ha. Non-Dairy subzone fields made up 28% of the TGA, on average.
- **Silage subzone:** Grassland fields that did not carry dairy, or other livestock fully throughout a year and, instead, were used to produce grass for conserved winter forage. Silage was normally harvested from these fields twice annually, and, occasionally, animals grazed for a short time after harvest. Typically, these fields were larger in size, but less abundant. On average, there were 3.6 grass silage fields per farm, at 2.6 ha in area. Silage subzone fields made up 15% of the TGA, on average.

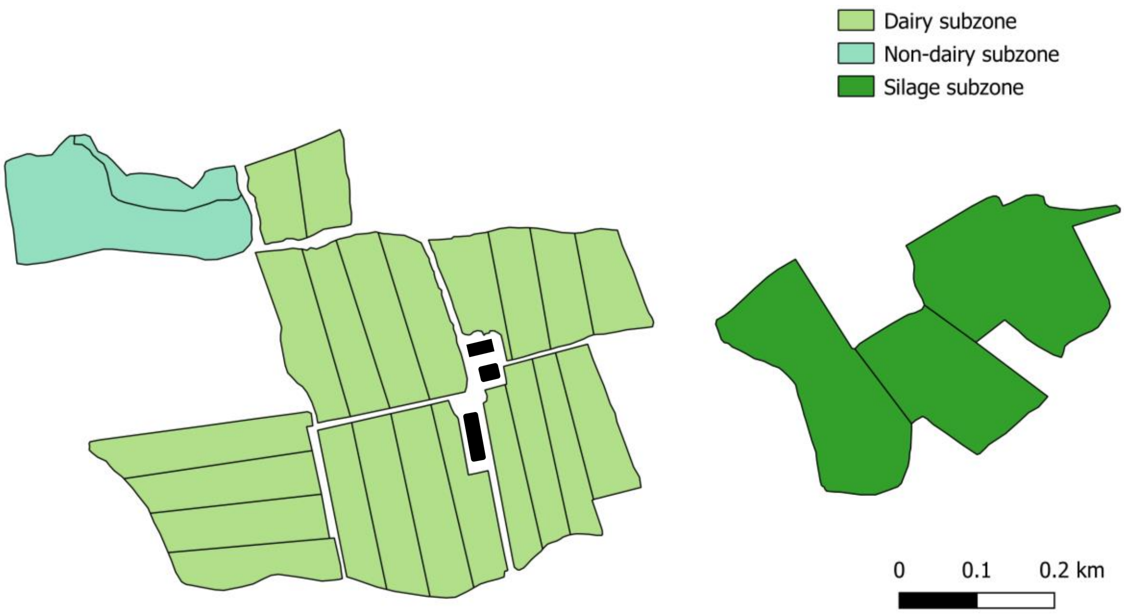

Figure 3-2. Example of a typical grazed grass dairy farm spatial layout. Field and paddock boundaries are indicated by lines. Subzone (Dairy, Non-dairy and Silage) boundaries indicated by colour difference. Infrastructure (milking parlour and winter housing) is located in the centre of the farm. Drawn with ArcGIS, (2016).

3.3.4 Data acquisition

Over the three years of the study, changes in farm size occurred on 8 of the 15 farms. This was due to temporary or permanent increases or decreases in area of land that was rented or purchased. This predominantly occurred on Non-Dairy and Silage subzone area and changes in Dairy subzone fields were fewer. As a consequence, some areas or fields in these subzones were not included in the final analysis as they didn't have three years of consecutive data or lacked data altogether. For the analysis, 100% of Dairy fields were represented and 44% and 48% of Non-Dairy and Silage fields respectively were represented on average. The total project area (area with consistent monitoring data), was slightly smaller than the actual TGA, due to these changes in overall farm size, but the fields within that total project area were monitored in full throughout the three years. To accommodate these 8 farms, farm scale records and data that were representative of the total project area, were used to represent the TGA. The remaining 7 farms had sufficient data for consistent monitoring at both scales.

3.3.4.1 Farm scale

Farm scale Nitrogen imports of total inorganic N fertiliser, and concentrate feed, were recorded by the farmer. Data on annual total milk, and animals sold, was collected with permission, from an online farm database; Irish Cattle Breeding Federation (ICBF, 2019). Nitrogen concentration of feed, milk and meat from animals sold, were based on typical values, used by McDonald, (2011). Wet and dry N deposition were not included in the balance, as it generally contributes a small and variable proportion of N (2 to 22 kg N ha^{-1} yr^{-1}; Doyle et al., 2014). Biological fixation was excluded as it varies greatly with grassland management practices (25 to 142 kg N ha^{-1} yr^{-1}; (Munters, (1997), Watson et al., (2002)). Losses via gaseous emissions, immobilisation, leaching and erosion were not measured nor included in the balance approach used here.

3.3.4.2 Field scale

Field scale imports included N from inorganic and organic fertilisers, and concentrate feed. Fertiliser imports were based on individual field records made by the farmer. Total concentrate feed purchased, was apportioned across the farm as were total exports via total milk and animals sold using an apportionment procedure.

On the Dairy subzone, information on pasture production (grazing and silage yield, no. of defoliations and harvests) was recorded at field and paddock scale by the farmer on a weekly basis by cut and weigh, plate meter, and eye-ball estimates, depending on farmer preference. Grazing yield is the total herbage consumed by livestock and silage yield is the total herbage removed by machinery. This information included the number of defoliations due to grazing and/or harvests, as well as the herbage dry matter (DM) accumulation (kg DM ha^{-1}) per field. Only a portion of Non-Dairy and Silage fields had pasture production data recorded (average 44% and 48% per farm respectively) and so comparisons were limited to fields with sufficient data The Dairy subzone was 100% represented. Annual values for Farm scale total N imported, via concentrate feed, and exported via milk and animals sold, were then apportioned to each paddock, relative to the total number of fields with grazing defoliations at the end of each calendar year. This was carried out because Aarons et al.; Topp et al. (2015; 2007, respectively) showed that a balance calculated across grazing rotations (i.e. number of defoliations) reveals variation due to grassland management practices. For this study, this approach was adopted to capture the variation in N cycling data at the paddock scale. Note also, for N flow via animals sold from Dairy paddocks, an assumption was made that, annually, a dairy cow had a 50 kg calf that was removed from the subzone and was replaced by a maiden heifer, leaving a final balance of 225 kg imported. This calculation was based on a typical annual replacement rate of 20% of the dairy herd, with a maiden heifer target weight of 275 to 330 kg (Kennedy, 2014). It could be expected that variation in this value would be truer to the operations of commercial farms but the data required to be this accurate was not available.

Non-Dairy fields had different animal types (young cattle or sucklers) and lower stocking rates than the Dairy subzone. The total N export via animals sold off these fields, was calculated based on animal type and age. Number of animals sold each year was collected online (ICBF, 2019).

For Silage fields, in practice, some farmers carry out grazing after harvest events but this is commonly for a very short time and at very low stocking rates (Hyland et al., 2018). Where animals did graze on silage fields, the N in animals sold was captured in the Non-Dairy fields balance calculation, as values overall were small.

To calculate N export via silage harvest events, three herbage samples were taken and pooled from each monitored field to determine herbage N concentration. Samples were taken in spring, summer and autumn, and a mean annual N value was calculated. Herbage N value (g/kg DM), was chemically determined by colorimetric analysis, following hot acid digestion using sulphuric acid, as outlined by Byrne, (1979). Export of N via silage, was then calculated, based on mean annual herbage N value, and total annual herbage silage yield.

3.3.5 Farm scale factors

At the farm scale, a number of factors that were expected to impact the N cycle were assessed: soil drainage class and farm stocking rate class.

3.3.5.1 Farm soil drainage class

The national soil map (1:250,000), developed by the Irish Soils Information Systems (Teagasc, 2019), was used to determine the farm level drainage class. At each farm location, the soil association was identified firstly. The predominant soil type, and its drainage characteristics, were then identified using the soil map and catalogue system.

3.3.5.2 Stocking rate

Stocking rate as a factor was based on kg Org N ha^{-1} instead of LU ha^{-1}, and used for regression analysis. Animal organic stocking rates are based on annual nutrient excretion rates for livestock, as specified in the NAP regulations (S.I. No. 605 of 2017 and S.I. no 65 of 2018, EU, 2017 and 2018).

3.3.6 Field environmental factor classification

At the field scale, two environmental factors that might be expected to impact the N cycle were assessed: gradient and field soil drainage class (Whitehead, 1995).

3.3.6.1 Gradient class

A soil survey was carried out in 2016, by mapping the topography of each farm and individual field. The survey delineated areas of the farm that were uniform in topographic properties and were classified as sloping or level (see section 2.9). Level was defined as < 10 degrees of slope, and Gradient as >10 degrees of slope (Roberts and Watkins, 2014).

3.3.6.2 Field soil drainage class

During the survey, areas of uniform soil drainage were also delineated (See methods section). A soil auger was used in each area of uniform drainage to categorise the soil drainage type [see Appendix 1A: Figure 9-2]. A depth of 1 m was sampled where soil depth allowed. Each auger sample was laid out sequentially on a tray. A scale was placed beside the tray and the soil profile was photographed. Soil texture field test assessment was carried out according to (FAO, 2006) at each auger location. Some farms had data on soil drainage type available (Fealy et al., 2010) and these data were used as a cross reference. Each field was then assigned a drainage class based on the survey, auger and other available information. Fields were classified as well drained (Sandy loam), moderately drained (Clay loam) and poorly drained (Clay) (CTECO, 2010; Simo et al., 2007).

3.3.7 Field management factor classification

Four field management factors that might be expected to affect N cycling were assessed; soil fertility (Whitehead, 1995), sward species composition (Bryant et al., 2017), cropping rate (Carey et al., 2017), and reseeding interval (Necpálová et al., 2013).

3.3.7.1 Soil fertility

Each field and paddock were soil sampled for nutrient analysis in Winter (December and January) 2014, 2015, 2016 and 2017. For every management unit, 20 soil cores to a depth of 100 mm in a W shaped pattern were taken and bulked together. The sample was then placed in a soil sample box and labelled according to the Field ID. Samples were dried at 40 °C and sieved through a 2mm sieve. Morgan's extract (Peech and English, 1944) was used to measure the soil test plant available Phosphorus (STP) and soil test Potassium (STK) levels using the method described by Byrne (1979). Soil analysis was quality checked by including one blank and ten control per run of 89 soil samples. Soil pH was measured in a 1:2 soil to water ratio and determined using a Mettler Toledo glass calomel electrode. Soil nitrogen levels were not tested as there is no widespread reliable method to determine N release rate from soil (Schepers and Raun, 2008). The Soil Index system for STP and STK in Ireland, which describes nutrient status and associated fertiliser recommendations (Wall and Plunkett, 2016), was applied to each soil test result, so that each field was assigned one of four Index categories (Table 3-3). There was no Soil Index system available to apply to pH test results, so four broad Index categories were developed based on agronomic optimum for pH with pasture on mineral soils (pH 6.5) (Wall and Plunkett, 2016).

Table 3-3. Soil Index system and associated soil status for plant available phosphorous (STP), plant available potassium (STK) (both using Morgan's extractant), and soil pH indices and ranges used for this study). Adapted from Wall and Plunkett, (2016)

Soil Index	Soil status	STP (mg l^{-1})	STK (mg l^{-1})	pH
Index 1	Very Low	0.0 to 3.0	0 to 50	< 5.5
Index 2	Low	3.1 to 5.0	51 to 100	5.6 to 6.0
Index 3	Medium/Adequate	5.0 to 8.0	101 to 150	6.1 to 6.5
Index 4	Sufficient/High	> 8.0	> 150	> 6.5

3.3.7.2 Sward species composition

A herbage survey was carried out in 2017 on each field to estimate the proportions of herbage species that make up the swards full composition. A 0.5 m X 0.5 m quadrat was systematically placed three times in a diagonal line moving down the long axis of each field, following Sheridan et al. (2008). The plant species inside the quadrat were identified using Feehan et al 2012, and Fossit (2000). A percentage proportion was recorded for each plant by visual estimation, using the DAFOR scale (Kent et al., 2001). For this study, only the proportions of Perennial Ryegrass *(Lolium Perenne)* and White clover (*Trifolium Repens)* species were included in the statistical analysis. Perennial Ryegrass (PRG) was grouped into three different proportions: 0 to 75% PRG (Low), 76 to 90% PRG (Medium), and 91 to 100% PRG (High). White clover (WC) was grouped into two levels: WC absent and WC present.

3.3.7.3 Cropping rate

Each field was classified into one of three cropping classes; Grazing Only (GO), Grazing + Harvest (G+H), and 2 Cut Harvest (2CH). Grazing events will remove N in grazed grass, but will also add N, in grazing animal excreta. Harvest events will remove N in harvested grass, but there will be no N returned in grazing animal excreta. Typically, farmers designate a field to a particular cropping type, but in-season management decisions, may lead to grazing or harvest events that were not predetermined. **Grazing only:** Rotational grazing systems require the movement of the dairy herd through all available Dairy paddocks (milking platform) to allow cows access to fresh pasture every 12 hours (Kavanagh, 2016b). During peak grass growth the herd might be expected to rotate around the Dairy paddocks on a 21 day cycle (Finch et al., 1994). The annual mean number of defoliations is 7 for the south east region of Ireland (O'Leary et al., 2013). **Grazing + Harvest:** During peak grass growth in Summer, pasture yield can exceed the feed demand of the dairy herd. The farmer may then intervene by cropping or harvesting fields with high grass cover (e.g. grass dry matter at >1,600 kg DM ha^{-1}) i.e. beyond the leafy vegetative stage where nutritive value is at its highest (Kilcher, 1981). These fields would not only have been grazed but would have also experienced a pasture harvesting event and so is classified separately as Grazing + Harvest. Grass harvesting events may occur one or more times on any of the Dairy paddocks throughout the year and is a common grassland management practice (Finch et al., 1994). Non-Dairy fields can be managed for Grazing only or Grazing + Harvest. They are excluded from the rotational grazing by the dairy herd. Other animal types may graze short term. **2 Cut Harvest:** Grass Silage fields differ to Dairy paddocks, as they are set aside for at least two harvesting events per year. Silage fields are excluded from rotational grazing by the dairy herd to allow the silage crop to grow.

3.3.7.4 Reseeding interval

Participants were asked to identify what year each paddock was reseeded. Where an exact date was not possible to record, the date was recorded within a range. Fields were grouped into three categories based on the number of years since a reseeding event occurred; < 5 years, 5 to 10 years, and > 10 years.

3.3.8 Statistical analysis

Linear regression was used for farm scale and stocking rate analysis. To investigate differences in N cycling between scales, results of farm and field N balances and NUE were compared using Analysis of Variance (ANOVA). Analysis of Variance was also used to test for effects of field environmental factors (gradient and soil drainage), and management factors (soil fertility, sward species composition, cropping rate, and reseeding interval). All data were screened for outliers and data entry errors. Measures of centrality and variation were determined in SPSS (IBM SPSS, 2016). Tests for normal distribution were carried out with Shapiro Wilk (farm scale dataset) and also Kolmogorov Smirnov where n > 100 (field scale dataset). For normally distributed data, univariate ANOVA was carried out to determine significant differences. The Bonferroni post hoc test was used to distinguish significant differences (pairwise comparison). For data that was not normal, non-parametric independent tests were carried out to determine significance. Kruskal Wallis and Mann Whitney U tests were used. The following equations were used to investigate the central hypotheses:

Farm scale: $\hat{Y}1$ & $\hat{Y}2 = \beta o + \beta 1X1 + \beta 1X2 + \beta 1X3 + \beta 1X4$ **Equation 3-3.**

where $\hat{Y}1$ = Farm scale N balance, $\hat{Y}2$ = Farm scale N use efficiency,
X1 = Year, X2 = Farm, X3 = Farm soil Drainage class, X4 = Stocking Rate

Field scale: $\hat{Y}3$ & $\hat{Y}4 = \beta o + \beta 1X1 + \beta 1X2 + \beta 1X3 + \beta 1X4$ **Equation 3-4.**

where $\hat{Y}3$ = Field scale N balance, $\hat{Y}4$ =Field scale N use efficiency,
X1 = Year, X2 = Subzone, X3 = Environmental factors, X4 = Management factors

3.4 Results

3.4.1 Farm scale N cycle

Farm scale N flows are shown in Table 3-4. Nitrogen imports via inorganic fertiliser dominated at farm scale, with a mean import of 183 kg N ha^{-1} (SD 51 kg N ha^{-1}), which was 81% of imports. Inorganic N fertiliser imports were higher in 2017 (190 kg N ha^{-1}), than in 2015 and 2016 (179 and 180 kg N ha^{-1}, respectively). Mean N import via concentrate feed was 41 kg N ha^{-1} (SD 18 kg N ha^{-1}), which was 19% of imports. Imports of N in feed were higher in 2016 and 2017 (43 and 41 kg N ha^{-1}, respectively) than in 2015 (38 kg N ha^{-1}).

N in milk dominated farm scale N exports (mean 58 kg N ha^{-1}, SD 17 kg N ha^{-1}), and made up 98% of total exports. Export of N in milk was higher in 2016 and 2017 (57 and 61 kg N ha^{-1}, respectively) than in 2015 (54 kg N ha^{-1}). Mean export via animals was 1 kg N ha^{-1} SD 2 kg N ha^{-1}), 2% of total exports. Nitrogen export in animals was static in 2015 and 2016 (1 kg N ha^{-1}) and slightly lower in 2017 (1 kg N ha^{-1}).

Mean farm scale NBAL was 165 kg N ha^{-1} (CV 34%). However, farm scale NBAL showed a large range, from 20 to 246 kg N ha^{-1}, indicating large differences in N management, and N source pressures between farms. Mean N surplus was relatively stable year to year (161 – 169 kg N ha^{-1}). Mean NUE was 28% and was also relatively stable year to year (Table 3-4). However, a large range in NUE (10 – 66%) again indicates large differences in N management between farms.

3.4.2 Field scale N cycle

Field scale N flows are shown in Table 3-5. A total of 446 fields had sufficient data for full analysis. At field scale, N import was also dominated by inorganic fertiliser, at 238 kg N ha^{-1}, on average (SD 84 kg N ha^{-1}), representing 77% of imports. As at the farm level, import of inorganic fertiliser N was higher in 2017 (259 kg N ha^{-1}) than in 2015 and 2016 (227 and 228 kg N ha^{-1}). Mean N import via organic manure was 13 kg N ha^{-1} (SD 15 kg N ha^{-1}) and varied little year to year, making up 19% of imports. Mean N import in concentrate feed was 57 kg N ha^{-1} (SD 51 kg N ha^{-1}) and was lowest in 2015 (53 kg N ha^{-1}), increasing slightly in 2016 and 2017 (58 and 60 kg N ha^{-1}, respectively), making up 4% of imports.

As at the farm scale, N export at the field scale was dominated by milk (120 kg N ha^{-1}, SD 99 kg N ha^{-1}), and was 57% of the export total. Mean field scale N export in milk increased over the three years of the study, from 111 in 2015 to 132 kg N ha^{-1} in 2017. Mean N export in animals was 9 kg N ha^{-1} (SD 7 kg N ha^{-1}), increasing only slightly over the study period, from 9 kg N ha^{-1} in 2015 to 10 kg N ha^{-1} in 2017, and was 4% of the total. Mean field scale N export in silage harvest was 38 kg N ha^{-1} (SD 68 kg N ha^{-1}), increasing over the study period from 32 kg N ha^{-1} in 2015 to 47 kg N ha^{-1} in 2017, and was 39% of total exports.

Mean field scale NBAL was 139 kg N ha^{-1}, and showed a high degree of variability between fields (CV 77%). This indicates large differences in N management, and resultant N source pressures, between fields. NBAL ranged from 253 to 425 kg N ha^{-1}. Mean NBAL was slightly higher in 2017 (143 kg N ha^{-1}) than in 2016 (136 kg N ha^{-1}) and 2015 (139 kg N ha^{-1}). Mean field scale NUE was 57% and was also highly variable between fields (CV 38%), ranging from 0% to 393%. Again, this indicates large differences in N management between fields. Mean field scale NUE was almost static for the three years of this study (56, 57 and 57%).

3.4.3 Difference in N sustainability between scales

NBAL was not significantly different between farm and field scale ($p > 0.05$) (Figure 3-3). However, NBAL at field scale (CV 77%) showed markedly greater variability than at farm scale (CV 34%). In contrast, mean NUE was significantly higher ($p < 0.05$) at field scale (57%) than at farm scale (28%) (Figure 3-3). As with NBAL, NUE showed markedly greater variability at field scale (CV 67%) than at farm scale (CV 39%), This indicates much greater variability in N management and resultant N source pressures at field scale than at farm scale.

Table 3-4. Mean, standard deviation, minimum, maximum and coefficient of variation for nitrogen imports (inorganic fertiliser, organic manure, and concentrate feed), exports (milk sold, animals sold, and silage), balance, and use efficiency for farm (*n* = 15) and field (*n* = 446) scale between 2015 and 2017

Scale	Variable	2015	2016	2017	Mean	SD	Min	Max	CV
					Imports (kg N ha^{-1})				(%)
Farm (*n* = 15)	Inorganic fert.	179	180	190	183	51	49	266	28
	Conc. feed	38	43	41	41	18	4	79	44
					Exports (kg N ha $^{-1}$)				
	Milk sold	54	57	61	58	17	27	91	30
	Animals sold	1	1	1	1	2	0	7	141
	NBAL	161	164	169	165	56	20	246	34
	NUE	28%	28%	28%	28%	11%	10%	66%	39
					Imports (kg N ha^{-1})				
Field (*n* = 446)	Inorganic fert.	227	228	259	238	84	21	489	35
	Organic manure	13	13	13	13	15	0	83	113
	Conc. feed	53	58	60	57	51	0	439	90
					Exports (kg N ha^{-1})				
	Milk sold	111	118	132	120	99	0	733	82
	Animals sold	9	9	10	9	7	0	71	80
	Silage	33	35	47	38	68	0	415	240
	NBAL	139	136	143	139	107	253	425	77
	NUE	56%	57%	57%	57%	38%	0%	393%	67

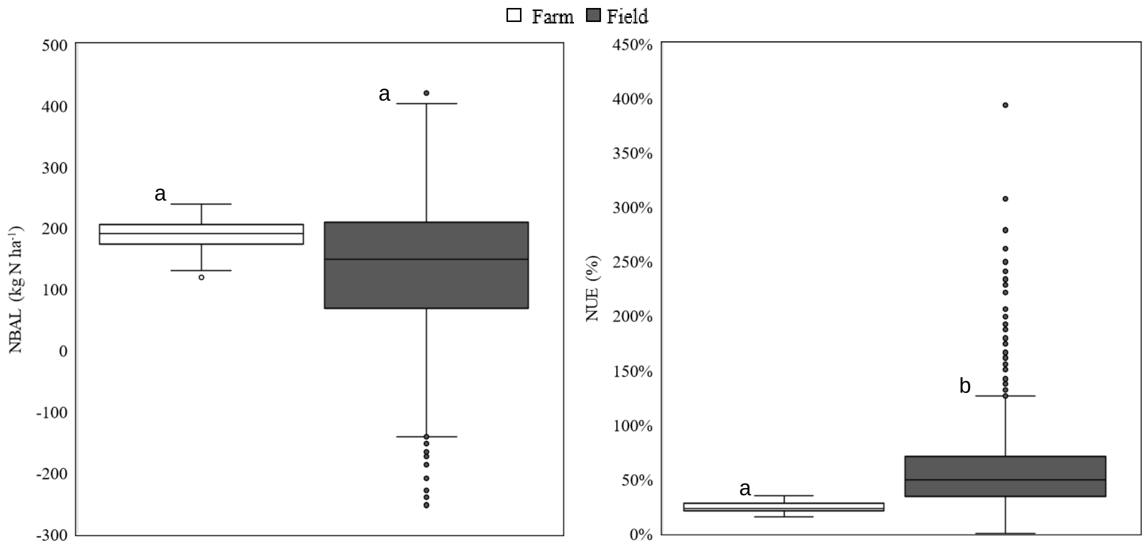

Figure 3-3. Mean nitrogen balance and use efficiency at farm (*n* = 15) and field scale (*n* = 446) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post-hoc test).

3.4.4 Farm scale factors affecting N balance and use efficiency

Mean farm scale NBAL and NUE were not significantly impacted by year ($p > 0.05$). Mean farm scale NBAL was significantly different between farms ($p < 0.05$), varying from 145 to 227 kg N ha^{-1}, as was mean NUE, varying from 16 to 32% ($p < 0.05$). Mean NBAL was not significantly impacted by farm level drainage class differences ($p > 0.05$). However, NUE was significantly higher for well and moderately drained farms (31 and 25%, respectively), than poorly drained farms (19%). Stocking rate increased slightly each year (186, 190, and 196 kg Org N ha^{-1}) but the regression model for NBAL and stocking rate (kg N ha^{-1}) was not significant: $F(1,13) = 0.6$, $p > 0.05$, $r^2 = 0.05$. The regression model for NUE and stocking was significant however: $F(1,13) = 5.9$, $p < 0.05$, $r^2 = 0.31$ (Figure 3-4).

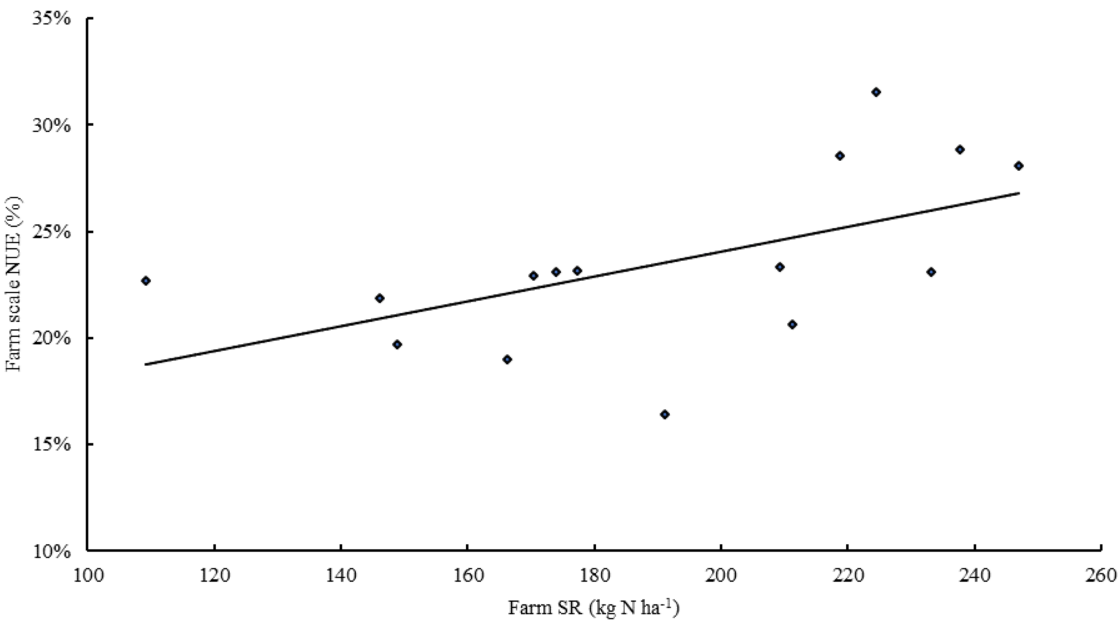

Figure 3-4. Regression analysis for farm scale NUE and stocking rate, fitted to the line y = 0.0006x + 0.1236, $r^2 = 0.31$.

3.4.5 Differences between subzones

Non-Dairy NBAL and NUE were not significantly different to the Dairy subzone (*$p > 0.05$*). Silage subzone mean NBAL was significantly lower, and NUE was significantly higher, than the other two subzones (*$p < 0.05$*) (Figure 3-5). The spatial distribution of mean NBAL and NUE at field scale for one farm is shown in Figure 3-6.

Table 3-5. Mean, standard deviation, minimum, maximum and coefficient of variation for nitrogen imports (inorganic fertiliser, organic manure, and concentrate feed), exports (milk sold, animals sold, and silage), balance, use efficiency, and herbage accumulation at field scale for Dairy paddocks (*n* = 399), Non-Dairy (*n* = 29), and Silage (*n* = 18) fields between 2015 and 2017

Subzone	Variable	2015	2016	2017	Mean	SD	Min	Max	CV
Dairy paddocks (*n* = 399)	Imports (kg N ha^{-1})								(%)
	Inorganic fert.	226	228	263	239	85	21	489	35
	Organic manure	13	13	13	13	15	0	83	116
	Conc. feed	58	65	66	63	50	0	439	80
	Exports (kg N ha^{-1})								
	Milk sold	122	131	147	133	95	0	733	71
	Animals sold	9	10	11	10	7	0	52	70
	Silage	21	25	37	28	51	0	368	185
	NBAL	144	140	148	144	106	253	399	74
	NUE (%)	56%	57%	57%	57%	37%	3%	393%	65
	Herbage accumulation (kg DM ha^{-1})								
	Grazing yield.	10,669	10,297	10,628	10,529	3,839	1,000	22,984	36
	No. Defoliations	7	6	7	7	2.4	1	13	34
	Silage yield	2,553	2,409	3,488	2,852	3,841	400	10,950	135
	No. Harvests	1	1	1	1	2	1	4	240
Non-Dairy Fields (*n* = 29)	Imports (kg N ha^{-1})								
	Inorganic fert.	211	212	208	210	55	75	434	26
	Organic manure	8	12	10	10	11	0	39	109
	Exports (kg N ha^{-1})								
	Animals sold	2	4	3	3	9	0	71	272
	Silage	78	74	89	80	89	0	365	111
	NBAL	151	158	141	150	118	62.8	425	79
	NUE (%)	33%	35%	41%	37%	4%	0%	150%	108
	Herbage accumulation (kg DM ha^{-1})								
	Grazing yield.	8,752	7,380	8,581	8,170	3,011	1,496	15,570	37
	No. Defoliations	5	5	6	5	3	0	13	50
	Silage yield	3,741	3,092	3,448	3,387	2,438	95	9,500	72
	No. Harvests	5	1	1	1	1	1	3	60
Silage Fields (*n* = 18)	Imports (kg N ha^{-1})								
	Inorganic fert.	272	257	226	251	87	86	480	35
	Organic manure	23	27	22	24	16	0	70	67
	Exports (kg N ha^{-1})								
	Silage	238	219	209	222	81	84	415	37
	NBAL	22	33	25	27	45	41	157	167
	NUE	93%	80%	85%	86%	3%	42%	179%	30
	Herbage accumulation (kg DM ha^{-1})								
	Grazing yield	5,091	5,056	6,189	5,405	2,431	1,088	10,525	45
	No. Defoliations	3	3	4	4	1.6	1	7	40
	Silage yield	7,156	6,942	6,579	6,892	2,593	2,500	13,200	38
	No. Harvests	2	2	1	2	0.6	1	4	30

3.4.5.1 Dairy subzone

On dairy paddocks, mean N import, in the form of inorganic fertiliser N was 239 kg N ha^{-1} (SD 85 kg N ha^{-1}) (Table 3-5). This import was highest in 2017 (263 kg N ha^{-1}), compared to 2015 and 2016 (226 and 228 kg N ha^{-1}, respectively). Mean N import via organic manure was much lower at 13 kg N ha^{-1} (SD 15 kg N ha^{-1}) and did not vary greatly between years. Mean N imported in concentrate feed was 63 kg N ha^{-1} (SD 50 kg N ha^{-1}). The highest concentrate feed imports were in 2016 and 2017 (65 and 66 kg N ha^{-1}), and lowest in 2015 (58 kg N ha^{-1}).

The dominant N export from dairy paddocks was via milk (133 kg N ha^{-1}; SD 95 kg N ha^{-1}). Milk exports from the dairy paddocks (milking platform) increased incrementally over the study period, from 122, to 131, to 147 kg N ha^{-1}). Mean export via animals was 10 kg N ha^{-1} (SD 7 kg N ha^{-1}), varying little over the study period. Mean N export via harvested silage was 28 kg N ha^{-1}, and, as with milk N exports, was highest in 2017 (37 kg N ha^{-1}) compared to 2015 and 2016 (21 and 25 kg N ha^{-1}, respectively).

Mean NBAL for Dairy paddocks was 144 kg N ha^{-1} (CV 74%) (Table 3-5, Figure 3-5). NBAL was (144 kg N ha^{-1}) in 2015 and slightly higher in 2017 (148 kg N ha^{-1}) and lowest in 2016 (140 kg N ha^{-1}). Mean NUE for Dairy paddocks was 57% (CV 65%). Mean NUE was almost static for the duration of the study; 56% in 2015 and 57% in both 2016 and 2017.

On Dairy paddocks the mean grazed herbage yield was 10,529 kg DM ha^{-1} (SD 3,839 kg DM ha^{-1}) coming from, on average, 7 grazing's (SD 2). Grazed herbage yield was highest in 2015 and 2017 (10,669 and 10,628 kg DM ha^{-1}) and lowest in 2016 (10,297 kg DM ha^{-1}). Mean silage yield from Dairy paddocks was 2,852 kg DM ha^{-1} (SD 3,841 kg DM ha^{-1}) and the mean number of harvests was 1 (SD 2). Mean silage yield was highest in 2017 (3,488 kg DM ha^{-1}) and lowest in 2015 and 2016 (2,553 and 2,409 kg DM ha^{-1}, respectively).

3.4.5.2 Non-Dairy subzone

On Non-Dairy fields, mean N import via inorganic fertiliser was 210 kg N ha^{-1} (SD 55 kg N ha^{-1}). Inorganic fertiliser N imports were almost static for the three years (211, 212 and 208 kg N ha^{-1}). Mean N imported in organic manure was 10 kg N ha^{-1} (SD 11 kg N ha^{-1}), varying little over the study period (8, 12 and 10 kg N ha^{-1}).

Mean N export via animals from Non-Dairy fields was 3 kg N ha^{-1} (SD 9 kg N ha^{-1}). This result includes animals sold after grazing silage fields because the grazing was short term. Mean N export in harvested silage was 80 kg N ha^{-1} (SD 89 kg N ha^{-1}). The largest mean export was in 2017 (89 kg N ha^{-1}) while 2015 and 2016 were similar (78 and 74 kg N ha^{-1}, respectively).

Mean field scale NBAL for the Non-Dairy subzone was 150 kg N ha^{-1} (CV 79%). The lowest NBAL was in 2017 (141 kg N ha^{-1}) and the highest was in 2016 (158 kg N ha^{-1}). NBAL was

151 kg N ha^{-1} in 2015. Mean NUE was 37% for the Non-Dairy subzone (CV 108%) and NUE was highest in 2017 (41%) and lowest in 2015 and 2016 (33 and 35%, respectively).

Mean grazed herbage yield for Non-Dairy fields was 8,170 kg DM ha^{-1} (SD 3,011 kg DM ha^{-1}) coming from, on average, 5 grazing's (SD 3). Grazing yield was similar in 2015 and 2017 (8,752 and 8,581 kg DM ha^{-1}, respectively) and lowest in 2016 (7,380 kg DM ha^{-1}). Mean silage yield from Non-Dairy fields was 3,387 kg DM ha^{-1} (SD 2,438 kg DM ha^{-1}) and the mean number of harvests was 1 (SD 1). Mean silage yields were higher in 2015 and 2017 (3,741 and 3,448 kg DM ha^{-1}, respectively) than in 2016 (3,092 kg DM ha^{-1}).

3.4.5.3 Silage subzone

On silage fields, mean N import via inorganic fertiliser was 251 kg N ha^{-1} (SD 87 kg N ha^{-1}). Mean import was lowest in 2017 (226 kg N ha^{-1}), and higher in 2015 and 2016 (272 and 257 kg N ha^{-1}, respectively). Mean N import via organic manure was 24 kg N ha^{-1} (SD 16 kg N ha^{-1}), being higher in 2016 (27 kg N ha^{-1}) than in 2015 and 2017 (23 and 22 kg N ha^{-1}, respectively).

Mean N export via silage was 222 kg N ha^{-1} (SD 81 kg N ha^{-1}). The highest N exported in silage was in 2015 (238 kg N ha^{-1}), and 2016 and 2017 were incrementally lower (219 and 209 kg N ha^{-1}, respectively). Only short term or temporary grazing (low covers and less grazing's) occurred on silage fields, before animals were moved back to Non-Dairy fields, as a result, N exported via animals sold was captured on the Non-Dairy subzone rather than the silage subzone.

Mean NBAL for Silage fields was 27 kg N ha^{-1} (CV 167%). NBAL was highest in 2016 (33 kg N ha^{-1}) and lowest in 2015 and 2017 (22 and 25 kg N ha^{-1}). Mean NUE was 86% for the Silage subzone (NUE 3%). NUE was slightly higher in 2015 (93%), compared to 2016 and 2017 (80 and 85%, respectively).

Mean grazed herbage yield for Silage fields was 5,405 kg DM ha^{-1} (SD 2,431 kg DM ha^{-1}) coming from, on average, 4 grazing's (SD 2). Grazed herbage yield was highest in 2017 (6,189 kg DM ha^{-1}), and lowest in 2015 and 2016 (5,091 and 5,056 kg DM ha^{-1}, respectively). Mean silage yield from Silage fields was 6,892 kg DM ha^{-1} (SD 2,593 kg DM ha^{-1}), and the mean number of harvests was 2 (SD 1). Silage yield was highest in 2015 and 2016 (7,156 and 6,942 kg DM ha^{-1}, respectively) and lowest in 2017 (6,579 kg DM ha^{-1}).

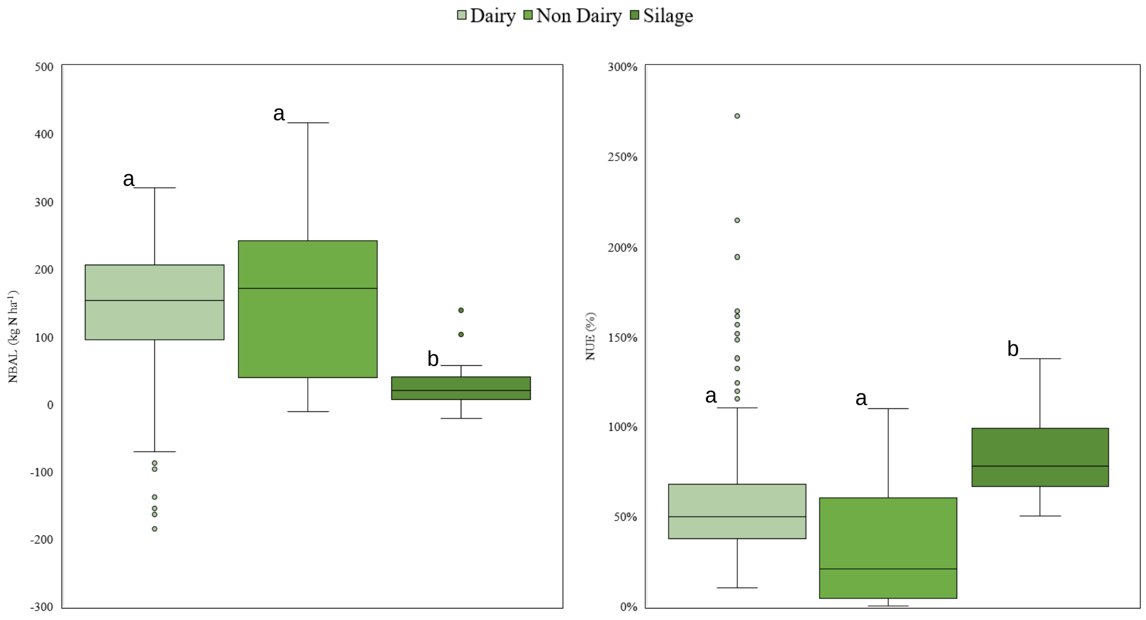

Figure 3-5. Mean nitrogen balance and use efficiency at field scale for Dairy paddocks (*n* = 399), Non-Dairy (*n* = 29), and Silage (*n* = 18) fields between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post-hoc test).

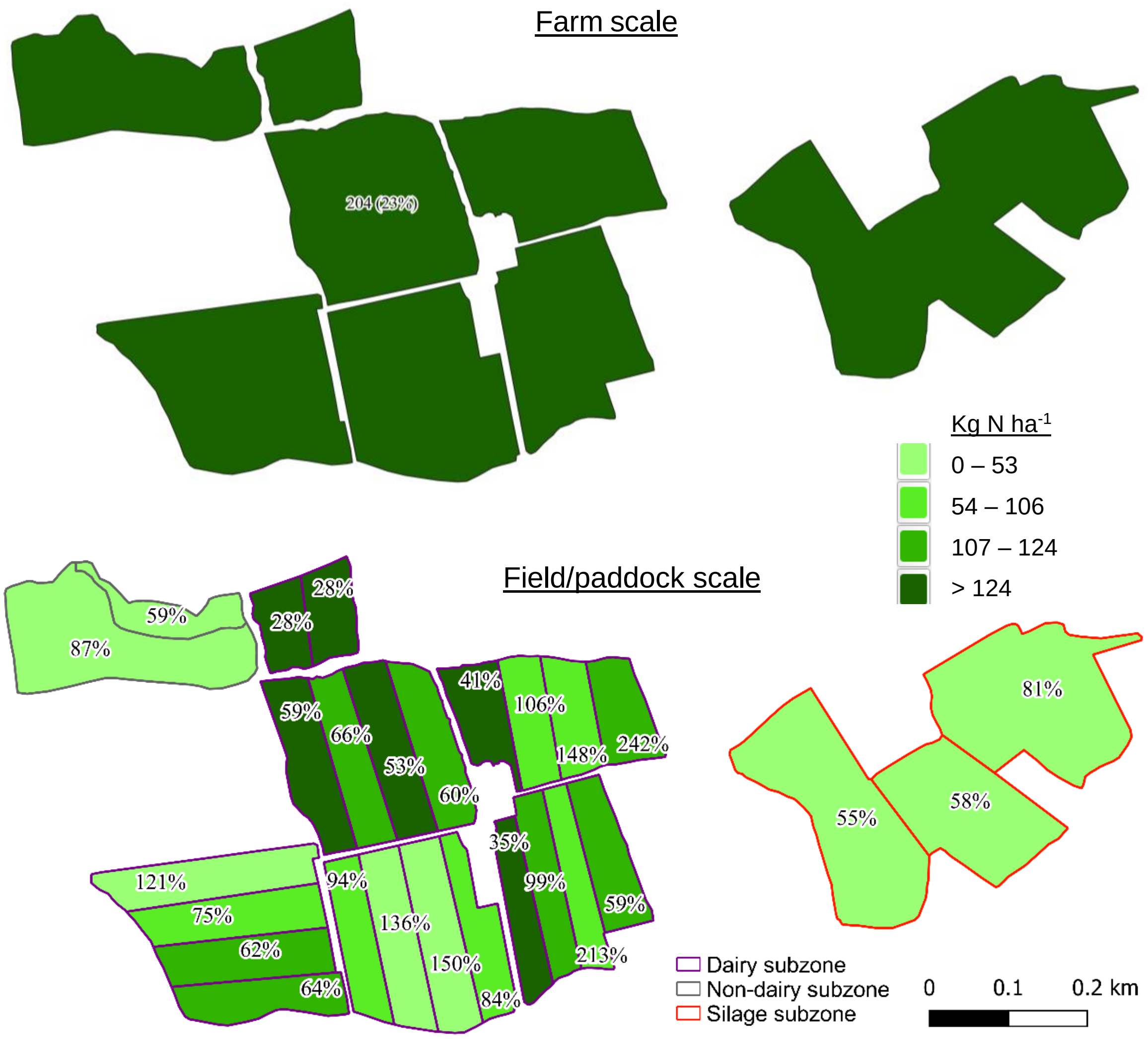

Figure 3-6. Map of mean nitrogen balance and use efficiency for one farm at two scales, farm and field. Spatial distribution of these nitrogen management indicators is shown for each individual paddock and field across the whole farm and each subzone. Dairy subzone is indicated by fields with purple boundary. Non dairy subzone is indicated by fields with grey boundary. Silage fields indicated by fields with red boundary. Mean nitrogen balance value is indicated by colour differences, darker shades indicate higher values. Mean nitrogen use efficiency value (%) is labelled per field.

3.4.6 Field scale factors affecting N balance and use efficiency

Field scale NBAL and NUE were significantly affected both by environmental and management factors (p < 0.05). However, NBAL and NUE were impacted differently by each factor i.e. NUE was significantly affected by more factors than NBAL. Also, both mean NBAL and NUE tended to be impacted by pasture management related factors (sward species, cropping and reseed), while only mean NUE was impacted by soil fertility management factors (soil P and K).

3.4.6.1 Environmental factors

Gradient class did not significantly impact field scale N sustainability. However, for soil drainage class, mean NUE was significantly higher on well drained fields (59%) compared to poorly drained (46%) (Figure 3-7).

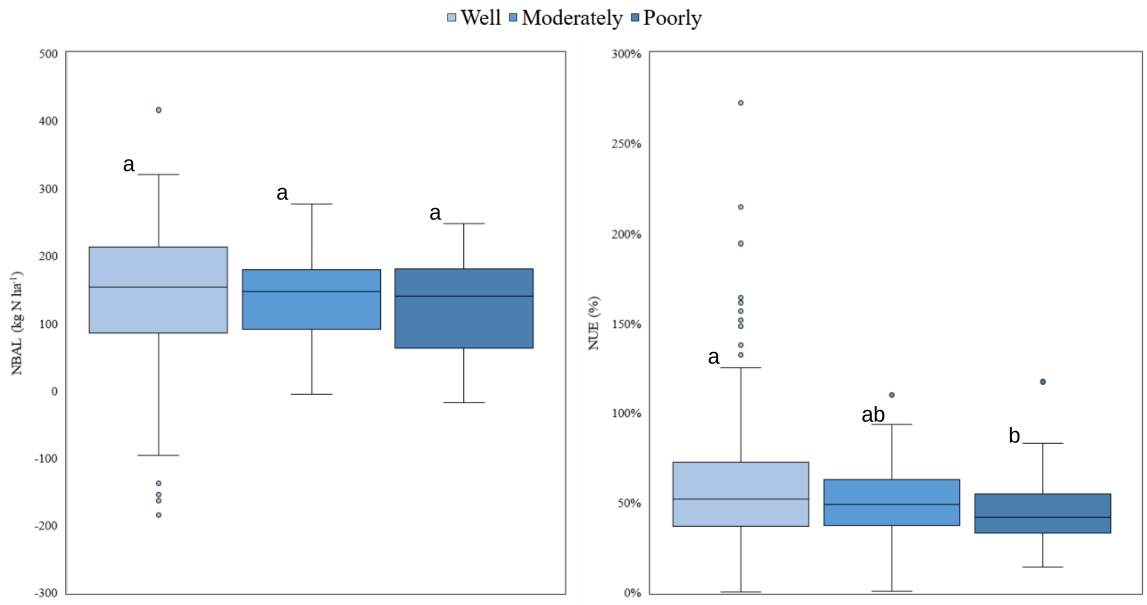

Figure 3-7. Mean nitrogen balance and use efficiency at field scale as affected by soil drainage class (well, moderate or poorly drained fields) between 2015 and 2017. Differences in letters represent significant differences between groups (p < 0.05, Bonferroni post hoc test).

3.4.6.2 Management factors

For soil fertility, only soil P and K index were found to significantly impact mean NUE. Generally, the lowest soil index was significantly different to one of the higher indices (Figure 3-8). Mean NUE for P Index 1 (50%) was significantly lower than P Index 3 (59%). Similarly, mean NUE for K Index 1 (44%) was significantly lower than both K Index 3 and 4 (57 and 60%, respectively).

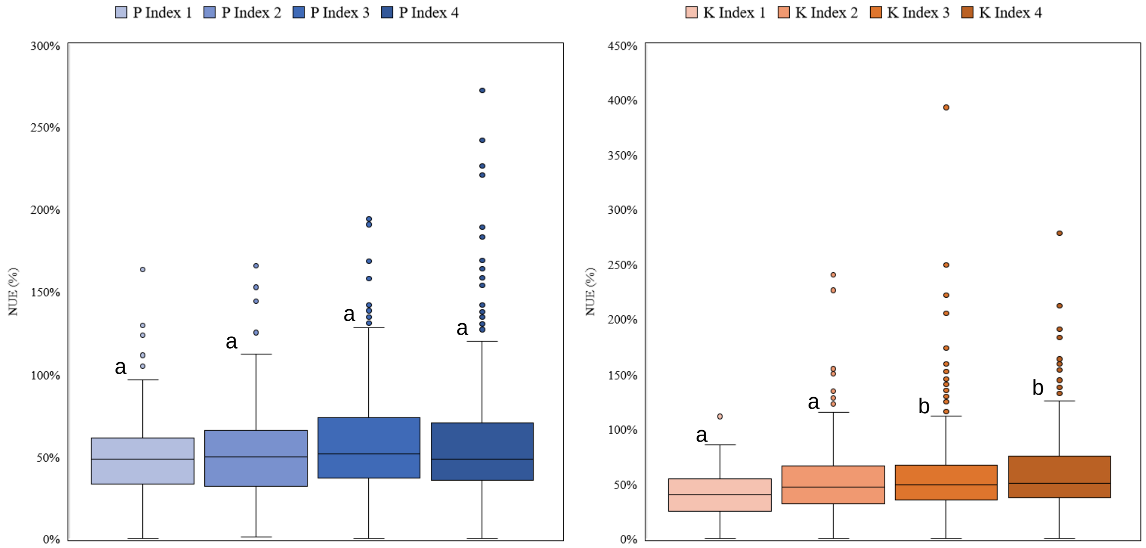

Figure 3-8. Mean nitrogen balance and use efficiency at field scale as affected by phosphorous and potassium soil status (Index 1: Very Low, 2: Low, 3: Medium/Adequate, 4: Sufficient/High) between 2015 and 2017. Differences in letters represent significant differences between groups (*p* < 0.05, Bonferroni post hoc test).

The proportion of perennial ryegrass (PRG) had a significant impact on N sustainability. Mean NBAL was significantly higher on PRG (High) (168 kg N ha^{-1}), compared to PRG Medium and Low (139 and 115 Kg N ha^{-1}, respectively). However, mean NUE was significantly higher for PRG (Low) (61%) compared to PRG (High) (51%). It was found that white clover (WC) class did not significantly impact N sustainability.

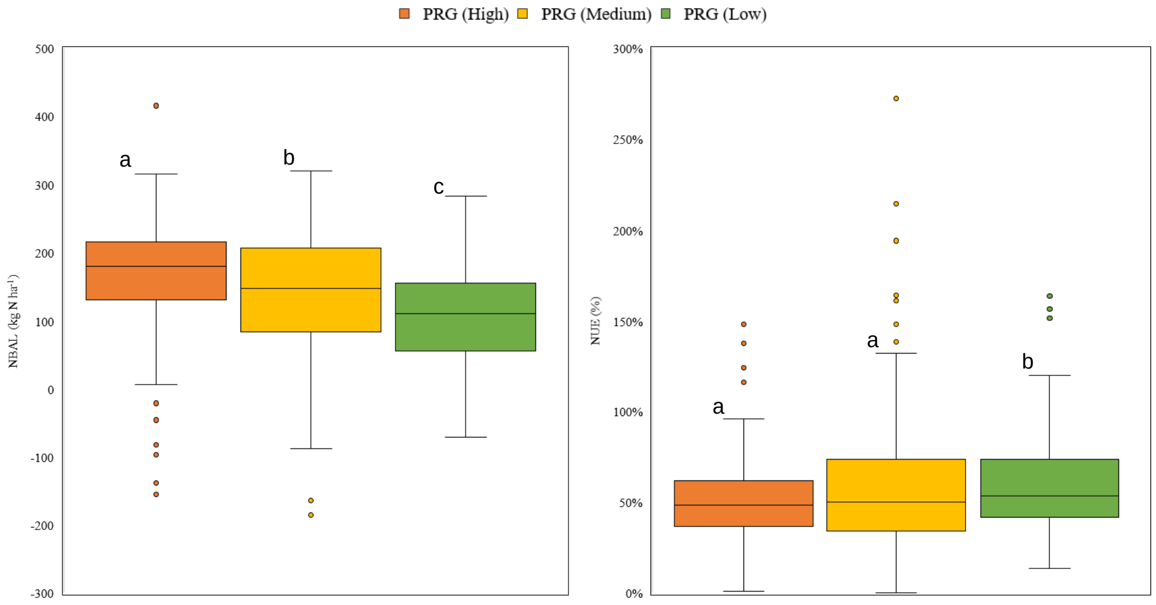

Figure 3-9. Mean nitrogen balance and use efficiency at field scale as affected by perennial ryegrass class (high, medium or low sward level) between 2015 and 2017. Differences in letters represent significant differences between groups (p < 0.05, Bonferroni post hoc test).

Both cropping and reseed impacted N sustainability at field scale (Figure 3-10 and Figure 3-11). Mean NBAL was significantly higher on Grazing Only fields (179 kg N ha^{-1}), compared to Grazing + Harvest and 2 Cut Harvest (68 and 35 kg N ha^{-1}). Mean NUE was significantly lower for Grazing Only (44%) however, compared to Grazing + Harvest and 2 Cut Harvest (83 and 78%).

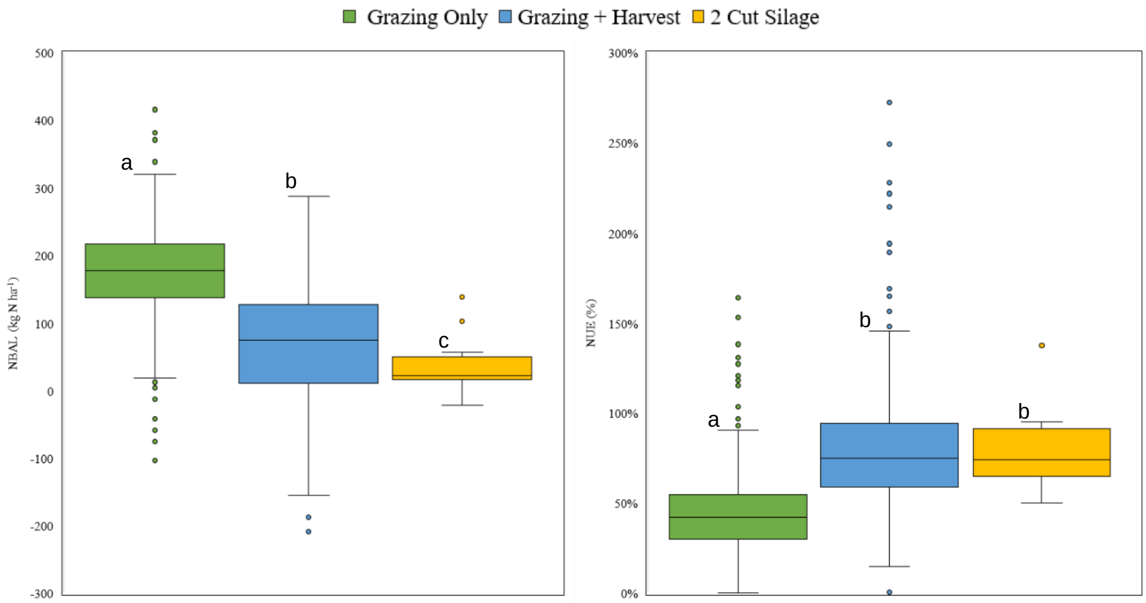

Figure 3-10. Mean nitrogen balance and use efficiency at field scale as affected by cropping strategy (grazing only, grazing + harvest or 2 cut silage/harvest) between 2015 and 2017. Differences in letters represent significant differences between groups (p < 0.05, Bonferroni post hoc test).

Mean NBAL was significantly lower on the newest pastures (< 5 years) (125 kg N ha^{-1}) compared to pastures that were 5 to 10 years old or > 10 years old (138 and 160 kg N ha^{-1}) (Figure 3-11). Mean NUE was significantly higher on newer pastures (59%) compared to older pastures (> 10 years) (52%).

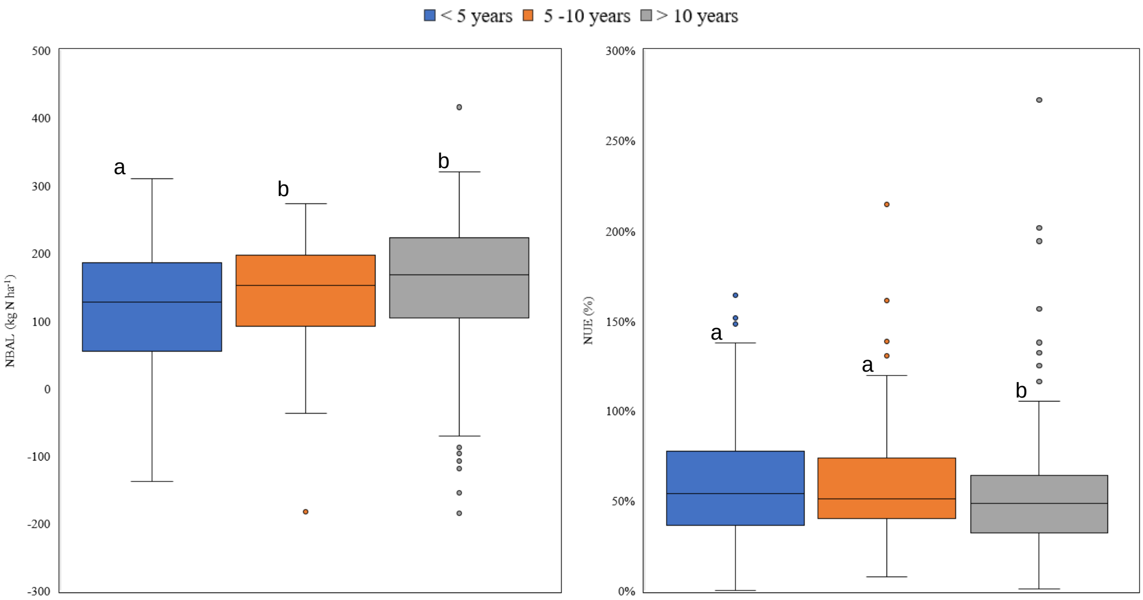

Figure 3-11. Mean nitrogen balance and use efficiency at field scale as affected by time since reseeded (< 5 years, 5 - 10 years or > 10 years) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post hoc test).

3.5 Discussion

3.5.1 Variation in NBAL and NUE between scales

Firstly, farm and field scale NBAL, were both driven largely by inorganic N fertiliser import (81% of imports at farm scale and 77% at field scale), and milk export (98% of exports at farm scale and 72% at field scale) (Table 3-4). Efforts to improve N management on specialised grass-based dairy farms such as these, should therefore, focus on optimising fertiliser N management and milk production, as has been suggested by (Erisman et al., 2007; G.J. Hilhorst et al., 2001; Galloway et al., 2003; Oenema et al., 2012). This may include, strategies to reduce fertiliser N input, through the use of clover or multispecies swards, for example Egan et al. (2017) and Murphy et al., (2018).

Although there was no significant difference in mean NBAL between farm and field scale, variability in NBAL (CV 34 and 77%, respectively) and NUE (CV 39 and 67%, respectively) was far greater at field scale (Figure 3-3). This has important implications for the N source pressure and efficiency of N management. While farm scale NBAL appears to be a good indicator of the *average* N source pressure across a farm area, NBAL at the farm scale does not reflect the large variability in *actual* NBAL at the field scale or its distribution (Figure 3-6). It is NBAL at the field and soil scale that will reflect actual N cycling between soil and plant and losses to the wider environment (Di et al., (1998) and Van Leeuwen et al., (2019). For example, the quantity of N that is captured via silage events at field scale (and not only specific to fields in the subzone) was shown to be an important factor contributing to field scale NBAL variability. This has important implications for policies that target improved N management, and reduced losses to the environment, and suggests that management practices targeted at areas of high NBAL and low NUE at field scale, might be most effective. On dairy farms such as these, the paddock scale (ca. 0.5 – 2.5 ha) might be a suitable scale to target improved N management practices as these are already established as spatial management units (Figure 3-2).

The fact that mean NUE was significantly higher at field (57%), than at farm scale (28%) (Figure 3-3), suggests that higher NUEs can be achieved at field scale than would be suggested by farm scale values. Differences in the number of and type of variables being measured at each scale is a factor in this result. The farm scale requires less variables than field scale for calculating a budget and therefore lacks resolution in comparison. Additionally, calculating the balance at farm scale and averaging it across the whole farm area is different to calculating the balance at field scale at multiple points and averaging those values. There is added resolution to the field scale result because there are more points of data to draw from. For example, for one farm scale nutrient balance calculation there are approximately 30 field scale nutrient balance calculations depending on the number of paddocks. Focusing improved N management on paddocks or fields with low

NUE, might be an effective strategy to improve NUE. This would then be expected to translate to improved NUE at the farm scale also. Such a strategy may not be currently practicable for farmers though. The sheer number of variables and number of changes that they are subject to make it impractical for farmers to calculate these nutrient balances by themselves. From a research perspective the approach does however provide insight into nutrient cycling pinch points i.e. the where and why key nutrient pressures occur on grass based dairy systems. Which can then be translated into practical advice by knowledge transfer authorities. Data recording software, although available in different capacities, does not yet support this approach in Ireland.

3.5.2 Farm scale factors affecting variation in NBAL and NUE

Farm scale NBAL and NUE differed significantly between farms, as was expected. Mean farm scale NBAL, was not significantly affected by soil drainage class, but NUE was. This suggests that, while N source pressure at the farm scale, may be more affected by other factors, rather than soil type, the efficiency of the conversion of imported N to exported N, was significantly affected by soil drainage class. Farms on poorly drained soils had the lowest mean NUE. This could be due to a range of factors associated with more poorly drained soils, such as greater N losses via denitrification, or lower yields due to reduced N uptake in herbage (Shalloo, et al., 2004a). This outcome was also found at field scale, and highlights the importance of soil type management and soil fertility management across both scales.

Although the relationship was not significant, NBAL tended to increase with stocking rate, reflecting greater N inputs in fertiliser and feed to support higher stocking densities, as has been found in many studies (Buckley et al., 2016a; Mihailescu et al., 2014). Stocking rate is a driver of leaching losses and it has been shown that losses can be higher on grazed pasture soils, particularly when stocking rates are also high (Horne, 1980; Ryden et al., 1984). The results of this study suggest that higher stocking rates have higher NUE. Nonetheless, studies have shown that even when a higher NUE is achieved, higher stocking rate classes can be more susceptible to losses to the environment due to the associated higher NBAL (Daatselaar et al., 2015).

The fact that mean farm scale NBAL and NUE was not significantly different between years, indicates that N source pressure and resource use efficiency, did not vary significantly over time on these farms, at least in the short term (1 to 3 years). Policy change (EU milk quota abolition) in 2015 and national targets for increased milk production under the Food Harvest 2020 programme, might have been expected to increase milk export and livestock numbers, and therefore, possibly NBAL at farm level over this period (2016 and 2017). In fact, mean NBAL was numerically highest in the final year of the study (2017), though not at a statistically significant level ($p > 0.05$). Despite changes in import and export values in between years, the NUE at both scales remained the same. This indicates that there is at least some capacity for dairy farms to alter N management strategies, and for example, increase herd size, without compromising NUE at farm and field scale. However, Goulding (2000) and others have shown that N loss is determined more by the N load per ha than NUE, and NBAL is a more direct indicator of the local nutrient source pressure than NUE. These results would indicate an overall increase in N source pressure in the final year of the study. Further monitoring, particularly using nationally representative data such as the Teagasc National Farm Survey (Buckley, 2012 and Buckley et al., 2015), would be required to confirm if this trend continues and is representative of dairy farms nationally.

3.5.3 Subzones and field scale factors affecting variation in NBAL and NUE

Further to the differences in N cycling between farm and field scale, the field scale evaluation highlighted notable differences between subzones (Table 3-5, Figure 3-5, and Figure 3-6). Somewhat surprisingly, Dairy and Non-Dairy subzone NBAL and NUE were not significantly different to each other. It had been expected that the Dairy subzone would have the highest NBAL due to the concentration of grazing dairy cows on this part of the farm (the milking platform). This result indicates that N source pressure can also be reasonably high on Non-Dairy grazed areas of the farm.

The high degree of variability in NBAL (CV 74, 79 and 167%) and NUE (CV 65, 108 and 3%) found between subzone paddocks/fields, suggests that there is significant scope to improve N management, by focusing at the paddock/field scale, and thereby reduce N source pressure and improve N use efficiency. Figure 3-6 illustrates this for one of the farms in the study, where NBAL is notably higher and more variable (greater proportion of > 124 kg N ha^{-1} paddocks) on the Dairy subzone (milking platform) than the silage subzone (only < 53 kg N ha^{-1} fields), and NUE is lower and more variable (32 – 93%, compared to 72 to 74%, respectively). These results would suggest that better matching of N inputs, particularly of inorganic N fertiliser, and N deposited by grazing animals, with the N requirement for herbage growth, particularly in grazing and silage fields, focused at the paddock/field scale could have significant potential, to improve N use efficiency and reduce N source pressure on the environment. Although this is expected to be very difficult to achieve in practice, grass budgeting strategies and timely knowledge transfer advice around herbage demand differences between months and seasons may help to increase the capacity of matching N inputs to herbage requirements.

In terms of herbage accumulation (measured weekly by cut and weigh, plate meter, and eye-ball estimates), it was shown that Non-Dairy fields had slightly lower grazing and slightly higher silage yield than Dairy paddocks. Dairy paddocks had the highest total herbage accumulation (grazing and cutting) of 13,381 kg DM ha^{-1}, on average, compared to 11,557 kg DM ha^{-1} for Non-Dairy and 12,297 kg DM ha^{-1} for Silage fields. Overall, Dairy paddocks had higher grazing accumulation and number of defoliations than both Non-Dairy and silage fields, which is expected from the grazed grass approach to dairying employed on these farms. Dairy paddocks had the lowest silage yield and number of harvest events also (Table 3-5). Harvest events are not always suitable on dairy paddocks due to the recovery time required afterwards (exclusion of the field from the grazing rotation temporarily) but are necessary to manage herbage production in the peak times of the growing season.

The fact that Silage subzone fields mean NBAL (27 kg N ha^{-1}) was significantly lower, and NUE (86%) was significantly higher, than both Dairy and Non-Dairy subzones indicates a significantly different N cycling regime and resultant N source pressure and use efficiency on this subzone of these dairy farms. As expected, silage fields had the lowest grazing yield and highest silage yield, which would be representative of the 2 Cut Harvest management system. As a result, the most notable factor that separated the subzones was the quantity of N that was captured by silage harvest. The N captured in silage harvests on the silage subzone was greater than that total N captured in milk and animals sold on Dairy or Non-Dairy fields which lead to silage fields having a lower NBAL and higher NUE than other fields. Essentially, fields in the silage subzone are managed as a crop. Many studies have shown that there are lower NBALs and higher NUEs on cropping (tillage) systems compared to livestock systems due to the greater incidence of N loss from livestock systems (Ledgard et al., (1999), Čop et al., (2009) Schlegel et al., (2016)). Di et al., (1998) have shown that pasture that is mown or harvested for hay or silage also have very low N leaching losses. The main influence in this study may be the total silage yield recorded by the farmer. It must be noted, that the N harvested in silage, is not exported off the farm boundary as product like milk and animals sold, but it is removed from the field in the year in question. Instead, the N in silage is recycled within the farm boundary through forage feeding animals in the winter period. The N that the animal does not capture by ingestion is collected in organic manures. This N is then redistributed across the farm to any paddock or field (but predominantly to the silage subzone) in the following growing season and/or lost via emissions in storage or application. This highlights the importance of budgeting for N appropriately in grass-based dairy systems; notably, the importance of accurately accounting for N applied in organic manures.

This evaluation of field/paddock scale subzones provides further insight into N variability than field and farm scale comparisons alone. The approach highlights the potential of managing N on a subzone basis. For example, it may be appropriate to designate silage cutting to parts of the farm with high N surpluses as an approach to improve NUE across the farm. Silage fields only occupy approximately 15% of the TGA (mean field size 2.6 ha), and were very effective at capturing N. Non-Dairy fields occupy approximately 28% of the TGA (mean field size 2.8 ha) and have been shown to have the least efficiency of the three subzones. Addressing N efficiency on this subzone, in particular, may be effective in reducing overall NUE on farms. Finally, Dairy paddocks occupy approximately 57% of the TGA (mean field size 1.4 ha) and so, represent the greatest area of the farm to reduce surpluses. Results indicate that there is significant capacity within both the Dairy and Non-Dairy subzones, in particular, to improve efficiency and reduce N source pressure by managing N more appropriately at the field/paddock scale. The data limitations on Non Diary and Silage subzones, although budgeted for, introduce uncertainties in their values which is an inherent problem with a field scale approach on commercial farms. The means to implement such changes may not be practicable for farmers however but the support and adoption of appropriate data capturing technology by policy makers may make it possible for knowledge transfer agencies to generate this level of N targeting.

Evaluation of field environmental and management factors highlighted a number of insights that are important in influencing differences in NBAL and NUE between fields/paddocks. These might be useful to guide more targeted field/paddock specific N management strategies. Interestingly, pasture management factors (sward species, cropping and reseed) impacted both indicators of N sustainability, while environmental factors (soil type) only impacted NUE. Overall, this highlights the importance of informed management strategies that consider all the factors that cause or influence within farm variability.

The fact that NBAL increased with increasing PRG content in the sward suggests higher N inputs to fields/paddocks with higher levels of PRG in the sward, likely to support higher herbage yields. PRG, as a grass species, responds well to synthetic fertiliser N input (Whitehead, 1995). However, as these results show, that can lead to higher N surpluses and therefore higher N source pressure. White Clover class (present/absent) did not significantly affect NBAL or NUE for any subzone, similar to plot studies by (Egan et al., 2017; Enriquez-Hidalgo et al., 2014), which suggests that PRG class was more important as a driver of NBAL in a sward. The potential benefits of WC in terms of reducing the need for N fertiliser inputs, which are well established (Anjo and Jan, 1997; Egan et al., 2018; Ribeiro Filho et al., 2003; Ribeiro Filho et al., 2005), may have been confounded by high inputs of fertiliser N which could mask the fixation process of rhizomes in clover. This suggests that swards may not have been managed to utilise the positive effects of clover in terms of N fixation and increased yield (Egan et al., 2018). It was noted that if biological fixation by clover was included in this methodology it may have significantly influenced the N cycling results and should be considered in future field scale studies. There are also benefits to milk production from mixed swards (Ribeiro Filho et al., 2005) and potential reduced risk of nutrient losses, by reducing fertiliser N input to appropriate levels. This assessment of sward species at the field/paddock scale highlights the potential to improve

NBAL and NUE through correctly managing N fertiliser inputs relative to sward species composition.

Nitrogen applied in fertilisers, or N fixed by legumes, may be efficiently captured by pasture plants, and so N losses can depend on the method of cropping (Cameron et al., 2013), such as grazing or harvesting (Monaghan et al., 2007; Ryden et al., 1984). Grazing fields have higher urine and dung patches per unit area compared to regularly harvested fields and this contributes to leaching rates also (Silva et al., 2000; Stott and Gourley, 2016; Stout et al., 1997). Consistent with this, fields which were Grazing Only fields had the highest NBAL and lowest NUE (Figure 3-10). This result was likely due to the lack of harvest events (no N offtake in silage cuts), and higher number of grazing defoliations, each of which is typically followed by fertiliser N application, throughout the growing season (Table 3-5). In contrast, Harvest Only fields, were found to have the lowest N surplus and highest NUE, because of the combination of a 2 cut silage cropping strategy and high yields (Table 3-5). These results indicate that N source pressure, and N losses to the environment, are likely to be highest on Grazing Only fields, and illustrates the effectiveness of silage harvesting at capturing N. This suggests that policy and management efforts to improve NUE and reduce risks of N loss to the wider environment might best be targeted at the grazing or milking platform on such specialised dairy farms.

The age of a sward can also influence within farm variability and the capacity to capture N from fertilisers. Previous studies, for example, found a difference of 40% in N leaching between an old and new pasture (129 and 50 kg N ha^{-1} yr^{1}, respectively) that received the same amount of fertiliser N (400 kg N ha^{-1}) (Jarvis, (2000), Scholefield et al., (1993)). Lower NBAL and higher NUE were associated with fields that were reseeded in the last 5 years or between 5 and 10 years, compared to fields reseeded over 10 years ago (Figure 3-11). This suggests that better N efficiency can be achieved through shorter reseeding intervals (less than 10 years). The gain in N efficiency may be due to a high demand for N by the high proportion of PRG in the early years of a newly established sward (Jarvis, 2000). Reseeding must be considered carefully however. Ploughing as part of reseeding can mineralise large amounts of nitrate (>100 kg N ha yr^{1}) (Wild and Cameron, 1980) from soil organic matter, which can be vulnerable to loss to the environment (McAleer et al., 2017). Minimising the area of ploughing, or alternative approaches to reseeding, may be more suitable for reducing N losses to the environment.

NUE was significantly lower on low P and K index than higher P (3) and K Index (3 & 4). Low N capture is associated with acidic soil and low levels of available K (Gibbons et al., 2014). Correct soil fertility can be expected to improve N capture (Scholefield et al., 1993; Watson et al., 2002). It was expected that soil pH would influence N use but it was not significant for this study. Where soil fertility levels are insufficient, the risk is highest for losses to the environment and economic costs (Gibbons et al., 2014; Scholefield et al., 1993; Scoones and Toulmin, 1998; Watson et al., 2002), for example through higher N inputs, which may confound the impact of poor soil fertility. These results suggest that more targeted (reduce low soil fertility levels), and optimised management of other nutrients at field/paddock scale, could help improve NUE and NBAL, and they also emphasise the importance of a holistic approach to nutrient management.

3.6 Conclusion

We conclude that, while the N use indicators NBAL and NUE, at farm scale, do reflect the average overall N source pressures and use efficiencies for a farm, they do not adequately reflect or capture the variation in N flows, N source pressures, and the likely resultant N losses to the environment, that occur at the individual paddock and field scale. This comparison of sustainability of N management provided valuable insights into how the N cycle transitions from farm to field scale, revealing some of the differences and factors that alter N flows and cause their variability. For example, the milking platform (mean paddock area 1.4 ha) was impacted significantly by environmental and management factors that the farm scale (mean area 69.3 ha) did not provide information on. The evaluation of field scale factors, shows that much of the efficiency of N use, is potentially controllable through farmer management decisions at the field scale. Although the strategy may not be practicable for farmers the approach does however provide insight form a research perspective into the pinch points on grass based dairy systems for efficient nutrient management. The range in NUE values suggests that high NUE was being achieved on some fields, while low NUE was achieved on others. Without measuring such indicators at the field/paddock scale, this variability is invisible to the farmer, advisor and policy maker. At the farm level, the NUE range is much narrower and shows less opportunity for improvement which affects the perception of the farms actual performance. Farm level reporting of NUE, therefore, does not inform managers and policy makers of the higher NUE values that are being achieved inside the farm boundary, as well as not highlighting the spatial variability which needs to be managed appropriately.

Results suggest that there may be significant potential to improve N efficiency and reduce N source pressure, and losses to the wider environment, through management practices, targeted at the field/paddock scale, that are appropriate to soil drainage conditions, topography, soil fertility, and sward composition. Further, they suggest that these management practices might differ between subzones of these dairy farms; notably, between the grazed area of the farm and the silage area. These management practices might include, for example, optimising soil fertility (pH, P, K), making more use of grass clover and multispecies swards, optimising N fertiliser management to sward composition and N demand, optimising and accounting for N applied in organic manures, strategic silage harvests, including on the grazed area of the farm, and appropriate reseeding intervals.

The current maximum soil sample area for soil nutrient analysis under the Nitrate Regulations in Ireland is 5 ha. Results from this study would suggest that this may be too large an area to effectively target nutrient management spatially, particularly on the dairy subzone fields of these farms. This study suggests that a field or paddock scale sampling and nutrient management approach, is likely to be effective at optimising nutrient management planning and minimising nutrient losses to the environment. Nutrient management planning strategies should ensure that field scale environmental and management factors are addressed appropriately, as well as being functional and practical, for both advisors and farmers, and transparent for policy regulators. In relation to national agricultural production and environmental goals in Ireland, this study suggests that there are opportunities at the paddock and field scale to produce specific N management strategies that would reduce N surplus and improve use efficiency without, necessarily, impeding production.

CHAPTER FOUR: A SPATIAL COMPARISON OF FIELD AND FARM SCALE PHOSPHORUS AND POTASSIUM BALANCE AND USE EFFICIENCY ON 15 IRISH PASTURE BASED DAIRY FARMS

Preface: This paper has been prepared for the submission to the journal *Nutrient Cycling in Agroecosystems*. Formatting and reference style have been edited for consistency throughout the thesis. Figure and table captions have been assigned with a chapter prefix. Acknowledgements have been removed.

Keywords: Phosphorus and Potassium cycling, Nutrient balance, Use efficiency, Indicators, Scale, Farm, Field, Paddock, Subzone

4.1 Abstract

Phosphorous (P) and Potassium (K) are both important nutrients in pasture-based dairy systems and are typically required in high quantities. Potassium is important for plant production as it improves disease resistance, enables increased protein content, and improves drought resistance capacity. Similarly, Phosphorus in agriculture contributes to increased yield and quality of plants and crops, but losses of P increase the risk eutrophication of waterways. A number of important management and environmental factors impact nutrient cycling spatially, particularly on dairy farms, as there are high nutrient requirements for adequate pasture production, and the need to reduce nutrient losses in a spatially appropriate way is crucial to maintain production of milk and meat. This study uses primary data from 15 commercial grazed grass dairy farms and a total of 389 fields/paddocks in Ireland (2015 to 2017), to compare farm, field, and paddock scale indicators of Phosphorus and potassium nutrient management (balance and use efficiency). It was hypothesised that 1) P and K nutrient cycling indicators would be significantly different, 2) nutrient distribution would be significantly different between fields and paddocks and 3) P and K nutrient cycling indicators would be impacted by environmental and management factors at the field scale. A total of 15 "farm-gate" and 389 soil surface nutrient balance (BAL) and use-efficiency (UE) calculations were carried out to capture the major Phosphorus and potassium flows at each scale. Results of farm and field P and K nutrient BAL and UE were compared using Analysis of Variance (ANOVA). Analysis of Variance was also used to test for effects of field environmental factors (gradient and soil drainage) and management factors (sward species composition, cropping rate, and reseeding interval). Phosphorus balance i.e. nutrient source pressure (PBAL) was significantly higher ($p < 0.05$) at the farm scale than at the field scale (9 and 6 kg P ha^{-1}, respectively). Potassium balance (KBAL) (23 and 39 kg K ha^{-1}, respectively) was not significantly different between scales. Phosphorus use efficiency (PUE) was significantly lower ($p < 0.05$) at farm scale (PUE 52%), compared to results at field scale (PUE 100%). Mean potassium use efficiency (KUE), was not significantly different between these scales (KUE: 56% and 91%, respectively). It was found that Phosphorus balance was significantly higher at farm scale compared to field scale ($p < 0.05$). It was also shown that higher nutrient balance variability occurred at field scale (CV 294%) compared to farm scale (CV 58%). Nutrient imports, via inorganic fertilisers and exports via milk sold, were the main drivers for P and K cycling between both farm and field scales. However, results also showed that much of the variation in nutrient P and K indicators, could be attributed to controllable grassland management factors, like sward species composition, cropping and reseeding rate, but also uncontrollable environmental factors like gradient and soil drainage class. Results of the

study also highlighted that there may be opportunity to target nutrient management strategies to specific subzones within dairy farms to reduce P nutrient source pressure for example. Mean field scale PBAL on Dairy fields (or the “milking platform”) was lowest in comparison to Non-Dairy and Silage fields, but variability was actually highest on these fields overall (CV 272%, 107%, and 72%, respectively). The milking platform, might be a suitable scale to target improved nutrient management practices, particularly for P, as these are already established as spatially explicit management units (paddocks ranged from ca. 0.5 – 2.5 ha). The differences observed between the farm and field scales, have implications for environmental risk, nutrient regulation and policy, compared to when these indicators are observed at farm scale alone. Information on increased precision in nutrient management strategies at subzone or field scale may help to guide future advisory service of best practices on dairy systems. Overall, the large spatial distribution and variation in field scale P and K nutrient management, highlights that there were areas of high risk of environmental losses but also areas of very low risk, which were invisible to farmers, advisors and policy makers.

4.2 Introduction

Global phosphorus (P) reserves are predominantly mined for agricultural use which make up approximately 80-90% of the total demand (Vaclav, 2000). Phosphorous is finite and expected to run out in a small number of centuries (Withers et al., 2015). In contrast, potassium (K) is the 7th most abundant component of the earth's crust. However, natural release rates of K are slower than the demand of it for agricultural production, so it is mined from deposits around the world (Zörb et al., 2014). Phosphorus is required for better root development, increased stem strength, and also, increased Nitrogen (N) fixing capacity of legumes (Zörb et al., 2014). It's been described as an irreplaceable nutrient for both plants and animals (Dawson and Hilton, 2011). Potassium is equally important for plant production, as it aids in disease resistance, enables increased protein content, and improves drought resistance capacity (Kayser and Isselstein, 2005), which is becoming more essential with the impact of climate change. P use in agriculture contributes to increased yield and quality of plants and crops, but losses of P increase the risk eutrophication of waterways (Buczko et al., 2018; Buczko and Kuchenbuch, 2007; Correll, 1998; Vaclav, 2000).

There are a number of important management and environmental factors that impact nutrient cycling spatially (McCormick et al., 2009; Van Leeuwen et al., 2019). The risk of nutrient loss varies relative to site specific hydrological, and geophysical differences which alter the transport of nutrients, via erosion, run-off, leaching or drainage (McDonald et al., 2019; Buczko and Kuchenbuch, 2007; McDowell and Nash, 2012; Quinton et al., 2003; Sharpley et al., 1988). Distance from waterways also plays a role in the delivery of nutrient losses to water (Kleinman et al., 2011; McDowell et al., 2016; Melland et al., 2018). Farm scale nutrient regulation and prescription tends to differ to field scale nutrient management, which indicates a disconnect between the two scales. The lack of linkages between the two scales, in management or monitoring for example, means that spatial differences in nutrient losses may not be addressed appropriately, and could lead to unintentionally high nutrient losses to the wider environment.

McCormick et al. (2009), showed that variability is significant within fields, and that site-specific nutrient management had the potential to reduce nutrient (P) excesses through an intensive grid sampling approach on grass-based systems. Gourley et al., (2015), highlighted that at paddock scale, there was a need to reduce nutrient surpluses (P, K and Sulphur) on grass-based systems, and that a greater understanding of within farm nutrient distribution was needed. Similarly, McDonald et al., (2019) found that there was a need for a better farm and soil specific nutrient management planning strategy, to improve within farm P distribution, and that identifying agronomically balanced versus unbalanced fields, was critical. Both Ruane et al., (2014) and Rugoho et al., (2018), found that fluxes and variability in nutrient balances and variation in soil nutrients across farms (within and between farms) were agronomically sub-optimal, and that nutrient balance excess, beyond plant requirement, risks environmental damage. Rugoho et al., (2018), suggested that there is a need for a standardised assessment of nutrient fluxes, balances, and efficiency, at the farm scale, so as to provide dairy farmers, and advisors a better capacity to determine agronomic goals and environmental performance. Furthermore, Van Leeuwen et al., (2019) found that a field balance provided more informative data than a farm balance, specifically when variation in soil types and/or management factors is found within the farm. In that study it was recommended that the use of field scale nutrient balances, can better detect extreme surpluses and deficits, and lead to improved management decisions.

Farm scale nutrient management, planning, and regulation does not categorically address all of the source, mobilisation, and transport factors (Wall et al., 2011) that affect nutrient loss at field scale. There is a need to address these factors at a scale that's manageable for farmers, and will help to minimise nutrient losses. This study aimed to investigate the nutrient distribution of P and K within and between farms and fields (including paddocks) on commercial grazed grass dairy farms, and assess the impacts of field scale environmental and management factors on P and K cycling and risk of nutrient loss.

To do this, P and K balance and use efficiency were used as indicators. It was hypothesised that 1) farm and field scale P and K nutrient cycling indicators would be significantly different, 2) nutrient distribution would be significantly different between fields and 3) P and K nutrient cycling indicators would be impacted by environmental and management factors at the field scale.

4.3 Materials and methods

A summary of the materials and methods used is provided here. Further details of materials and methods can be found in Chapter Three.

4.3.1 Farm selection and farm characteristics

Farms were selected from the south and south east regions of Ireland, in counties Cork, Tipperary, and Wexford. These regions are typically associated with long annual growing seasons (~242 days), and a high proportion of well drained soils (NFS, 2017). Average annual rainfall for these regions ranged from 967 to 1,208 mm (2015 to 2017). Average air temperature ranged from 9.7 to 10.3 °C (2015 to 2017). Personal communication with agricultural advisors was used to identify dairy farmers willing to participate in the study. A total of 21 dairy farms were selected for this study, but only 15 participated fully (provided sufficient data to carry out the full suite of analysis for this study). All participants were specialised dairy farmers, with dairy cows contributing to more than 3/4 of standard agricultural output (CSO, 2016).

Table 4-1. Mean values and standard deviation for total grassland area, crop area, rainfall, air temperature, total utilisable agricultural area stocking rate, milk solids, and concentrate feed per cow and soil texture class for 15 Irish dairy farms between 2015 and 2017

Farm ID	TGA (ha)	Crop (ha)	Rainfall (mm yr^{-1})	Air temp. (°C)	Texture Class	TUAA SR (LU ha^{-1})	TUAA SR (kg Org N ha^{-1})	MS (kg ha^{-1})	Conc. (kg LU^{-1})
1	63.8	0.0	1,007	10.2	Sandy loam	2.5	209	813	674
2	35.0	0.0	1,208	10.1	Sandy loam	2.5	211	687	757
3	96.6	0.0	1,032	10.1	Sandy loam	2.2	191	631	298
4	44.7	2.3	1,032	10.1	Clay loam	2.6	225	1,185	866
5	58.3	0.0	967	9.7	Clay	1.7	146	624	1,106
6	34.9	1.8	1,007	10.2	Sandy loam	2.7	233	821	753
7	38.0	0.0	1,032	10.1	Sandy loam	2.0	170	797	938
8	105.3	8.4	1,208	10.1	Sandy loam	2.9	247	1,041	850
9	103.0	4.6	996	10.3	Sandy loam	2.8	238	1,128	1,128
10	83.0	0.0	967	9.7	Clay loam	2.1	177	734	1,027
11	130.1	18.2	1,007	10.2	Sandy loam	2.0	174	607	236
12	60.0	0.0	967	9.7	Clay	2.0	166	534	1,315
13	32.0	0.0	1,208	10.1	Sandy loam	2.6	219	1,089	867
14	73.3	40.4	992	9.8	Sandy loam	1.3	109	915	1,124
15	82.0	0.0	967	9.7	Clay loam	1.8	149	487	733
Mean	69.3	5.0	1,040	10		2.2	191	806	845
S.D.	30.3	11.0	90	0.2		0.5	39	222	295

TGA, Total Grassland Area; Temp, temperature; TUAA, Total Utilisable Agricultural Area; Sandy loam, Well drained soil; Clay loam; Moderately drained soil, Clay; Poorly drained soil; SR, Stocking Rate; LU, Livestock Unit; Org, organic; MS, Milk Solids; Conc., Concentrate feed; S.D., Standard Deviation.

Farm characteristics for the duration of the study period are summarised in Table 4-1. Average TUAA (Total Utilisable Agricultural Area) was 53.1 ha (includes total grassland and crop area), which was slightly below the national average of 56.0 ha for dairy enterprises in Ireland (NFS, 2017). Average stocking rate was 2.9 LU ha^{-1} (1 livestock unit is equivalent to one adult cow producing 3,000 kg milk annually, (Eurostat, 2013), which was higher than the national average of the previous three years: 1.75 LU ha^{-1} in 2012 to 2014 (Butler et al., 2017). Average milk solid production (fat plus protein) was 806 kg ha^{-1}, which was above the national average (649 kg ha^{-1}) for the 2012 to 2014 period. Supplementary fodder in the form of concentrate feed imported onto the farm was, on average, lower on these farms

(847 kg cow ^{1}) compared to the national average (926 kg cow ^{1}) (Butler et al., 2017). Ten out of the fifteen farms in this study were on average above the 170 kg organic N ha^{-1} restriction on stocking rate (SR) and were, therefore, in nitrates derogation conditions. Some farms crossed over the derogation threshold during the study period. Ten farms were on well drained soils (Sandy loam), three farms on moderately drained soils (Clay loam), and two farms on poorly drained soils (Clay). Meteorological data (rainfall and air temperature), was gathered from meteorological stations (Met Éireann, 2019) in the region.

4.3.2 Nutrient balance methodology

At farm scale or "farm gate", nutrient balance (BAL) and use-efficiency (UE) methodology (Eurostat, 2013) was used to account for the main nutrient flows at the farm scale that were under the direct control of the farmer. A soil surface methodology (OECD, 2007) was used to capture the main Phosphorus (P) and potassium (K) flows for field and paddock scale balance and use efficiency. Details of differences in imports and exports for both scales are shown in Table 4-2. Nutrient (P or K) balance (kg nutrient ha^{-1}) was calculated as:

$$P\ and\ K\ Balance\ (kg\ ha) = \frac{(P\ and\ K\ imports - exports\ (kg))}{Spatial\ area\ (ha)} \quad \textbf{Equation 4-1.}$$

$$P\ and\ K\ Use\ Efficieny\ (\%) = \frac{P\ and\ K\ exports\ (kg)}{P\ and\ K\ imports\ (kg)} X\ \frac{100}{1} \quad \textbf{Equation 4-2.}$$

Calculations represent grassland zones only (Total Farm Area subtract cereal crop area). Crop area nutrient flow data were excluded in the scope of the study to keep the results relevant to the pastoral P and K cycle. Farm scale was represented by the nutrient balance calculations carried out on the Total Grassland Area (TGA) of the farm. Field scale was represented by nutrient balance calculations carried out on individual fields/paddocks. All data were annual (calendar year) totals.

Table 4-2. Import variables (inorganic fertiliser, concentrate feed and organic manure) and export variables (milk sold, animals sold and silage) used to calculate nitrogen balance and use efficiency for farm and field (Dairy, Non-Dairy and Silage) scale

Import (kg P or K ha^{-1})		Farm	Export (kg P or K ha^{-1})	
Inorganic fertiliser	(+)		Milk sold	(-)
Concentrate feed	(+)		Animals sold	(-)
		Field		
Inorganic fertiliser	(+)		Milk sold	(-)
Organic manure	(+)	Dairy Paddocks	Animals sold	(-)
Concentrate feed	(+)		Silage	(-)
Inorganic fertiliser	(+)			
Organic manure	(+)	Non-Dairy Fields	Animals sold	(-)
			Silage	(-)
Inorganic fertiliser	(+)			
Organic manure	(+)	Silage Fields		
			Silage	(-)

In Table 4-2, inorganic fertiliser represents synthetic fertiliser (manufactured to contain specific minerals and chemicals) that is purchased and imported onto farms through the farm gate at farm scale or onto individual fields at field scale. Data on inorganic fertiliser was recorded by participants (purchase receipts) throughout each year. Nutrient values for synthetic fertiliser was considered to be 100% available to the plant. Concentrate feed represents the nutrient import value calculated from its protein content and total tonnage purchased which were recorded by participants each year. Nutrient import via concentrate feed was only counted for 36 weeks of each year to prevent double counting of nutrients in organic manure collected during the animal housing period. Nutrients imported by feeding animals concentrate feed during the NAP 16-week housing period (S.I. No. 31 of 2014, EU 2014) were counted in the typical values used for nutrient content of organic manures. Nutrient availability values were tailored according to application rate and season (based on date) in accordance Wall and Plunkett, 2016. Organic manure application data was recorded by participants throughout each year. Organic manure collected during the housing period was imported at field scale mechanically throughout the growing season. Organic manure represents animal excreta collected during the housing period only and excludes animal excreta deposited in-situ during the outdoor grazing season. The assumption was made that nutrients in excreta deposited during the grazing season were in constant cycle throughout the farm system. Milk sold represents the nutrients exported off farm via milk collection by processors. Data on milk sold was collected by the farmer annually through the Irish Cattle Breeder Federation database (ICBF) (ICBF, 2019). Nutrient value was based on total quantity sold and tailored by the farm’s average protein level. Animals sold represents the nutrients exported in the sale or movement of animal’s permanently off farm. Nutrient value was based on typical values used by McDonald, 2011. Nutrient values were further tailored by the use of animal age and weight (kg) in each individual case, which was collected by the farmer each calendar year through the ICBF database. Silage represents the nutrients exported via silage harvest events in a given year. Silage harvest events are not specific to fields in the silage subzone, and so silage harvests represent the action which may occur on any subzone. Nutrient value was based on the total silage yield and herbage nutrient concentration of an individual paddock or field. Silage yield data was recorded by the farmer on a weekly base and collated at the end of the calendar year. Herbage nutrient concentration data was based on the average of three herbage samples taken from each paddock in spring, summer, and autumn on each farm in 2015 and 2016. Field scale nutrient import values via concentrate feed, and export via milk sold and animals sold were proportionally weighted by each paddocks number of grazing defoliations and spatial area (ha). Grazing defoliations were a proxy for the number of visits each paddock experienced annually. Where a paddock had high or low defoliations and/or a large or small spatial area, nutrient load and nutrient export were weighted towards those differences. This weighting procedure does not apply to inorganic fertiliser, organic manure or silage harvest events.

4.3.3 Subzone characterisation

The TGA was divided into multiple zones for each farm. Total Grassland Area averaged 69.3 ha over the three years (Table 4-1). This grassland area was subdivided into Dairy, Non-Dairy and Silage subzones, within which, the individual fields and paddocks were monitored. ArcGIS® software (ArcGIS, 2016), was used to map individual paddock, field and farm boundaries. Bing maps aerial photography was used as a basemap (Bing™ Maps, 2005) and so paddocks and field boundaries were traced digitally first, then later, a ground truth was carried out.

The three subzones were characterised as follows:

- **Dairy subzone**: Fields or paddocks carrying dairy cows fully throughout each growing season. Dairy paddocks were the most numerous individual management unit of the three grassland subzones, and were commonly closest in distance, to the farm infrastructure (milking parlour and winter housing). On average, there were 27 Dairy paddocks per farm, with an average area of 1.4 ha. Dairy subzone fields made up, on average, 57% of the TGA of these participating farms.
- **Non-Dairy subzone**: Fields and paddocks that carried Non-Dairy livestock. These fields were used for pasture grazing, but for the purpose of calf rearing, or a beef enterprise. There were, on average, 7 Non-Dairy fields on each farm, with an average area of 3 ha. Non-Dairy subzone fields made up 28% of the TGA, on average.
- **Silage subzone**: Grassland fields that did not carry dairy, or other, livestock fully throughout a year and, instead, were used to produce grass for conserved winter forage. Silage was normally harvested from these fields twice annually, and, occasionally, animals grazed for a short time after harvest. Typically, these fields were larger in size, but less abundant. On average, there were 3.6 grass silage fields per farm, at 2.6 ha in area. Silage subzone fields made up 15% of the TGA, on average.

4.3.4 Data acquisition

Over the three years of the study, changes in farm size occurred on 8 of the 15 farms. This was due to temporary or permanent increases or decreases in area of land that was rented or purchased. For the analysis, 100% of Dairy fields were represented and 44% and 48% of Non-Dairy and Silage fields respectively were represented on average. The total project area (area with consistent monitoring data) was slightly smaller than the actual TGA for these farms (due to these changes in overall farm size), but the fields that were in the total project area were monitored in full throughout the three years. To accommodate these 8 farms, farm scale records and data that were representative of the total project area were used to represent the TGA. The remaining 7 farms had sufficient data for consistent monitoring at both scales.

4.3.4.1 Farm scale

Farm scale imports of total inorganic fertilisers (P and K), and imports of concentrate feed were recorded by the farmer. Nutrient exported, was based on P and K concentration in annual milk and animals sold off the farm. Annual total milk and animals sold was collected, with permission, using an online farm database; Irish Cattle Breeding Federation (ICBF, 2019). P and K concentration in feed, milk and animals, were based on typical values used by Macdonald et al., (2017). Losses via runoff and leaching were not measured.

4.3.4.2 Field scale

Field scale imports included N from inorganic and organic fertilisers, and concentrate feed. Fertiliser imports were based on individual field records made by the farmer. Total concentrate feed purchased, was apportioned across the farm as were total exports via total milk and animals sold using an apportionment procedure.

On the Dairy subzone, information on pasture production (grazing and silage yield, no. of defoliations and harvests) was recorded at field and paddock scale by the farmer on a weekly basis by cut and weigh, plate meter, and eye-ball estimates, depending on farmer preference. Grazing yield is the total herbage consumed by livestock and silage yield is the total herbage removed by machinery. This information included the number of defoliations due to grazing and/or harvests, as well as the herbage dry matter (DM) accumulation (kg DM ha^{-1}) per field. Only a portion of Non-Dairy and Silage fields had pasture production data recorded (average 44% and 48% per farm respectively) and so comparisons were limited to fields with sufficient data Annual values for Farm scale total N imported, via concentrate feed, and exported via milk and animals sold, were then apportioned to each paddock, relative to the total number of fields with grazing defoliations at the end of each calendar year. This was carried out because Aarons et al., (2015) and Topp et al. (2007) showed that a balance calculated across grazing rotations (i.e. number of defoliations) reveals variation due to grassland management practices. For this study, this approach was adopted to capture the variation in N cycling data at the paddock scale. Note also, for N flow via animals sold from Dairy paddocks, an assumption was made that, annually, a dairy cow had a 50 kg calf that was removed from the subzone and was replaced by a maiden heifer, leaving a final balance of 225 kg imported. This calculation was based on a typical annual replacement rate of 20% of the dairy herd, with a maiden heifer target weight of 275 to 330 kg (Kennedy, 2014).

Non-Dairy fields had different animal types (young cattle or sucklers) and lower stocking rates than the Dairy subzone. The total N export via animals sold off these fields, was calculated based on animal type and age. Number of animals sold each year was collected online (ICBF, 2019).

For Silage fields, in practice, some farmers carry out grazing after harvest events but this is commonly for a very short time and at very low stocking rates (Hyland et al., 2018). Where animals did graze on silage fields, the nutrients in animals sold was captured in the Non-Dairy fields balance calculation, as values overall, were small.

To calculate nutrient export via silage harvest events, three herbage samples were taken and pooled from each monitored field to determine herbage nutrient concentration. Samples were taken in spring, summer and autumn, and a mean annual P and K value was calculated. Herbage value (g/kg DM), was chemically determined by colorimetric analysis, following hot acid digestion using sulphuric acid, as outlined by Byrne, (1979). Export of P and K via silage, was then calculated, based on mean annual herbage value, and annual total herbage silage yield.

4.3.5 Farm scale factors

At the farm scale, two major factors that were expected to impact the P and K cycle were assessed: soil drainage class and farm stocking rate class.

4.3.5.1 Farm soil drainage class

The national soil map (1:250,000), developed by the Irish Soils Information Systems (Teagasc, 2019), was used to determine the farm level drainage class. At each farm location, the soil association was identified firstly. The predominant soil type, and its drainage characteristics, were then identified using the soil map and catalogue system.

4.3.5.2 Stocking rate

Stocking rate as a factor was based on kg Org N ha^{-1}, instead of LU ha^{-1}, and used for regression analysis. Animal organic stocking rates are based on annual nutrient excretion rates for livestock, as specified in the NAP regulations (S.I. No. 605 of 2017 (EU, 2017) and S.I. No. 65 of 2018, EU, (2018)).

4.3.6 Field environmental factor classification

At the field scale, two environmental factors that might be expected to impact the P and K cycle were assessed: gradient and field soil drainage class.

4.3.6.1 Gradient class

Fields were classified as sloping or level. Level was defined as < 10 degrees of slope and Gradient as >10 degrees of slope (Roberts and Watkins, 2014).

4.3.6.2 Field soil drainage class

Each field was then assigned a drainage class based on the survey, auger and other available information. Fields were classified as well drained (Sandy loam), moderately drained (Clay loam) and poorly drained (Clay) (CTECO, 2010; Simo et al., 2007)

4.3.7 Field management factor classification

Three field management factors that might be expected to affect P and K cycling were assessed: sward species composition, cropping rate, and reseeding interval (Necpálová et al., 2013). Soil fertility as a factor was explored separately in Chapter Five.

4.3.7.1 Sward species composition

Perennial Ryegrass (PRG) was grouped into three different proportions: 0 to 75% PRG (Low), 76 to 90% PRG (Medium), and 91 to 100% PRG (High). White clover (WC) was grouped into two levels: WC absent and WC present.

4.3.7.2 Cropping rate

Each field was classified into one of three cropping classes; Grazing Only (GO), Grazing + Harvest (G+H), and 2 Cut Harvest (2CH). Grazing events will remove nutrients in grazed grass, but will also add nutrients, in grazing animal excreta. Harvest events will remove nutrients in harvested grass but there will be no nutrients returned in grazing animal excreta.

4.3.7.3 Reseeding interval

Fields were grouped into three categories based on the number of years since a reseeding event occurred; < 5 years, 5 to 10 years, and > 10 years.

4.3.8 Statistical analysis

Linear regression was used for farm scale and stocking rate analysis. To investigate differences in P and K cycling between scales, results of farm and field nutrient BAL and UE were compared using Analysis of Variance (ANOVA). Analysis of Variance was also used to test for effects of field environmental factors (gradient and soil drainage) and management factors (sward species composition, cropping rate, and reseeding interval). All data were screened for outliers and data entry errors. Measures of centrality and variation were determined in SPSS (IBM SPSS, 2016). Tests for normal distribution were carried out with Shapiro Wilk (farm scale dataset) and also Kolmogorov Smirnov where $n >$ 100 (field scale dataset). For normally distributed data, univariate ANOVA was carried out to determine significant differences. The Bonferroni post hoc test was used to distinguish significant differences (pairwise comparison). For data that was not normal, non-parametric independent tests were carried out to determine significance. Kruskal Wallis and Mann Whitney U tests were used. The following equations were used to investigate the central hypotheses:

$$\textbf{Farm scale}: \hat{Y}1 \;\&\; \hat{Y}2 \;= \beta o \,+\, \beta 1X1 \,+\, \beta 1X2 \,+\, \beta 1X3 \,+\, \beta 1X4 \qquad \textbf{Equation 4-3.}$$

where Ŷ1 = Farm scale nutrient balance, Ŷ2 = Farm scale nutrient use efficiency,
X1 = Year, X2 = Farm, X3 = Farm soil Drainage class, X4 = Stocking Rate

$$\textbf{Field scale}: \hat{Y}3 \;\&\; \hat{Y}4 \;= \beta o \,+\, \beta 1X1 \,+\, \beta 1X2 \,+\, \beta 1X3 \,+\, \beta 1X4 \qquad \textbf{Equation 4-4.}$$

where Ŷ3 = Field scale nutrient balance, Ŷ4 =Field scale nutrient use efficiency,
X1 = Year, X2 = Subzone, X3 = Environmental factors, X4 = Management factors.

4.4 Results

4.4.1 Farm scale P cycle

Farm scale P flow variables (imports, exports and balances) are shown in Table 4-3.

4.4.1.1 Imports

Mean import of P via inorganic fertiliser was 12 kg P ha^{-1} (CV 55%), and P import via concentrate feed was 8 kg P ha^{-1} (CV 44%), Table 4-3. They each made up 60 and 40% of total P import, respectively, at this scale. Inorganic fertiliser P increased in 2016 but decreased again in 2017, and overall, ranged from 1 to 31 kg P ha^{-1}. Similarly, concentrate feed increased in 2016, but didn't decrease in 2017. Values for concentrate feed ranged from 1 to 15 kg P ha^{-1}.

4.4.1.2 Exports

Mean export of P via milk and animals sold was 9 and 1 kg P ha^{-1} (CV 30% and 133%, respectively). P export, via milk sold, increased slightly between 2016 and 2017, and ranged from 4 to 15 kg P ha^{-1} overall. P export via animals sold, decreased between 2016 and 2017, and ranged from 0 to 3 kg P ha^{-1} overall. P exported via milk sold made up 90% of the export total, and P export via animals sold only made up 10%.

4.4.1.3 Balances

Mean farm scale PBAL was 9 kg P ha^{-1} (CV 58%), Table 4-3. PBAL increased in 2016 but decreased again in 2017, and ranged from 0 to 25 kg P ha^{-1} overall. Mean farm scale PUE was 53% (CV 33%) after three years. PUE decreased in 2016, and increased again in 2017. PUE values ranged from 27 to 97%. Overall, the variation in PBAL and PUE between farms, indicate large differences in P management at this level. Imports of P were predominantly driven by inorganic fertiliser P which showed larger variation (CV 55%) compared to milk sold (CV 33%), which was the main driver of P exports.

4.4.2 Field scale P cycle

Field scale P flow variables (imports, exports and balances) are shown in Table 4-3.

4.4.2.1 Imports

At field scale, a total of 389 fields had sufficient data for full analysis. P import, via inorganic fertiliser, dominated the total imports (44%), followed by concentrate feed (34%) and organic fertiliser (22%). Mean P import via inorganic fertiliser was 14 kg P ha^{-1} (CV 93%), which had a large range from 0 to 67 kg P ha^{-1}. Mean P import via concentrate feed and organic fertiliser was 11 and 7 kg P ha^{-1} (CV 90 and 111%), respectively. P import in concentrate feed ranged from 0 to 86, and P in organic fertiliser ranged from 0 to 44 kg P ha^{-1}.

4.4.2.2 Exports

The largest export of P was via milk sold, making up 70% of the total P export, whereas animals sold, and silage made up 15% each. Mean P export in milk sold was 19 kg P ha^{-1} (CV 80%) after three years and ranged from 0 to 114 kg P ha^{-1}. Export of P via silage and animals sold, both had a mean of 4 kg P ha^{-1}, but variation levels were different (CV 180% and 79%, respectively). P export values for silage, ranged from 0 to 55 kg P ha^{-1} and for animals sold, from to 0 to 22 kg P ha^{-1}.

4.4.2.3 Balances

Mean field scale PBAL was 6 kg P ha^{-1} after three years, and there was a substantial amount of variation associated with it (CV 294%). P balance values ranged from -50 to 64 kg P ha^{-1}. Mean PBAL was highest in 2015, decreased in 2016, and increased again in 2017 (Table 4-3). Mean field scale PUE was 100% after three years, and had smaller variation (CV 74%) compared to PBAL. Mean PUE values ranged from 0 to 750%. Mean PUE was highest in 2016, but slightly lower in 2015 and 2017 (Table 4-3). Field scale analysis of P flows highlights the substantial variation associated with each aspect (imports, exports and balances) of the P cycle. For example, the resultant variation in PBAL and PUE were likely driven by the high variation in the imports and exports of P that were also found. As was found with the farm scale, the field scale P cycle was largely driven by inorganic fertiliser, but also concentrate P imports, and export of P via milk sold (Table 4-3).

4.4.2.4 Difference in P use between scales

PBAL was significantly higher at farm scale compared to field scale ($p < 0.05$) (Table 4-3 and Figure 4-1). There was also higher variability in PBAL at field scale (CV 294%), compared to farm scale (CV 58%). PUE at farm scale was significantly lower compared to field scale ($p < 0.05$) (Table 4-3 and Figure 4-1). Variability in PUE was higher at field scale (CV 74%) compared to farm scale (CV 33%). This variation observed at farm and field scale for both PBAL and PUE, suggests there is significant difficulty in managing the P cycle at both these levels. Addressing this variability will likely require optimising the efficiency of the main drivers of P flow; inorganic fertiliser P, concentrate feed levels, and milk production.

Table 4-3. Mean, standard deviation, minimum, maximum and coefficient of variation for phosphorous imports (inorganic fertiliser, organic manure, and concentrate feed), exports (milk sold, animals sold, and silage), balance, and use efficiency for farm (n = 15) and field (n = 389) scale between 2015 and 2017

Scale	Variable	2015	2016	2017	Mean	SD	Min	Max	CV
					Imports (kg ha^{-1})				
Farm (n = 15)	Inorganic fert.	11	13	11	12	6	1	31	55%
	Conc. feed	7	8	8	8	3	1	15	44%
					Exports (kg ha^{-1})				
	Milk sold	9	9	10	9	3	4	15	30%
	Animals sold	1	1	0	1	1	0	3	133%
	BAL	9	11	8	9	6	0	25	58%
	UE	52%	49%	56%	52%	17%	27%	97%	33%
					Imports (kg ha^{-1})				
Field (n = 389)	Inorganic fert.	14	12	17	14	13	0	67	93%
	Organic manure	8	7	7	7	8	0	44	111%
	Conc. feed	10	11	12	11	10	0	86	90%
					Exports (kg ha^{-1})				
	Milk sold	18	19	21	19	15	0	114	80%
	Animals sold	4	4	4	4	3	0	22	79%
	Silage	3	4	5	4	8	0	55	180%
	BAL	7	4	6	6	17	-50	64	294%
	UE	98%	102%	99%	100%	73%	0%	750%	74%

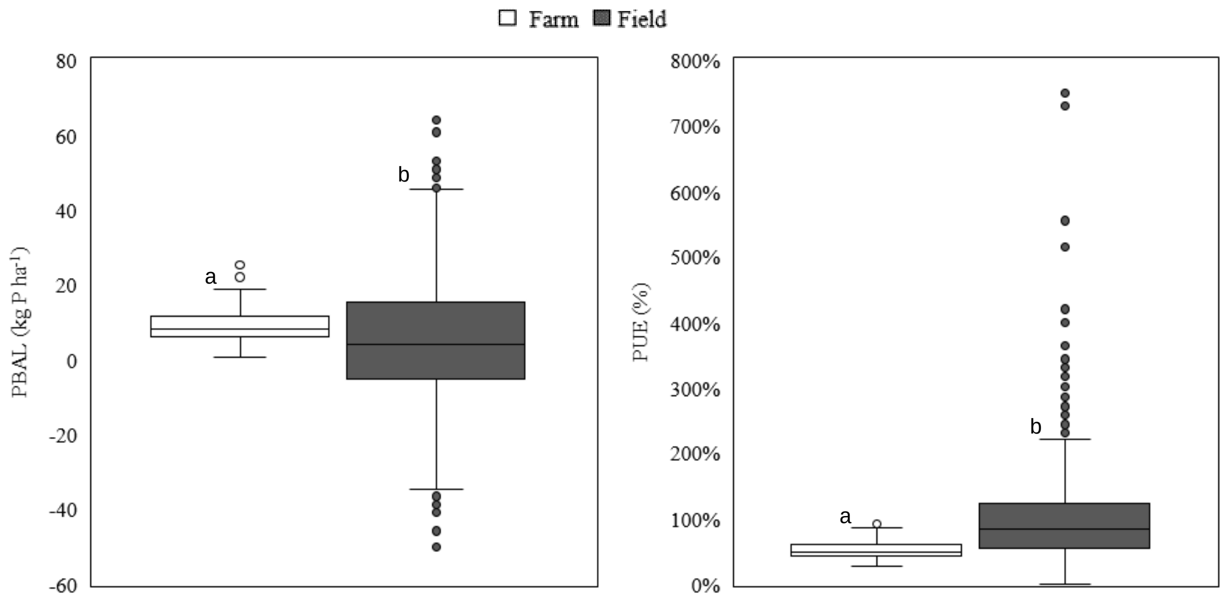

Figure 4-1. Mean phosphorus balance and use efficiency at farm (*n* = 15) and field scale (*n* = 446) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post-hoc test).

4.4.3 Farm scale K cycle

Farm scale K flow variables (imports, exports and balances) are shown in Table 4-4.

4.4.3.1 Imports

At the end of the study, mean import of inorganic fertiliser K was 23 kg K ha^{-1} (CV 83%) and K imported via concentrate feed was 16 kg K ha^{-1} (CV 44%). Inorganic fertiliser K made up 59% of the total K imports and K in concentrate feed made up 41%. Mean inorganic fertiliser K decreased between 2015 and 2016, but increased again in 2017, overall, values ranged from 0 to 67 kg K ha^{-1}. K imported via concentrate feed didn't change substantially between years but overall, values ranged from 1 to 31 kg K ha^{-1}.

4.4.3.2 Exports

K export via animals sold was practically negligible. Mean K in milk and animals sold was 15 and 0 kg K ha^{-1} (CV 30 and 24%), respectively. K exported via milk varied slightly between years, ranging from 6 to 25 kg K ha^{-1}. K exported via animals sold didn't change substantially between years either and values had a very small range (0 to 1 kg K ha^{-1}).

4.4.3.3 Balances

Mean farm scale KBAL was 23 kg K ha^{-1} (CV 86%), after three years. Farm scale KBAL was highest in 2015 and 2017 and slightly lower in 2016 (Table 4-4). For farm scale KBAL, values ranged from -7 to 72 kg K ha^{-1} overall. Mean farm scale KUE was 56% (CV 76%). Farm scale KUE was highest in 2016 and lowest in 2015 and 2017 (Table 4-4), and values ranged from 10 to 270% overall. Farm scale K flows, were primarily driven by K in inorganic fertiliser, and concentrate feed, as well as the K exported via milk sold.

4.4.4 Field scale K cycle

Field scale K flow variables (imports, exports and balances) are shown in Table 4-4.

4.4.4.1 Imports

Mean inorganic fertiliser K was 30 kg K ha^{-1} (CV 111%) after three years, but only made up 29% of the total K imports at field scale. The import of inorganic fertiliser K decrease in 2016, but increased again in 2017, and overall, the values ranged from 0 to 175 kg K ha^{-1}. Mean import of K, via organic manure was higher in comparison to inorganic fertiliser, at 50 kg K ha^{-1} (CV 110%), after three years showed a larger range also, 0 to 308 kg K ha^{-1}. K import via organic manure made up 49% of total K import. Mean import of K via concentrate feed was 22 kg K ha^{-1} (CV 90%) and made up 22% of the total K import at this scale. The mean K import per year, via concentrate feed, didn't vary substantially, but ranged from 0 to 171 kg K ha^{-1}.

4.4.4.2 Exports

Mean export of K, via milk and silage, were 32 and 31 kg K ha^{-1} (CV 80 and 179%), and made up 50 and 48% of the total K export at field scale. K export, via milk and silage, both increased in 2016 and increased again in 2017. K export via milk ranged from 0 to 189 kg K ha^{-1}, and via silage ranged from 0 to 396 kg K ha^{-1} at this scale. Mean export of K via animals sold was almost negligible (1 kg K ha^{-1}, CV 81%).

4.4.4.3 Balances

Mean field scale KBAL was 39 kg K ha^{-1} (CV 179%) at the end of the study. Mean KBAL was highest in 2015, but decreased in 2016 and 2017, overall, values ranged from -203 to 282 kg K ha^{-1}. Mean field scale KUE was 91% (CV 127%), and ranged from 0 to 1130%. Mean KUE increased between 2015 and 2016, but decreased again in 2017.

4.4.4.4 Difference in K use between scales

KBAL was not significantly different between farm and field scale ($p > 0.05$). There was higher variability in KBAL at field scale (CV 179%) compared to farm scale (CV 86%) however. KUE wasn't significantly different between scales either ($p > 0.05$). Variability in KUE was higher at field scale (CV 127%) compared to farm scale (CV 76%).

The main drivers of K flow were the same for both scales (K in inorganic fertiliser, concentrate feed and milk sold), however at field scale, K export via silage was an additional, and large factor, that contributed to KBAL and KUE. With the exception of variation in K export via silage (CV 179%), the variation was greatest in K imports via inorganic fertiliser, for both farm and field scale (CV 83% and 11%, respectively). Field scale analysis also revealed the large contribution that K import via organic manure contributes to the K cycle.

Table 4-4. Mean, standard deviation, minimum, maximum and coefficient of variation for potassium imports (inorganic fertiliser, organic manure, and concentrate feed), exports (milk sold, animals sold, and silage), balance, and use efficiency for farm (n = 15) and field (n = 389) scale between 2015 and 2017

Scale	Variable	2015	2016	2017	Mean	SD	Min	Max	CV
					Imports (kg ha^{-1})				
Farm (n = 15)	Inorganic fert.	26	18	24	23	19	0	67	83%
	Conc. feed	15	17	16	16	7	1	31	44%
					Exports (kg ha^{-1})				
	Milk sold	14	15	16	15	5	6	25	30%
	Animals sold	0	0	0	0	0	0	1	24%
	BAL	26	19	23	23	20	-7	72	86%
	UE	46%	71%	52%	56%	43%	10%	270%	76%
					Imports (kg ha^{-1})				
Field (n = 389)	Inorganic fert.	35	22	33	30	33	0	175	111%
	Organic manure	52	48	50	50	55	0	308	110%
	Conc. feed	21	23	23	22	20	0	171	90%
					Exports (kg ha^{-1})				
	Milk sold	29	32	35	32	26	0	189	80%
	Animals sold	1	1	1	1	1	0	4	81%
	Silage	26	28	38	31	55	0	396	179%
	BAL	51	32	32	39	69	-203	282	179%
	UE	78%	104%	90%	91%	116%	0%	1130%	127%

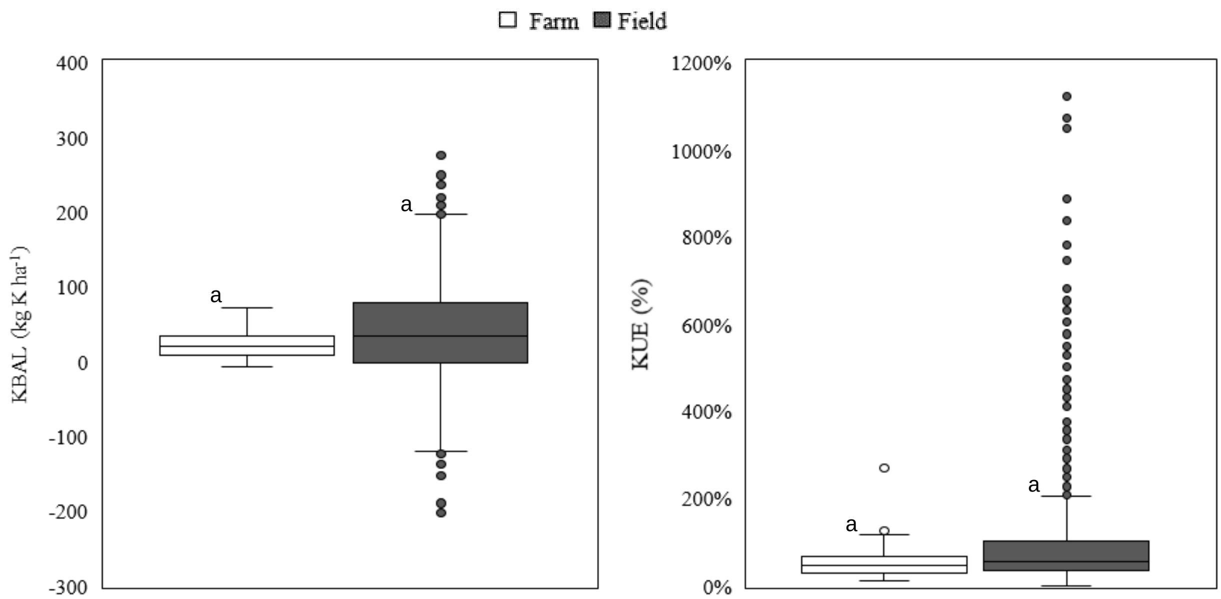

Figure 4-2. Mean potassium balance and use efficiency at farm (n = 15) and field scale (n = 446) between 2015 and 2017. Differences in letters represent significant differences between groups (p < 0.05, Bonferroni post-hoc test).

4.4.5 Farm scale factors affecting P and K balance and use efficiency

Mean farm scale PBAL and PUE, were not significantly different between years ($p > 0.05$). Mean PBAL and PUE were both significantly different between farms however ($p < 0.05$) (Table 4-3). Farm scale PBAL and PUE were not found to be significantly impacted by farm level drainage class differences ($p > 0.05$) or stocking rate differences either ($p > 0.05$).

Mean farm scale KBAL and KUE followed the same trend as PBAL and PUE. Significant differences were found between farms ($p < 0.05$), but not years, farm level drainage class, or stocking rate ($p > 0.05$).

4.4.6 Differences between subzones

4.4.6.1 Phosphorus flows between subzones

Mean PBAL, on the Dairy subzone, was significantly lower than mean PBAL on both the Non-Dairy, and Silage subzones ($p < 0.05$) (Table 4-5). Mean PUE, was significantly higher on the Dairy subzone, than the Non-Dairy and Silage subzones ($p < 0.05$) (Table 4-5). The spatial distribution of mean PBAL and PUE for one farm, is shown in Figure 4-3. Total herbage accumulation per subzone was described in Chapter Three. In summary; Dairy subzone fields had the highest mean grazing and lowest silage yield (10,529 and 2,852 kg DM ha^{-1}, respectively); Non-Dairy had a slightly lower grazing yield but a higher mean silage yield (8,170 and 3,387 kg DM ha^{-1}); and Silage subzone had the lowest grazing yield but highest mean silage yield (5,405 and 6,892 kg DM ha^{-1}, respectively).

Table 4-5. Mean, standard deviation, minimum, maximum and coefficient of variation for phosphorous imports (inorganic fertiliser, organic manure, and concentrate feed), exports (milk sold, animals sold, and silage), balance, use efficiency, and herbage accumulation at field scale for Dairy paddocks (*n* = 354), Non-Dairy (*n* = 20), and Silage (*n* = 15) fields between 2015 and 2017

	Variable	2015	2016	2017	Mean	SD	Min	Max	CV
Dairy (*n* = 354)					Imports (kg ha^{-1})				
	Inorganic fert.	14	12	17	14	10	0	52	74%
	Organic manure	7	7	7	7	6	0	24	85%
	Conc. feed	11	13	13	12	9	2	69	75%
					Exports (kg ha^{-1})				
	Animals sold	4	4	4	4	3	0	17	66%
	Milk sold	19	21	23	21	14	2	93	65%
	Silage	2	3	4	3	4	0	24	132%
	PBAL	7	3	5	5	13	-35	43	272%
	PUE	101%	107%	104%	104%	57%	20%	451%	55%
Non-Dairy (*n* = 20)					Imports (kg ha^{-1})				
	Inorganic fert.	16	15	17	16	15	0	43	91%
	Organic manure	6	6	8	7	6	0	17	91%
					Exports (kg ha^{-1})				
	Animals sold	1	1	1	1	1	0	5	124%
	Silage	8	8	10	8	8	0	25	96%
	PBAL	13	12	14	13	13	-9	53	107%
	PUE	54%	44%	42%	47%	46%	1%	208%	98%
Silage (*n* = 15)					Imports (kg ha^{-1})				
	Inorganic fert.	18	22	26	22	13	5	44	60%
	Organic manure	16	20	15	17	7	0	25	39%
					Exports (kg ha^{-1})				
	Silage	25	24	22	24	9	11	48	38%
	PBAL	9	17	20	15	11	2	30	72%
	PUE	87%	74%	63%	74%	32%	38%	153%	43%

On Dairy subzone paddocks, import of P via inorganic fertiliser was the largest mean contributor (14 kg P ha^{-1}, CV 74%), making up 42% of the total, however P import via concentrate feed was substantial also (12 kg P ha^{-1}, CV 75%), making up 36% of the total. Import of P via organic manure, was the smallest contributor (7 kg P ha^{-1}, CV 85%), making up 22%. Export of P, on Dairy paddocks, was almost completely driven by P exported via milk sold (21 kg P ha^{-1}, CV 65%), and contributed to 75% of the total P export. Export of P via silage and animals sold, was much smaller in comparison (3 and 4 kg P ha^{-1}, CV 132 and 66%), making up 11 and 14% of the total.

On Non-Dairy paddocks and fields, P import via inorganic fertiliser (16 kg P ha^{-1}, CV 91%), was higher, compared to Dairy paddocks, but P import via organic manures was almost the same (7 kg P ha^{-1}, CV 91%). They each made up 70 and 30% of the total. Exports of P, on Non-Dairy fields, were driven almost entirely via silage harvests, (8 kg P ha^{-1}, CV 96%), which contributed to 88% of the total. Export of P via animals sold made up the remaining 11% (1 kg P ha^{-1}, CV 124%).

Import of P, via inorganic fertiliser and organic manure, were the largest on the Silage subzone, compared to any other subzone (22 and 17 kg P ha^{-1}, CV 60% and 39%), and made up 56 and 44% of the total. Although the only recorded export of P on silage fields, mean export of P via silage harvest (24 kg P ha^{-1}, CV 34%), was still large relative to fertiliser imports.

Although PBAL was lowest on the Dairy subzone, variation on these fields (CV 272%), was actually highest compared to Non-Dairy fields (CV 107%) and Silage fields, which had the lowest variation (CV 72%). Silage fields also had the lowest variation in mean PUE (CV 43%). The Dairy subzone had less variation in PUE (CV 55%), compared to the Non-Dairy subzone (CV 98%). Overall, the P cycle in the Dairy subzone was driven predominantly by P imports via inorganic fertiliser, and concentrate feed, and P exports via milk sold. For the Non-Dairy and Silage subzones, the P cycle was driven by imports via inorganic fertiliser P, and P export via silage. However, P import via organic manures, had a larger contribution to the P cycle on the silage subzone, than any other subzone. Results indicate that P exports via silage and milk sold, contribution substantially to PUE at the field scale.

4.4.6.2 Potassium flows between subzones

Mean KBAL on the Dairy subzone, was significantly higher than the Non-Dairy subzone ($p < 0.05$) (Table 4-6). Mean KUE was significantly lower for the Dairy subzone than the Silage subzone ($p < 0.05$). Field scale spatial distribution of mean KBAL and KUE for one farm are shown in Figure 4-3.

Table 4-6. Mean, standard deviation, minimum, maximum and coefficient of variation for potassium imports (inorganic fertiliser, organic manure, and concentrate feed), exports (milk sold, animals sold, and silage), balance, use efficiency, and herbage accumulation at field scale for Dairy paddocks (*n* = 354), Non-Dairy (*n* = 20), and Silage (*n* = 15) fields between 2015 and 2017

	Variable	2015	2016	2017	Mean	SD	Min	Max	CV
Dairy (*n* = 354)					Imports (kg ha^{-1})				
	Inorganic fert.	34	21	31	28	23	0	121	81%
	Organic manure	50	45	47	47	40	0	169	85%
	Conc. feed	23	25	26	25	18	3	137	74%
					Exports (kg ha^{-1})				
	Animals sold	1	1	1	1	1	0	3	66%
	Milk sold	32	35	39	35	23	4	155	65%
	Silage	18	20	30	22	30	0	192	134%
	KBAL	56	36	34	41	48	-87	195	115%
	KUE	77%	100%	89%	89%	77%	12%	539%	87%
Non-Dairy (*n* = 20)					Imports (kg ha^{-1})				
	Inorganic fert.	36	8	38	25	23	0	69	92%
	Organic manure	41	39	56	41	41	0	118	99%
					Exports (kg ha^{-1})				
	Animals sold	1	0	0	0	0	0	1	137%
	Silage	59	62	75	60	61	0	206	102%
	KBAL	19	-13	19	7	48	-105	111	677%
	KUE	76%	136%	85%	104%	114%	0%	369%	110%
Silage (*n* = 15)					Imports (kg ha^{-1})				
	Inorganic fert.	67	53	70	63	37	14	134	59%
	Organic manure	111	137	107	118	46	0	174	39%
					Exports (kg ha^{-1})				
	Silage	177	165	160	166	68	84	351	41%
	KBAL	1	26	16	15	59	-128	104	397%
	KUE	109%	107%	91%	102%	37%	55%	167%	36%

On the Dairy subzone, mean imports of K via organic manures were greater than imports via inorganic fertiliser (47 and 28 kg K ha^{-1}, CV 85 and 81%), and these made up 47 and 23% of the total K import. Mean import via concentrate feed (25 kg K ha^{-1}, CV 74%) made up 18% of the total K import. Mean exports of K via milk sold and silage (35 and 22 kg K ha^{-1}, CV 65 and 134%, respectively) made up 60 and 38% of the total. Export of K via animals sold (1 kg K ha^{-1}, CV 66%) made up just 2% of the total.

On the Non-Dairy subzone, imports of K were again greater via organic manures than inorganic fertilisers (41 and 25 kg K ha^{-1}, CV 92 and 99%), and made up 62 and 38% of the total. Mean K export via animals sold was < 1 kg K ha^{-1} overall, and export via silage was 60 kg K ha^{-1} (99%), contributing almost entirely to the total.

For the silage subzone, imports of K via organic manures were again greater than via inorganic fertiliser (118 and 63 kg K ha^{-1}, CV 39 and 59%) and were the largest of all subzones. Organic manures made up 65% of the total K import and inorganic fertilisers made up 35%. Export of K via silage was 166 kg K ha^{-1} (CV 41%). Mean field scale KBAL was highest on the Dairy subzone, however, it had the least variation (CV 115%) compared to the Non-Dairy and Silage subzone (677% and 397%). The substantial variation associated with these two subzones indicates there is a lot overlap in results, and large differences in nutrient management practices. Mean field scale KUE was lowest on the Silage subzone, but had the least variation (CV 36%). Non-Dairy subzone fields had the largest KUE but also the largest variation (110%). Dairy paddocks had the lowest KUE and a slightly lower variation than Non-Dairy fields (87%).

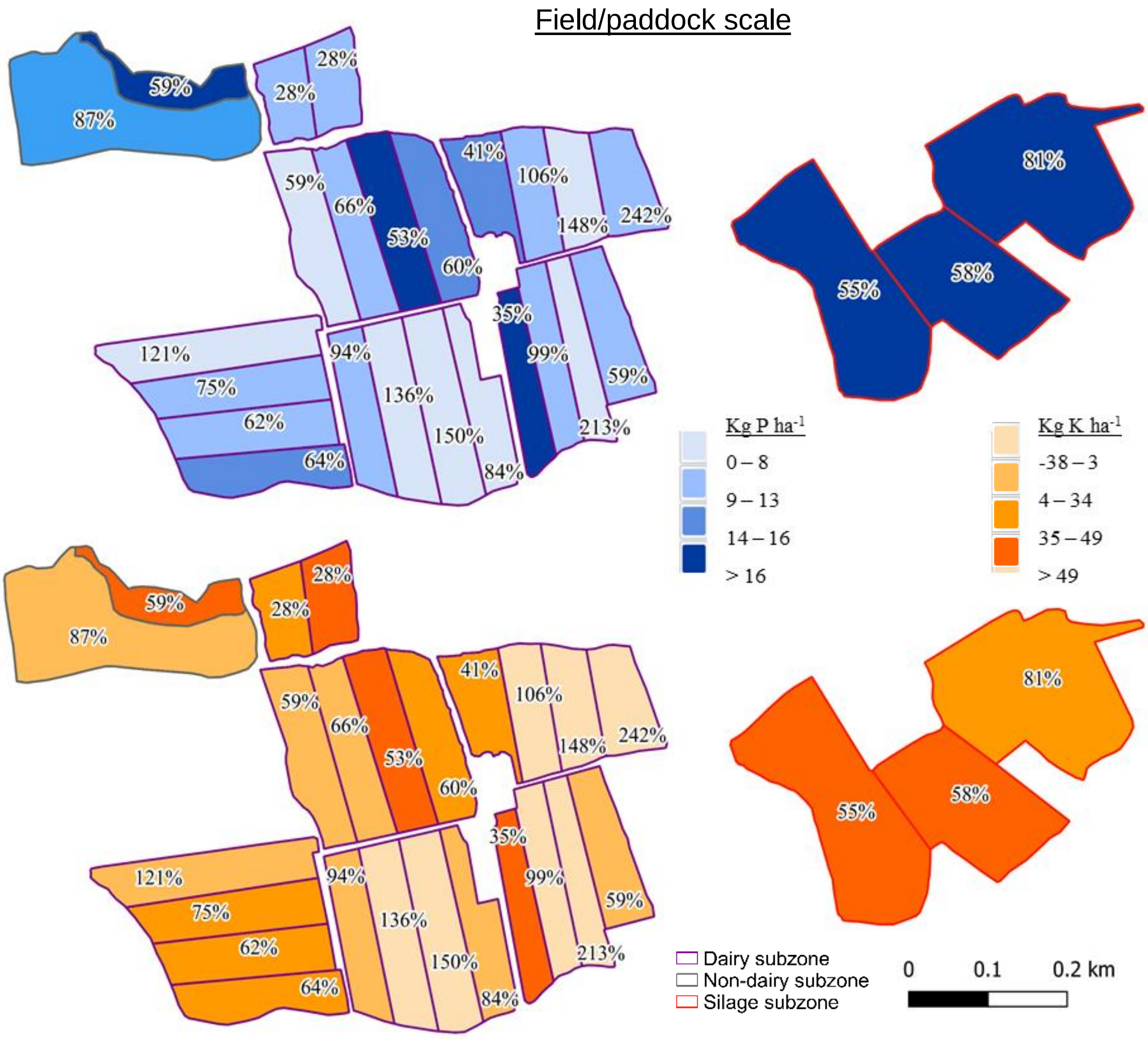

Figure 4-3. Maps of mean phosphorus and potassium balance and use efficiency for one farm at field scale. Dairy subzone is indicated by purple colour field boundaries, Non dairy subzone is indicated by grey colour field boundaries and silage subzone is indicated by red colour field boundaries. Spatial distribution of these phosphorus and potassium management indicators are shown for each individual paddock and field across the whole farm. Mean phosphorus and potassium balance value is indicated by colour differences, darker shades indicate higher values. Mean nitrogen use efficiency value (%) is labelled per field.

4.4.7 Field scale factors affecting P and K balance and use efficiency

Overall, results indicate that both environmental and management factors are significant in contributing to the variability and capacity to manage variability in field scale P and K cycling.

4.4.7.1 Environmental factors

Table 4-7. Phosphorus and potassium balance and use efficiency indicators as affected by environmental factors gradient (gradient or level) and drainage class (well, moderately or poorly drained) at field scale

		Environmental factors		
Level	**PBAL**	**PUE**	**KBAL**	**KUE**
		Gradient		
Gradient	4	107%	37	93%
Level	7	90%	39	88%
Significant	G < L	G > L	N/S	N/S
		Soil drainage class		
WD	4	107%	33	99%
PD	13	72%	55	57%
MD	11	77%	55	67%
Significant	WD < PD & MD	WD > PD & MD	WD < PD	WD < PD and MD

G, Gradient; L, Level, PD, Poorly drained; MD, Moderately drained; WD, Well drained; N/S, Not Significant ($p > 0.05$)

Gradient class significantly impacted field scale P cycling ($p < 0.05$), but not K cycling ($p > 0.05$). Mean PBAL was significantly higher on fields that were level (7 kg P ha^{-1}) compared to fields with a gradient (4 kg P ha^{-1}), and mean PUE was significantly lower on level fields also (91% and 107%, respectively) (Figure 4-4).

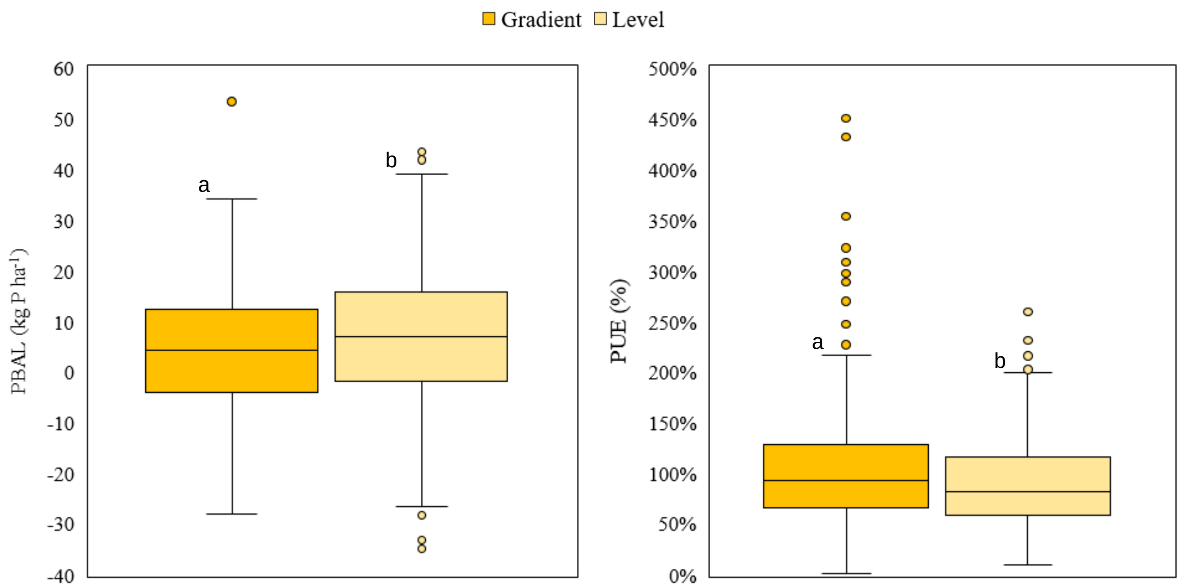

Figure 4-4. Mean phosphorus balance and use efficiency at field scale as affected by gradient class (gradient or level fields) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post hoc test).

Soil drainage class significantly impacted both field scale P and K cycling ($p < 0.05$). Mean PBAL was significantly lower on well drained fields (4 kg P ha^{-1}) compared to moderately drained fields and poorly drained fields (11 and 13 kg P ha^{-1}, respectively) (Figure 4-5).

Mean PUE was significantly higher on well drained fields (107%) compared to moderately and poorly drained (78 and 72%, respectively) (Figure 4-5).

For K cycling, mean KBAL was significantly lower on well drained fields compared to poorly drained fields (34 and 55 kg K ha^{-1}, respectively) (Figure 4-6). Mean KUE was significantly higher on well drained fields (98%) compared to both moderately and poorly drained fields (68 and 57%, respectively) (Figure 4-6).

For field scale environmental factors overall, nutrient P and K cycling had significantly lower balances and higher use efficiencies on fields with well drained soils compared to fields with less than well drained soils. Also, fields with a level gradient class had a higher P nutrient balance and a lower P nutrient use efficiency compared to fields with a gradient.

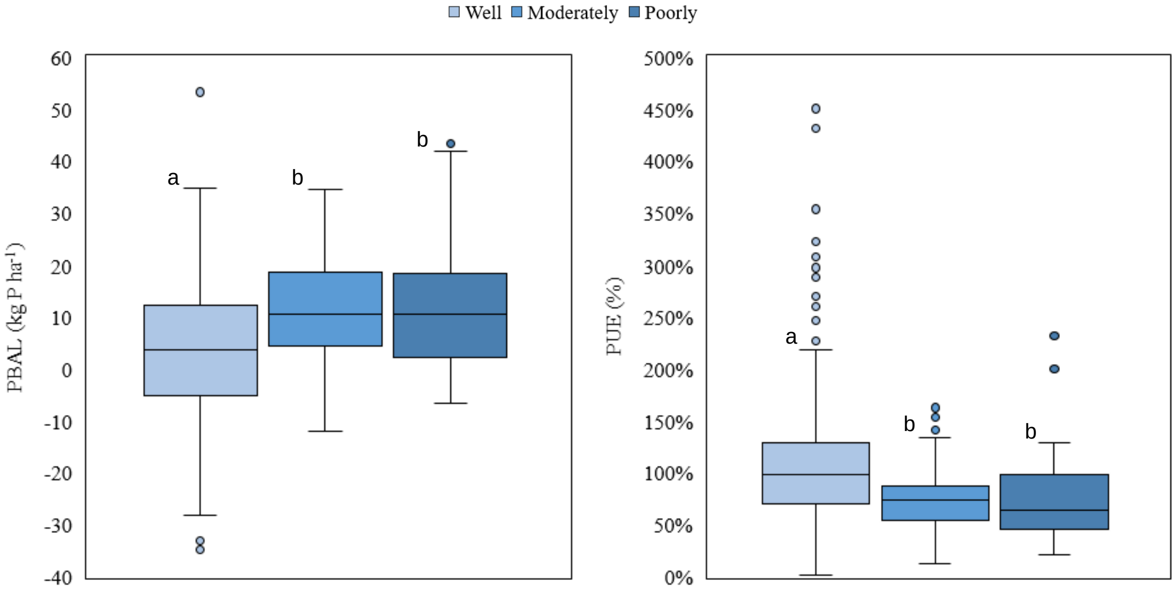

Figure 4-5. Mean phosphorous balance and use efficiency at field scale as affected by soil drainage class (well, moderate or poorly drained fields) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post hoc test).

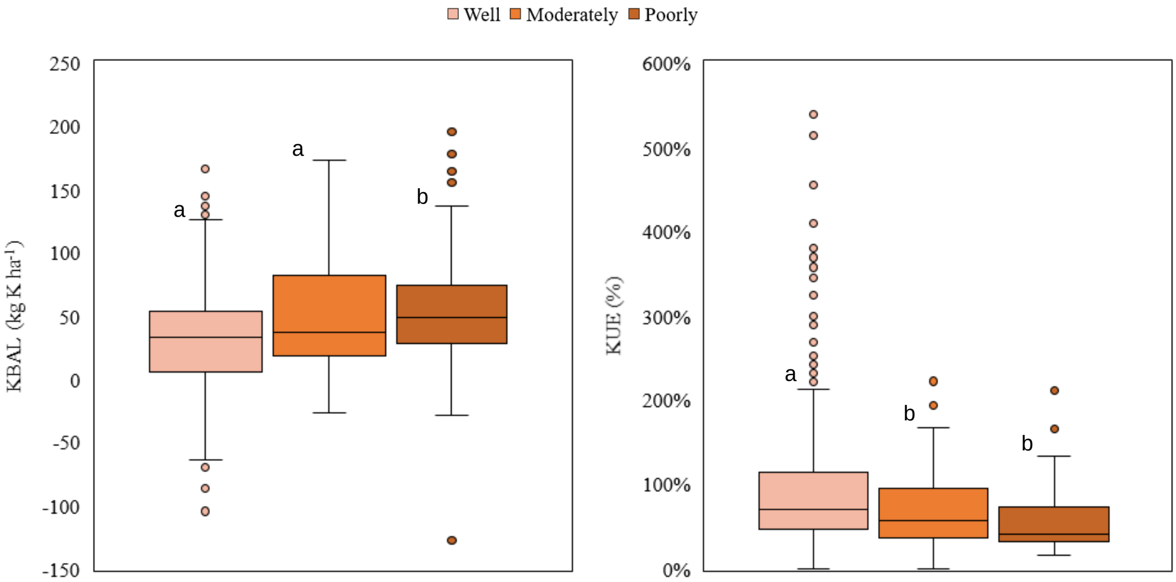

Figure 4-6. Mean potassium balance and use efficiency at field scale as affected by soil drainage class (well, moderate or poorly drained fields) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post hoc test).

4.4.7.2 Management factors

Table 4-8. Phosphorus and potassium balance and use efficiency indicators as affected by management factors species composition (perennial ryegrass high, medium or low, white clover absent or present), cropping (grazing only, grazing + harvest or 2 cut harvest), and time since reseeded (< 5 years, 5 to 10 years, or greater than 10 years) at field scale

		Management factors		
Level	**PBAL**	**PUE**	**KBAL**	**KUE**
		PRG		
Low PRG	10	93%	40	94%
Med PRG	5	103%	39	90%
High PRG	3	99%	39	87%
Significant	High < Low	Low < High	N/S	N/S
		WC		
Absent WC	6	98%	45	87%
Present WC	5	101%	34	92%
Significant	N/S	N/S	Absent < Present	N/S
		Cropping		
GO	7	93%	56	58%
G + H	2	124%	-1	172%
2CH	18	68%	31	95%
Significant	G+1 < GO & 2CS & GO < 2CS	2CS < G+1 GO < G+1	G+1 < GO	GO < 2CS & G+1
		Reseed		
Less than 5 yrs	12	93%	46	104%
5 to 10 yrs	3	117%	42	94%
Greater 10 yrs	6	90%	43	73%
Significant	Greater 10 yrs < Less than 5 yrs	Less than 5yrs < 5 to 10 yrs Greater than 10yrs < 5 to 10 yrs	N/S	N/S

PRG, Perennial Ryegrass; WC, White Clover; G+H, Grazing + Harvest; GO, Grazing Only; 2CS, 2 Cut Harvest; Yrs, years

Sward composition differences for Perennial Ryegrass (PRG) impacted P cycling ($p < 0.05$) but not K cycling ($p > 0.05$) at field scale. Mean PBAL was significantly lower, and PUE significantly higher, for fields with a High proportion of PRG (3 kg P ha^{-1} and 99%) in the sward, compared to fields with a Low proportion of PRG in the sward (10 kg P ha^{-1} and 93%).

Differences in sward composition of White clover (WC) did not significantly impact P cycling ($p < 0.05$), but it did however impact K cycling ($p < 0.05$). Mean KBAL was significantly lower where WC was Absent in a field (34 kg K ha^{-1}), compared to where it was Present (45 kg K ha^{-1}). Mean KUE was not significantly impacted by WC composition ($p > 0.05$).

Field scale P and K cycling were impacted by differences in cropping rate ($p < 0.05$). Mean PBAL was significantly lower for Grazing + Harvest (2 kg P ha^{-1}) compared to Grazing Only and 2 Cut Harvest (7 and 18 kg P ha^{-1}, respectively) (Figure 4-8). Also, mean PBAL on Grazing Only was significantly lower than 2 Cut Harvest ($p < 0.05$). Mean PUE was higher for Grazing + Harvest (124%), than Grazing Only, and 2 Cut Harvest (93 and 68%) (Figure 4-8).

Mean KBAL, was significantly lower for Grazing + Harvest, than Grazing Only (-1 and 56 kg K ha^{-1}, respectively) (Figure 4-9). Mean KUE was significantly lower for Grazing Only (58%) than Grazing + Harvest, and 2 Cut Harvest (172 and 95%) (Figure 4-9).

Differences in reseed interval significantly impacted P cycling ($p < 0.05$) but not K cycling ($p > 0.5$). Mean PBAL was significantly lower on fields reseeded 5 to 10 years ago (3 kg P ha^{-1}), compared to fields reseeded less than 5 years ago (12 kg P ha^{-1}) (Figure 4-10). Mean PUE was significantly higher on fields reseeded 5 to 10 years ago (117%),m1<,n\n\\zm1 compared to fields reseeded less than 5 years ago (93%), and also fields reseeded longer than 10 years ago (90%) (Figure 4-10).

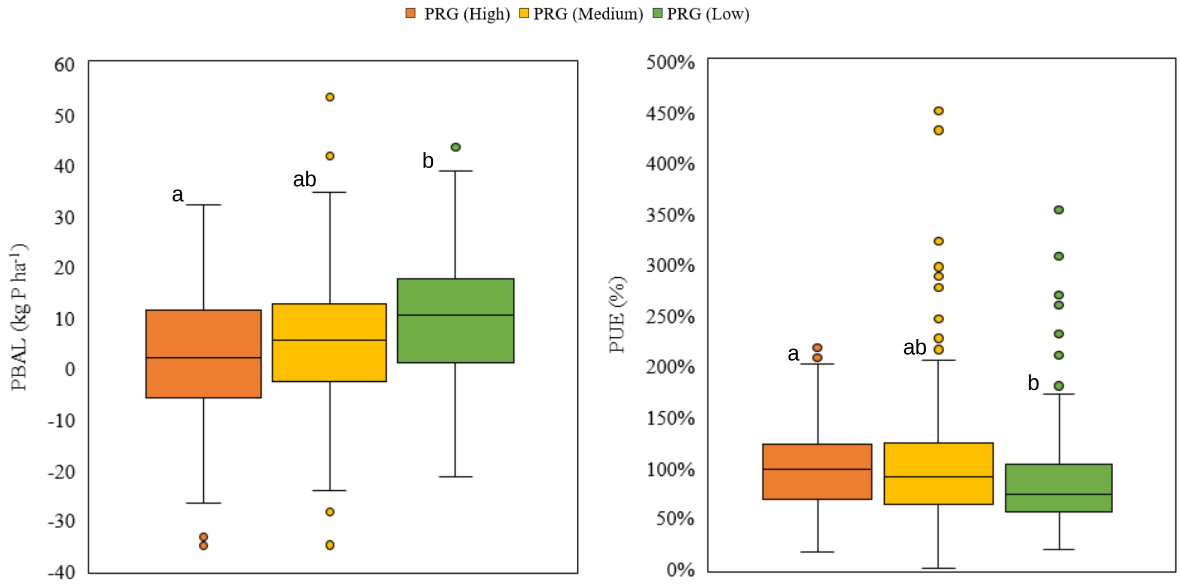

Figure 4-7. Mean phosphorous balance and use efficiency at field scale as affected by perennial ryegrass class (high, medium or low sward level) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post hoc test).

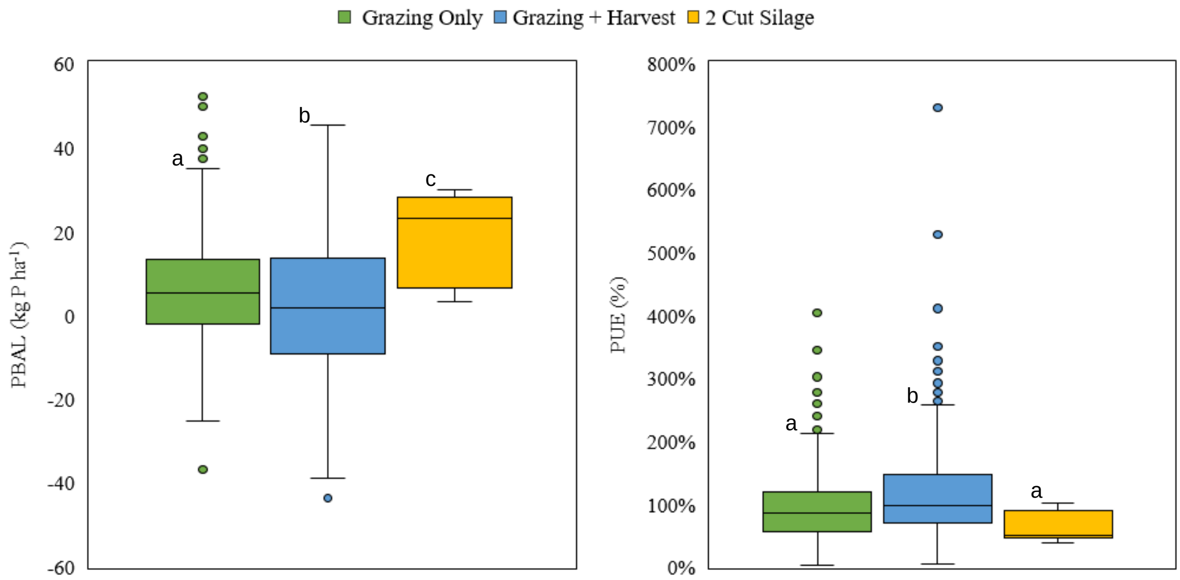

Figure 4-8. Mean phosphorus balance and use efficiency at field scale as affected by cropping strategy (grazing only, grazing + harvest or 2 cut silage/harvest) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post hoc test).

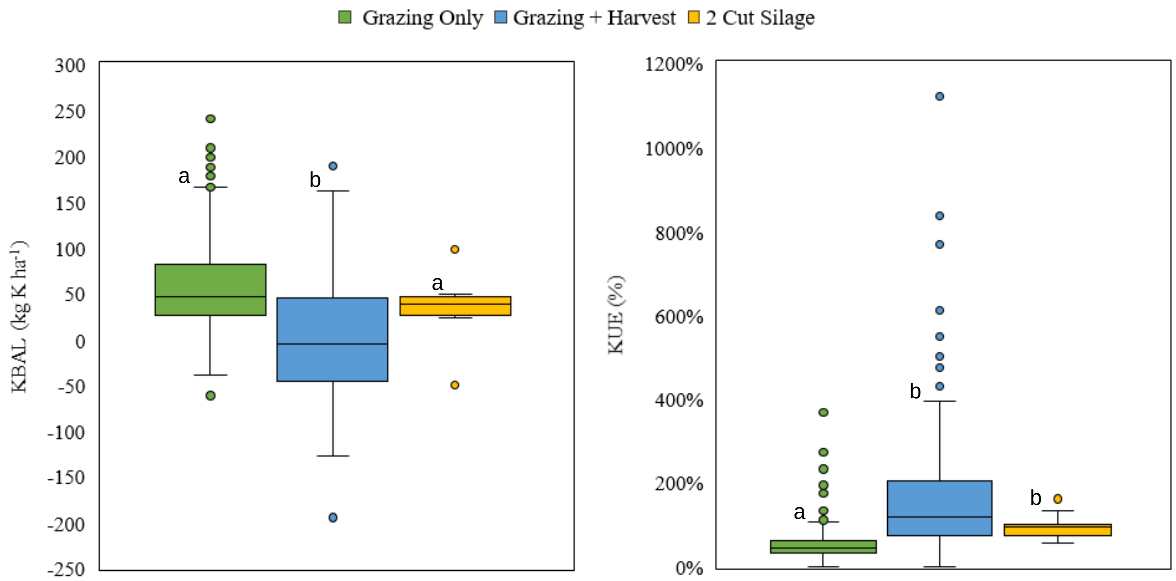

Figure 4-9. Mean potassium balance and use efficiency at field scale as affected by cropping strategy (grazing only, grazing + harvest or 2 cut silage/harvest) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post hoc test).

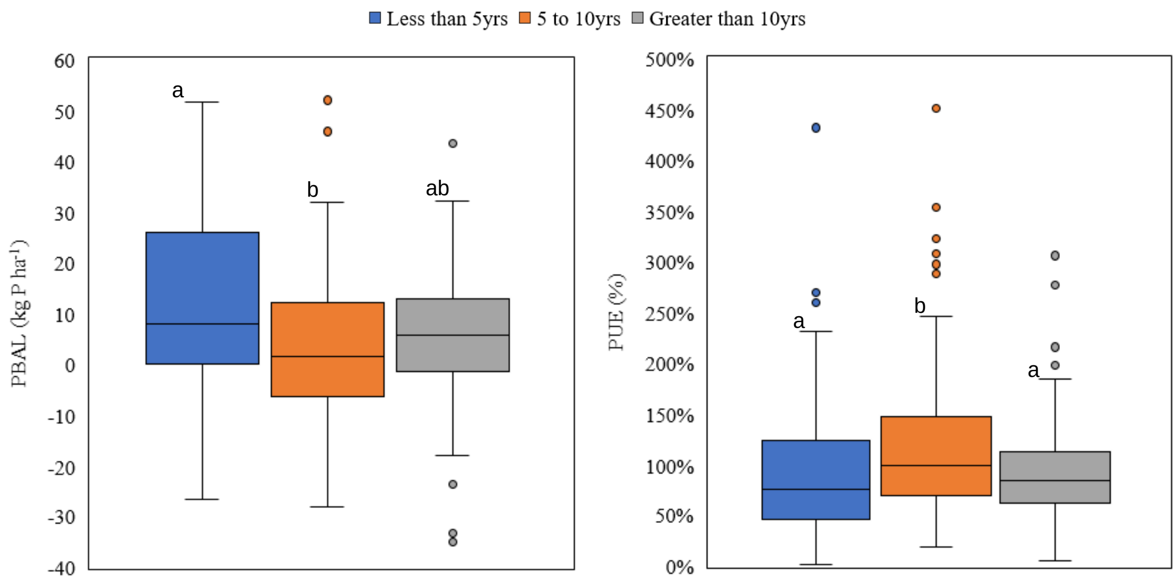

Figure 4-10. Mean phosphorus balance and use efficiency at field scale as affected by time since reseeded (< 5 years, 5 - 10 years or > 10 years) between 2015 and 2017. Differences in letters represent significant differences between groups ($p < 0.05$, Bonferroni post hoc test).

4.5 Discussion

4.5.1 Variation in drivers of nutrient management at both scales

Nutrient imports, via inorganic fertilisers and exports via milk sold, were the main drivers for P and K cycling between both farm and field scales, similar to the findings for Nitrogen cycling in Chapter Three. Large variation in nutrient management (both imports and exports), as shown in this study Table 4-3 and Table 4-4, were also highlighted by others who have conducted farm and field scale studies (Buckley et al., 2016b; McDonald et al., 2019; Mihailescu et al., 2015; Van Leeuwen et al., 2019). Decisions relating to nutrient imports likely contributed to the variation found in P and K nutrient indicators. Therefore, results indicate that efforts are needed to optimise nutrient management at field scale on grazed grass-based dairy systems. As explored in Chapter Five, focussing on soil fertility may be a more practicable approach for farmers, but this approach does at least provided insight for researchers as to where the pinch points are in the P and K cycle on grass based dairy farms. At field scale, targeted measures could be adopted, for example, to adjust for the substantial nutrient import to the P cycle via P in concentrate feed (Table 4-3) or for the K cycle, via silage harvest (Table 4-4). The levels of variation and ultimately, the opportunities to manage them rely on the decision making at field scale and the methodlgoy used to collect the information which may introduce uncertainties

A significantly higher PBAL and lower PUE between farm and field scale, respectively (Figure 4-1), indicates that source level pressure differs between these scales. This has implications for the environmental and nutrient management efficiency of these dairy farms. The difference between a farm scale nutrient balance and a field scale nutrient balance was also found by Adenuga et al., 2018a and Bassanino et al., (2007). Together, these findings highlight the importance of combining farm and field scale nutrient balance approaches, when assessing nutrient management efficiency.

The smaller range in PBAL values at farm level (CV 58%), compared to field (CV 294%), suggests that there is substantially more variability at field scale (Figure 4-1). The number of variables and number of changes that they are subject to greatly add to the difficulty in accurately measuring these nutrient flows and may contribute to the large range in values. This is where P flows interact with grassland management and environmental factors, which is something that farm scale does not reflect. The variation in nutrient balance values, are driven by nutrient management decisions on grazing strategies (inorganic and organic fertiliser applications) as shown by Lanyon, (1994) and Oenema et al., (2011) and is further altered by the field scale characteristics as shown by Bramley, (2009). The variation in PUE at field scale (CV 74%), exceeded that, which was observed at farm scale (CV 33%), and suggests that high levels of PUE are achievable, but also that low levels of PUE are occurring at this scale. This suggests that fields or paddocks units (1.4 to 3ha), which are the existing management unit boundaries on grazed grass-based dairy systems, may be an appropriate scale to reduce variation and optimise nutrient management strategies. Adenuga et al. (2018a), also found that within farm variability of management decision and soil characteristics, was best observed at field scale, and suggests that, where excess nutrient imports occur locally, it may lead to high nutrient loss risk, which is invisible in a farm scale nutrient balance. In reality, farm scale is suited as an indicator of *average* P cycling across the farm, but the significantly lower PBAL at field scale is an insightful indicator of the *actual* P cycling. This difference may also be due to P losses to laneways and soil adsorption for example. Additionally, the limitations of the scale at which indictors are observed at. It also depends on the selected nutrient balance methodology (Buckley et al., 2016b; Oenema et al., 2003). Although there wasn't a significant difference for KBAL and KUE between farm and field scale, the difference in variation, was still larger at field scale (CV 86 and 179% KBAL, 76 and 127% KUE) (Figure 4-2). As with P cycling, the variation of KBAL and KUE, may be best observed at field scale, where actual interactions with environmental and management factors occur, rather than at farm scale, where these factors are not accurately captured and reflected.

The current study didn't find significant differences in farm scale P and K cycling between individual years but there was a significant difference between farms. For P cycling at farm scale, the mean results were higher for PBAL (9 kg P ha^{-1}) compared to other similar farm scale studies. Ruane et al., (2014), found a mean balance of 5.61 kg P ha^{-1} between 2003 and 2006. Buckley et al., (2016b) found a substantial decrease (50%) in PBAL, between 2006 and 2012, on Irish dairy farms, which was 11.9 to 6.0 kg P ha^{-1}. Murphy et al., (2015) found low PBAL levels in the 2009 to 2011 period also (2.4 kg P ha^{-1} on 9 Irish dairy farms), and (Mihailescu et al., 2015) found a mean balance at farm scale of 5.09 kg P ha^{-1} and a higher PUE than this study (70% compared to 52%) from 2009 to 2011. These studies also found significant differences between farms and so were in line with the results of this study. Overall, this indicates there is a variety of nutrient management strategies between farms. On average, the variation in nutrient imports and exports for these studies, was in line with the variation found in this study (Table 4-3). In comparison to these previous farm scale studies, results of this study suggested that mean PBAL had increased at farm scale in the recent years (2015 to 2017). A more recent study by Adenuga et al., (2018a) of dairy farms in Northern Ireland had a similarly high mean PBAL (11 kg P ha^{-1}) to this study. However, if the cohort of farms observed in this study was representative of the national trend, it indicates that there has been an increase in source level pressure at the farm scale and therefore added risk to the environment. This scenario may have been influenced by the national target of a 50% increase in milk production by 2020, as set by policymakers in Ireland (Food Harvest 2020, 2010; FoodWise 2025, 2015) which saw increases dairy cow numbers in the lead up to and after 2015 (CSO, 2019b; NFS, 2017). However, the farms selected for this study were not representative of the national trend, but were an example of highly stocked dairy farms.

One of a few studies that measured K cycling at farm scale (Cela et al., 2014), found a mean KBAL of 29 kg K ha^{-1} on American dairy farms that ranged from -46 to 184 kg K ha^{-1}, which was similar to the results for this study (23 kg K ha^{-1}, ranging from -7 to 72 kg K ha^{-1}). Also, an Australian study of dairy farms (Gourley et al., 2012a) found a median value (no mean value reported) of 74 kg K ha^{-1} that ranged from 13 to 452 kg K ha^{-1}. Although this compares slightly different dairy systems, it highlights the variation that occurs in K cycling at farm scale. An absence of regulation by the EU and lack of significant environmental risk may partially explain why farm scale KUE was 56% farm scale. In Switzerland, a national scale 2008 analysis of potassium in swiss agriculture, found a similar KBAL result (28 kg K ha^{-1}) (Spiess, 2011), which indicates that farm scale K cycling in this study is within the range of the results found in other similar studies. Also, a long-term European study (~30 years), on potassium flows concluded that the recovery rate from inorganic fertilisers never exceeded 62% (Zörb et al., 2014), suggesting that KUE may typically be moderate or low in these agricultural systems.

Farm scale P and K cycling weren't found to be impacted by soil drainage class differences in this study. However, Buckley et al., (2016b), with a larger variety and sample size of farms, found that there was a significant difference between what was described as land that was "average" and "poor" in use potential (PUE levels were lower) compared to land that had "good" land use potential (PUE levels were higher). Van Leeuwen et al., (2019) compared nutrient balances on farms on sandy (homogenously well drained fields) and peat soils (homogenously poorly drained fields). They found that the least variation in nutrient management was on well drained soils. At farm level for this study, the sample was predominately made up of well drained soils and so farms with poorly drained soils may have been underrepresented at farm scale. It is also likely that the farm scale delineation of soil type (1:250,000), may not be sufficient for distinguishing between smaller differences within farm boundaries. Given this, the distribution of soil properties that impact nutrient cycling such as structure, organic matter levels or rooting depth (McDowell and Nash, 2012; Vaclav, 2000; Van Keulen et al., 2000) are not correctly represented by soil maps. Results of this study suggest that farm scale wasn't sufficient for delineating between nutrient cycles on different soil types.

With regard to P cycling and stocking rate, Ruane et al., (2014) didn't find a significant relationship between P cycling and stocking rate, the same as this study. However, a number of other studies did find a significant relationship between the two. For example, Adenuga et al., (2018a), found that PBAL increased by 1.05 kg unit where stocking rate increased by 1 Livestock Unit, using a unique analysis methodology. A similar outcome was found by (Buckley et al., 2015; Halberg et al., 2005) whom both found that nutrient surpluses increase with increased stocking rate. However, overall, the increase in stocking rate can be associated with an increase in inorganic fertiliser and concentrate requirements, but also an increase in milk output, which likely lead to the increase in PBAL, as shown by Adenuga et al., (2018a); Buckley et al., (2015), and Mihailescu et al., (2015). Additionally, a study of dairy farms in Finland, found that the relationship between PBAL and milk output, was due to its association with concentrate feed (Nousiainen and Virtanen, 2005). The relationship between P and K cycling and stocking rate was not significant, but stocking rate is typically a driver of many factors that affected nutrient management including increases in inorganic fertilise use and export via milk which was found in the third year of this study (Table 4-3).

The significant difference in P and K cycling between subzones (Table 4-5 and Table 4-6) further highlights the within farm variability that occurs at paddock and field scale. The main drivers of P and K flows (fertilisers, concentrate feed and exports via milk sold) varied largely between subzones (Table 4-3 and Table 4-4), and as a result contributed to the variation in the indicators. Variation in nutrient management which can be driven by many factors including soil test requirements of individual farms and fields (Buckley et al., 2016b), over application, from lack of nutrient planning (Roberts et al., 2017; Wall et al., 2012) and the mismatch in nutrient ratios between organic manures and crop requirements (Withers et al., 2014; Withers et al., 2015). Also, Gourley et al., (2015) and Powell et al., (2010) showed how distance from the main farm buildings was a significant factor that explained variation in nutrient inputs. Environmental and grassland management decisions, as explained below, also contributed significantly to variation in nutrient balances.

Figure 4-3 shows the field scale spatial distribution for P and K after many of the interactions of these factors have occurred. Such a map is effective in illustrating the variability that a farm manager, advisor, or policy maker has to address, but it is invisible to all of them when approached from farm scale only.

4.5.2 Variation in PBAL and PUE between subzones

Although there was no distinct pattern in Figure 4-3, on average, PBAL was lower and PUE was higher on the dairy subzone or “milking platform” of the farms that were studied. This indicates that nutrient source pressure was higher on to the Non-Dairy and Silage subzones for P. However, in this study, the variability in P cycling was actually greatest on the Dairy subzone (CV 272%), which occupied 57% of the Total Grassland Area on average, while Non-Dairy and Silage subzones had lower variability (CV 107 and 72%), and they occupy a smaller area on average (28 and 15%, respectively) as described in Chapter Three. Therefore, source pressure may be greatest on Non-Dairy and Silage subzones based on the indicators, but to minimise nutrient variability i.e. extreme values, the Dairy subzone should be targeted. Additionally, the Dairy subzone has a higher stocking rate, due to the dairy herd, than either of the other subzones. A higher stocking rate is associated with a higher density of urine and dung patches (Maire et al., 2020; Silva et al., 2000; Stout et al., 1997), which have high loadings of nutrients and are difficult to capture as they are in excess of plant requirements and are in soluble or organic form (Di et al., 1998).

Almost oppositely to P cycling, KBAL and KUE were significantly higher and lower, respectively, on the Dairy subzone compared to the Non-Dairy and Silage subzones (Table 4-6).The K cycle had substantial variation in the Non-Dairy and Silage subzone (CV 377 and 397%), which was greater than the “milking platform” (CV 115%).

Overall, meeting the dual goals of minimising extreme surplus balances for both P and K is difficult, but field scale analysis may be effective in identifying extreme values (surplus or deficit) where data uncertainty is minimised. The use of a tool with a similar output to Figure 4-3, may be effective in monitoring differences between fields in the long term for research purposes. Nutrient management is still best guided by soil fertility due to the higher certainty in data collection These relationships between soil fertility and nutrient balances are explored in Chapter Five. Reducing P surplus would gain important environmental benefits, specifically for water quality. Buckley et al., (2015) highlights how P (and K) “build up” (application of nutrients in excess of plant requirements) is necessary for soil fertility, however, it adds an increase in risk of nutrient transfer to the aquatic environment. A P surplus greater increases the risk of P losses to water, which shows the importance of appropriate source level management at field scale. Furthermore, where the soil in a field or paddock is already P enriched, it can take many years for soil P levels to decrease at a rate proportional to the PBAL of the field (Schulte et al., 2010).

Results of the evaluation of field scale environmental and management factors highlighted how variability in P and K cycling was significantly influenced by controllable management factors (sward species, cropping, and reseed rate), but also uncontrollable factors (gradient, and soil drainage class). Designing a nutrient management approach that achieves the dual targets of high grassland utilisation for milk output, while also accommodating for individual field drainage and gradient factors, remains challenging at the field scale, but could be an important step for future nutrient management strategies.

4.5.3 Environmental factors

Level fields are less likely to have surface run off (overland flow) compared to fields with a gradient, and so there is less risk of P losses from P attached to sediments or direct losses of P in inorganic fertiliser or organic manure (Figure 4-4). This, as a result, may have allowed PBAL to be higher, and PUE to be lower, on level fields. In this case, P source pressure was greater on level fields however, in terms of P loss risk to the aquatic environment, targeted management of fields with a gradient would be more important, which is a major component of the Critical Source Area approach (Doody et al., 2012; McDowell and

Srinivasan, 2009; Sharpley et al., 2011; Shore et al., 2014; Srinivasan and McDowell, 2009). It was expected that gradient class would impact K cycling also, it is possible the range in gradients were not large enough to capture slope differences that may impact K surface run off as K can have a strong bind to soil, depending on clay type and level (Sharpley et al., 1988; Wakeel et al., 2013; Zörb et al., 2014).

Variation in soil drainage class within farm the boundary, and the impact it has on nutrient cycling, was also explored by (Van Beek et al., 2009; Van Leeuwen et al., 2019). It was recommended that for farms with the highest variation in soil drainage class, measuring field scale nutrient balances would help identify extreme surpluses and deficits, as well as estimate the areas of greatest leaching or emission risks. For this study, the nutrient source pressure, and risk of nutrient losses for P and K, was greatest on poorly drained fields and paddocks as they had significantly higher PBAL (Figure 4-5 and Figure 4-6). Poorly drained fields on a farm, are likely the most hydrologically connected to waterways. Nutrient surpluses (high or low) on these fields, therefore are most likely to be subjected to surface run off, erosion and leaching (McDowell et al., 2003; Quinton et al., 2003). Additionally, there is a higher risk of soil compaction (for any drainage class), which can increase P loss via surface run off (Vaclav, 2000), or limit nutrient flow; as it negatively impacts root development, aeration, and ultimately yield**,** which in turn is affecting total nutrient uptake (Ball et al., 2018; Buczko et al., 2018; Correll, 1998; Douglas et al., 2007). Managing nutrients for soil type differences at field scale is still challenging, as the scale at which soil mapping is available is not sufficient, and it is left to the farm manager or advisor to acquire the relevant skills for soil typing. Van Leeuwen et al. (2019) suggested that nutrient management decisions that line up with soil drainage class differences can be a coincidence, this can occur where the fields that are the most accessible are also of the same soil drainage class. Tools are starting to become available for precision management of grassland soils (Higgins et al., 2019) but practice adoption is still a limitation (Creighton et al., 2011; Hyland et al., 2018).

4.5.4 Management factors

The dominance of PRG in swards is due to its characteristic persistence and productivity as a grass plant (McEvoy et al., 2011) compared to other grasses for example Bent, Fescues or Annual. It can be utilised for grazing or silage but it works best on high soil fertility levels (Kennedy and O'Donovan, 2014). As a result, the nutrient demand (P and or K) by PRG was expected to be greater on swards dominated by PRG. This is line with Figure 4-7, showing that PBAL was lower on High PRG fields compared to fields with Low PRG, which would be expected to have a lower nutrient demand. Surprisingly, K cycling was not impacted by differences in PRG levels suggesting that K levels were the same for PRG across each category, or that K demand from PRG was not as substantial as demand was for P. It was expected that WC proportions would have an impact on P cycling but it is likely that PRG was the main driver of P uptake and that WC requirements of P in comparison, were lower, on average (Anjo and Jan, 1997; Egan et al., 2018). WC requires K to sustain sward cover, and also contributes to overall clover production (Egan et al., 2017; Enriquez-Hidalgo et al., 2014)). This is in line with the result that was found in this study. A significantly lower KBAL on fields where WC was absent suggests that the requirement for K was lower compared to fields where WC was present. Overall, these results show that sward composition was an important contributor to the variation in P and K indicators.

Results for this study found that both PBAL and KBAL were significantly lower for Grazing + Harvest fields compared to Grazing Only and 2 Cut Harvest (for P only) Figure 4-8 Figure 4-9. Furthermore, mean PUE was significantly higher for Grazing + Harvest (2 kg P ha^{-1}) than Grazing Only and 2 Cut Harvest (7 and 18 kg P ha^{-1}). This outcome most likely occurred on the Dairy subzone where the combination of nutrient export via milk sold and harvest events, lead to a greater total nutrient export of P and K than Grazing Only or 2 Cut Harvest alone (Table 4-5 and Table 4-6). This also suggests that nutrient source pressure was greater on Grazing Only and 2 Cut Harvest fields or paddocks. (McDonald et al., 2019) also found that Grazing + Harvest was lower than Grazing Only (10.3 and 15.7 kg P ha^{-1}, respectively), however 2 Cut Harvest was substantially lower again (-26.3 kg P ha^{-1}), in a catchment scale study across multiple farm types, including livestock and dairy farms. KUE was significantly lower for Grazing Only than both Grazing + Harvest and 2 Cut Harvest, which overall highlights the effectiveness of harvest events for nutrient removal, for example on high P or K fertility soils. Without targeted management, these outcomes have implications for soil fertility on these fields due to the greater levels of nutrient export.

Shalloo et al., (2011) showed that an increase in the amount of reseeding resulted in an increase in total herbage production, and even a longer growing season due to increased persistency in newly reseed swards. However, with a reseeded area it was also shown that fertiliser imports increased to meet increases in herd size and fertiliser requirements, while concentrate imports declined, due to a longer grazing season (concentrate feed being replaced by herbage in the diet). High persistency becomes very important in a sward because frequency of cultivation and reseeding increases costs (Wilkins and Humphreys, 2003). Therefore, the level of persistency of the reseeded sward affects herbage production and the nutrient requirements for subsequent reseeding. Given this, the significantly lower PBAL, and higher PUE, for fields reseeded for 5 to 10 years, compared to fields reseeded for < 5 years, were most likely due to the higher nutrient requirement of newly established pastures. Furthermore, the lower PUE for fields reseeded for > 10 years, may be related to a decrease in sward persistency, which may require a higher nutrient import to compensate, thus inversely impacting PUE. Overall, the results suggest that establishment stage (< 5 years reseeded) and an absence of reseeding (> 10 years reseeded) significantly impact variation in P cycling (Figure 4-10). However, the same cannot be said for K cycling which was not impacted by frequency of reseeding.

4.6 Conclusion

Overall, the results of this study show that consideration of farm and field scale together is important for identifying differences in indicative nutrient management (farm scale) and actual nutrient management (field scale), which are in agreement with the findings of McDonald et al., (2019); Van Beek et al., (2009); Van Leeuwen et al., (2019). It was shown that the mean nutrient balance (source pressure) between the two scales can be significantly different, specifically for P (10 and 6 kg P ha^{-1}, respectively), which poses a higher risk of having an environmental impact than when lost from the soil to waterbodies, in comparison to K. The scale of monitoring and management of P and K on grassland farms therefore has implications for perceived environmental risk, and current nutrient regulation in agriculture, under the Nitrates Directive. Results from this study would suggest that the current maximum soil sample area (5 ha) under these regulations in Ireland, may be too large, to effectively target nutrient management spatially, particularly on dairy farms where dairy paddocks (the “milking platform”; mean paddock area 1.4 ha) are the effective scale that management actually occurs on these systems.

These results show that there was substantial variation at field scale (CV 294 and 179% for P and K balance, respectively), which farm scale measures may not be sufficient to address. The large variation as illustrated (Figure 4-3), highlights that there were areas of high risk of nutrient loss occurrences with potential environmental impacts, but also areas with of very low risk, within any one farm, may be invisible to farmers, advisors and policy makers if only looking at the farm scale. These results also show that much of the variation in the nutrient P and K indicators measured, could be attributed and grouped based on controllable grassland management factors like sward species composition, cropping and reseed rate but also uncontrollable environmental factors like gradient and soil drainage class. Although it can be expected that many other factors can contribute to variation too, including but not limited to, use of extension services and technology adoption (Higgins et al., 2019; Hyland et al., 2018; Zhang et al., 2002). Considering these factors in an optimised field/paddock scale nutrient management strategy, could eventually lead to reduced nutrient losses and improved production efficiency at the farm scale. The latest EPA water quality results, which indicate an increased risk of decline due to losses from agricultural (EPA, 2018b), combined with the need to improve soil fertility levels across farms nationally (Wall et al., 2015), may justify the development and promotion of new spatially explicit nutrient management strategies and tools to reduce nutrient surpluses and increase use efficiencies on dairy farms.

This study has the advantage of being based on commercial dairy farms, instead of research farmlets, or plot studies, where results may not reflect some of the realities of spatial heterogeneity in nutrient management, cycling, and potential losses. It is recommended that advisors and farmers should seek out decision support tools and technologies that enable them to identify and manage this spatial variability that occurs at field/paddock scale.

Further research could include implementation of farm and field scale measures, to address this spatial variability, on a larger sample size of farms and over a longer time period, with more intensive advisor-farmer engagement. Such an experiment could be carried out on a cohort of farms that participate in the national level reporting surveys for example – the Agricultural Catchments Programme (Jordan and Shortle, 2017). The results of this study could also be used in conjunction with farm systems modelling approaches to further test scenarios in order to optimise nutrient management strategies and advice and also to guide policy options. Consideration of the spatial variability of nutrient balances in conjunction with the spatial variability of soil fertility levels (soil plant-available P and K assessed by soil sampling and analysis) could also provide valuable insights into important relationships between changes in nutrient management and changes in soil fertility.

CHAPTER FIVE: CHANGES IN SOIL FERTILITY AND THE FACTORS THAT AFFECT IT AT PADDOCK SCALE ON COMMERCIAL GRASS-BASED DAIRY FARMS

Preface: This paper has been prepared for the submission to the journal *Nutrient Cycling in Agroecosystems*. Formatting and reference style have been edited for consistency throughout the thesis. Figure and table captions have been assigned with a chapter prefix. Acknowledgements have been removed.

Keywords: Soil Fertility, Scale, Paddock, Spatial Distribution, Nutrient Advice, Balance, Imports, Soil pH, Nitrogen, Phosphorus, Potassium

5.1 Abstract

Optimising soil fertility is essential for achieving farm agronomic and environmental goals simultaneously. However, not only are there barriers to adopting nutrient management best practices, there seems to be barriers to correctly prescribing nutrient management advice, specifically in relation to scale and spatial heterogeneity. These barriers can be particularly significant in pasture-based farm systems such as those prevalent in Ireland and other temperate climates. Specialised pasture-based dairy farms also represent the more intensive grassland farms in Ireland, in terms of stocking density, fertiliser and feed use. It was hypothesised that providing nutrient management advice at paddock scale (dairy paddocks being the smallest, ranged from 0.7 to 2.6 ha) would improve the sustainability of nutrient management, as indicated by nutrient balances, use efficiencies, and soil fertility (soil pH level, Morgan's soil test phosphorous and potassium) on specialised pasture-based dairy farms. The objective of this study was to better match the scale and spatial distribution of nutrient management advice to the scale and spatial heterogeneity in factors controlling nutrient cycling and losses, by providing soil test results and nutrient management recommendations on a per paddock/field basis i.e. the smallest areas of uniform management on these farms. A total of 15 specialised commercial dairy farms with 374 fields/paddocks were selected from the south and south east region of Ireland. Paddock scale characteristics (soil chemical and physical properties and sward management strategies), were evaluated as variables that could impact the capacity to change soil fertility levels. Multiple stepwise linear regression was carried out to investigate the relationship between dependent (soil pH, soil test phosphorous potassium) and independent (soil characteristics, nutrient and sward management) variables. It was found that paddock scale advice was not strictly adhered to by participants, with the exception of phosphorous (P). However, during the three-year advice period, soil fertility levels increased towards more optimum levels, and the distribution of soil fertility indices improved towards more agronomically optimum indices also. At paddock scale, mean soil pH increased towards a more agronomically ideal 6.3 level. Mean soil test P and soil test potassium (K) decreased from high fertility to 6.7 and 142.9 mg l^{-1}, i.e. closer in range to their agronomic optimum. As well as this, mean paddock scale nutrient balance surpluses decreased over time; P and K surplus balances decreased significantly between 2015 and 2016, from 7 to 4 kg P ha^{-1}, and from 52 to 34 kg K ha^{-1}, respectively. It was also shown that change in soil pH was significantly affected by soil organic matter ($p < 0.05$), which was also contributed the largest driver of variation in the regression model ($\eta^2 = 0.16$). The largest significant driver of change in soil test P was Aluminium (Al) levels in the soil ($p < 0.05$, $\eta^2 = 0.42$). Whereas the largest driver of change in soil test K, was an interaction between white clover (*trifolium*

folium) content in the sward and time since reseeded, ($p < 0.05$, $\eta^2 = 0.23$). These results highlight that different soil fertility components can have different drivers of spatial heterogeneity of nutrient cycles at paddock scale.

This study demonstrated that there is significant potential to improve the sustainability of nutrient management on these dairy farms through better matching the scale and spatial distribution of nutrient management advice to the scale and spatial heterogeneity in factors controlling nutrient cycling and losses. Specifically, the paddock scale (ca. 0.5 – 2.5 ha) might be a suitable scale to target improved nutrient management practices, as these are already established as spatial management units. The implication is that traditional methods of soil sampling based on larger areas do not seem to be as effective for monitoring soil fertility and nutrient source pressures or guiding nutrient management recommendations. In aiming to achieve optimum soil fertility levels sward management factors (species, cropping and reseeding) for example, may be best managed on a paddock by paddock basis which would be expected to increase in precision nutrient management practices.

5.2 Introduction

In grazed grass dairy systems with significant exports, it is necessary to replace exported nutrients with imported nutrients (e.g. phosphorous (P), potassium (K), calcium (Ca)) in order to maintain soil fertility and soil quality in the long term (Clercq et al., 2001; Krishna, 2013; Reijneveld et al., 2014; Schröder and Neeteson, 2008; Wheeler et al., 2004). A nationally representative study of long-term fertiliser use (2005 to 2015) in Ireland, found that fertiliser applications declined by 11 to 16% on grassland enterprises (Dillon et al., 2018). The largest decrease in fertiliser use during this period was attributed to an increase in fertiliser price, while the largest increase was attributed to severe weather events, which subsequently, lead to a fodder shortage. It was also highlighted by this study, that a very small proportion of farms were regularly importing lime. This would be expected to have implications for soil pH fertility and nutrient use efficiency for example. In 2012, national targets were set for the dairy industry to increase dairy output by 50% by 2020 (Food Harvest 2020, 2010). This target was met prior to 2020, as indicated by the substantial increase in milk production post milk quota abolition in 2015 (CSO, 2019; NFS, 2017). However, the increased production may have added environmental risk, for example, through increased herd sizes and increased fertiliser use (CSO, 2016).

A recent Irish water quality report (2017) indicates that nutrients from agriculture (diffuse source), and sewage (point source), are the main sources of eutrophication of waterways. The report concludes that there was a 3% reduction in water quality since 2015, and that the number of high-status waterbodies was declining (EPA, 2018b). Yet, at the same time, it was found that Irelands soil fertility was almost at its lowest in 2014. For example, a survey of soil test results of samples submitted to the national agricultural research and advisory service (Teagasc, 2018), found that only 11% of agricultural soils were at optimal levels for soil pH, P and K (Kelly et al., 2014; Teagasc, 2018). Approximately 65% of dairy farm soils (21,676 samples between 2014 and 2018) were found to be sub-optimal for pH (target: 6.1 to 6.5 on grassland mineral soils) and ca. 55% of soils were found to be sub-optimal for P and K (Teagasc, 2018). Therefore, even though soil fertility levels were low, water quality impacts were still occurring, as well as, potentially impacting the production levels and nutrient use efficiency of the soil. It is likely that indiscriminate or blanket nutrient applications are being carried out, instead of targeted nutrient management to specifically increase the proportion of fields in optimum levels of soil pH, P and K. As such, a legacy of poor distribution of nutrients at farm and field scale can contribute to a lack of appropriate change in soil fertility and increase the risk of nutrient loss to the wider environment.

In grazed grass dairy systems, the mismatch in the ratio of organic manure nutrient content to the ratio of plant nutrient uptake, makes optimising soil nutrient balances difficult (Sharpley and Moyer, 2000). However, at a small catchment scale (ca. 8 km^2), Murphy et al., 2015 found that improvements in nutrient management lead to improvements in soil P status (less soils in excess, more in optimum), and an emerging trend in decreased P concentrations in stream water, from a dairy dominated catchment in southwest Ireland. Additionally, Aarons et al., (2015) and Gourley et al., (2015) found that a better understanding of the nutrient distribution issue can lead to economic and environmental benefits at farm and national scale.

Soil fertility characteristics change over time due to changes in fertiliser import practises, for example, in response to fertilisation recommendations and governmental regulations (Buckley et al., 2016a; Clercq et al., 2001; Schröder and Neeteson, 2008; Wall et al., 2013). As a result, it is difficult to determine how quickly a change should occur after policy has been implemented. Reijneveld et al., (2010) showed how policy on manure application from 1984 resulted in increasingly tight restrictions on P application from manure and fertilisers, but the effects were not yet clearly reflected in changed trends in soil P. Making the necessary changes to nutrient management as efficiently as possible is important, especially with agriculture being expected to meet increasing production and environmental targets (Lanigan et al., 2018; Arneth et al., 2019).

It has also been shown that inter-paddock variability in soil nutrients in pasture grazing systems can predominately be driven by farm management decisions (Gourley et al., 2004; Mathews et al., 1994). To address this, the changes needed will mostly come through the knowledge transfer (KT) network of researchers, advisors and farmers. Although changes to best practice can be achieved through KT, the scale of advice is typically farm level. In practice, the smallest management unit relevant to nutrient distribution on a dairy farm is typically a paddock, which is a much smaller area. Management units however can vary in size depending on the objective of the operator (Krishna, 2013). Overall, there can be a mismatch between farm scale advice and field or paddock scale nutrient management. Therefore, the planning of nutrient management should match the scale of the smallest management unit i.e. the paddock on grass-based dairy system, so as to reduce the potential of mismatching the recommended application rate and the actual import rate. By using the paddock to guide the nutrient management planning process, it requires adjusting the soil sampling area appropriately. For example, farm scale studies have carried out soil analysis with 4 ha soil sample areas per farm (Mihailescu et al., 2015), which may be too broad, and paddock scale studies (farmlet experimental units) have carried out analysis with 1.9 ha samples per farm (Aarons et al., 2015) which may be more appropriate.

Ideally a sampling strategy should be influenced by spatial variability in soil chemical properties of soil fertility (Mathews et al., 1994). Zonal or grid sampling, a technique of precision agriculture, is often recommended to ensure that nutrient enriched zones are identified and excluded from sampling, and thus give an accurate estimate of the fertility of an area (Mathews et al., 1994; Anderson et al., 1992; Higgins et al., 2019). However, there may be difficulty in matching nutrient imports to enriched zones, if the appropriate technology and machinery is unavailable. Soil sampling methods need to be developed to accurately reflect the scale of nutrient management practice.

The adoption of precision technologies to better match the scale of soil management to the scale of variation in soil fertility is still limited by a lack of skill-training and costs, even though several options are now available (Zhang et al., 2002; Higgins et al., 2019). The paddock scale may be a sufficient for the scale that is spatially appropriate, to significantly improve the differences between soil variability and nutrient management. This scale may also have an added advantage in that it is already a scale (management unit) that is in operation on most Irish dairy farms in terms of grazing and grass management infrastructure.

This study focused on quantitative changes in soil fertility, and the soil properties and management factors that affect these changes, at the paddock/field scale. Nutrient management was monitoring over 3 growing seasons and soil fertility was taken over 4 years on 15 commercial grazed grass dairy farms in southern Ireland. The objective of the study was to encourage the participating farms to better match the scale of nutrient advice to nutrient management by providing soil test results from the smallest management unit on the farm (paddock/field). The changes were then monitored over time. The hypotheses tested were 1) Matching the scale of advice to the scale of imports improves soil fertility (soil pH, P and K) and distribution (soil indices) and 2) Changes in soil fertility (pH, P and K) is primarily driven by nutrient management (imports and balances) at paddock scale.

5.3 Materials and methods

A summary of the materials and methods used is provided here. Further details of materials and methods can be found in Chapter Three and Four.

5.3.1 Farm selection and farm characteristics

Farms were selected from the south and south east region of Ireland, in counties Cork, Tipperary and Wexford. These regions are typically associated with long annual growing seasons (~242 days) and a high proportion of well drained soils (NFS, 2017). Average annual rainfall for these regions ranged from 967 to 1,208 mm (2015 to 2017). Average air temperature ranged from 9.7 to 10.3 °C (2015 to 2017). Personal communication with agricultural advisors was used to identify dairy farmers willing to participate in the study. A total of 21 dairy farms were selected for this study, but only 15 participated fully (provided sufficient data to carry out the full suite of analysis for this study). All participants were specialised dairy farmers with dairy cows contributing to more than 3/4 of standard agricultural output (CSO, 2016).

Table 5-1. Mean values and standard deviation for total grassland area, crop area, rainfall, air temperature, total utilisable agricultural area stocking rate, milk solids, and concentrate feed per cow and soil texture class for 15 Irish dairy farms between 2015 and 2017

Farm ID	TGA (ha)	Crop (ha)	Rainfall (mm yr^{-1})	Air temp. (°C)	Texture Class	TUAA SR (LU ha^{-1})	TUAA SR (kg Org N ha^{-1})	MS (kg ha^{-1})	Conc. (kg LU^{-1})
1	63.8	0.0	1,007	10.2	Sandy loam	2.5	209	813	674
2	35.0	0.0	1,208	10.1	Sandy loam	2.5	211	687	757
3	96.6	0.0	1,032	10.1	Sandy loam	2.2	191	631	298
4	44.7	2.3	1,032	10.1	Clay loam	2.6	225	1,185	866
5	58.3	0.0	967	9.7	Clay	1.7	146	624	1,106
6	34.9	1.8	1,007	10.2	Sandy loam	2.7	233	821	753
7	38.0	0.0	1,032	10.1	Sandy loam	2.0	170	797	938
8	105.3	8.4	1,208	10.1	Sandy loam	2.9	247	1,041	850
9	103.0	4.6	996	10.3	Sandy loam	2.8	238	1,128	1,128
10	83.0	0.0	967	9.7	Clay loam	2.1	177	734	1,027
11	130.1	18.2	1,007	10.2	Sandy loam	2.0	174	607	236
12	60.0	0.0	967	9.7	Clay	2.0	166	534	1,315
13	32.0	0.0	1,208	10.1	Sandy loam	2.6	219	1,089	867
14	73.3	40.4	992	9.8	Sandy loam	1.3	109	915	1,124
15	82.0	0.0	967	9.7	Clay loam	1.8	149	487	733
Mean	69.3	5.0	1,040	10		2.2	191	806	845
S.D.	30.3	11.0	90	0.2		0.5	39	222	295

TGA, Total Grassland Area; Temp, temperature; TUAA, Total Utilisable Agricultural Area; Sandy loam, Well drained soil; Clay loam; Moderately drained soil, Clay; Poorly drained soil; SR, Stocking Rate; LU, Livestock Unit; Org, organic; MS, Milk Solids; Conc., Concentrate feed; S.D., Standard Deviation.

Farm characteristics for the duration of the study period are summarised in Table 5-1. Average Total Utilisable Agricultural Area (TUAA) was 53.1 ha (includes total grassland and crop area), which was slightly below the national average of 56.0 ha for dairy enterprises in Ireland (NFS, 2017). Average stocking rate was 2.9 LU ha^{-1} (1 livestock unit is equivalent to one adult cow producing 3,000 kg milk annually, (Eurostat, 2013), which was higher than the national average of the previous three years: 1.75 LU ha^{-1} in 2012 to 2014 (Butler et al., 2017). Average milk solid production (fat plus protein) was 806 kg ha^{-1}, which was above the national average (649 kg ha^{-1}) for the 2012 to 2014 period. Concentrate feed imported onto the farm was, on average, lower on these farms (847 kg cow^{-1}), compared to the national average (926 kg cow[1]) (Butler et al., 2017). Ten out of the fifteen farms in this study were on average above the 170 kg organic N ha^{-1} restriction on stocking rate (SR) and were, therefore, in nitrates derogation conditions. Some farms crossed over the derogation threshold during the study period. Ten farms were on well drained soils (Sandy loam), three farms on moderately drained (Clay loam), and two farms on poorly drained (Clay). Meteorological data (rainfall and air temperature), was gathered from meteorological stations (Met Éireann, 2019) in the region.

5.3.2 Nutrient management and nutrient balance methodology

Paddock scale nutrient management (imports and exports) and nutrient balance results for N, P and K calculated in Chapter Three and Four were used as explanatory variables in the analysis for the changes in soil fertility (Equation 5-1 and Equation 5-2). Details of the methodology for imports and exports are shown in Table 5-2. Losses via runoff and leaching were not measured. Paddock scale imports included P and K from inorganic and organic fertilisers and concentrate feed. Fertiliser imports were based on individual field records made by the farmer. Total concentrate feed imported and total milk and animals sold was apportioned across the farm based on the respective number of grazing defoliations for each paddock. Nutrient (N, P and K) balance (kg nutrient ha^{-1}) was calculated as:

$$N, P\ and\ K\ Balance\ (kg\ ha) = \frac{(N, P\ and\ K\ imports - exports\ (kg))}{Spatial\ area\ (ha)}$$ **Equation 5-1.**

$$N, P\ and\ K\ Use\ Efficieny\ (\%) = \frac{N, P\ and\ K\ exports\ (kg)}{N, P\ and\ K\ imports\ (kg)} X\ \frac{100}{1}$$ **Equation 5-2.**

Table 5-2. Import variables (inorganic fertiliser, concentrate feed and organic manure) and export variables (milk sold, animals sold and silage) used to calculate nutrient balance and use efficiency for farm and field (Dairy, Non-Dairy and Silage) scale

Import (kg N, P or K ha^{-1})		Paddock/Field	Export (kg N, P or K ha^{-1})	
Inorganic fertiliser	(+)		Milk sold	(-)
Organic manure	(+)	Dairy Paddocks	Animals sold	(-)
Concentrate feed	(+)		Silage	(-)
Inorganic fertiliser	(+)			
Organic manure	(+)	Non-Dairy Fields	Animals sold	(-)
			Silage	(-)
Inorganic fertiliser	(+)			
Organic manure	(+)	Silage Fields		
			Silage	(-)

5.3.3 Subzone characterisation

The Total Grassland Area (TGA) was divided into multiple zones for each farm. TGA averaged 69.3 ha over the three years (Table 5-1). This grassland area was subdivided into Dairy, Non-Dairy and Silage subzones, within which, the individual fields and paddocks were monitored. ArcGIS® software (ArcGIS, 2016) was used to map individual paddock, field and farm boundaries. Bing maps aerial photography was used as a base-map (Bing™ Maps, 2005). The three subzones were characterised as follows:

- **Dairy subzone**. On average, there were 27 Dairy paddocks per farm, with an average area of 1.4 ha. Dairy subzone fields made up, on average, 57% of the TGA of these farms.
- **Non-Dairy subzone**: There were, on average, 7 Non-Dairy fields on each farm, with an average area of 3 ha. Non-Dairy subzone fields made up 28% of the TGA, on average.
- **Silage subzone**: On average, there were 3.6 grass silage fields per farm, at 2.6 ha in area. Silage subzone fields made up 15% of the TGA, on average.

Paddocks on the Dairy sub-zone ranged from 0.7 to 2.6 ha and paddocks and fields on the Non-Dairy and Silage subzones ranged from 1.1 to 6.5 ha.

5.3.4 Paddock scale soil analysis, advice and monitoring

Surface (0-100 mm) soil sampling was carried out in accordance with S.I. No. 605 of 2017 (EU, 2017). Each management unit (paddock/field) was sampled in a "W" shaped pattern. A stainless-steel soil sampler with a 100 mm length sampling cone [see Appendix 1A: Figure 9-1) was used to take 20 soil cores from each sampling area and were boxed and labelled accordingly. Within each management unit, care was taken not to sample nutrient enriched areas, S.I. No. 605 of 2017 (EU, 2017), such as water troughs, field entrances, areas of heaped manure etc as they may overestimate the nutrient distribution. All sampling was carried out at least 3 months after the closing dates for fertiliser and manure spreading to avoid legacy soil nutrients. Soil sampling was mostly carried out between December and February of each year from 2015 to 2018, with 2015 as the baseline year of the study. Soil samples were dried at 40 ^{o}C for 24 hours using forced draught ovens and sieved to 2 mm. A representative sub-sample was then taken for chemical analysis.

5.3.4.1 Soil test (plant available) Phosphorus and Potassium

Morgan's extract (Peech and English, 1944) was used to measure the soil test plant-available P (STP) and soil test K (STK) levels using, the method described by Byrne, (1979). Morgan's solution was 1,400 ml of 40% sodium hydroxide (NaOH) in approximately 15 litres of water. 1,440 ml of glacial acetic acid (CH_3COOH) was then added. Solution was increased to 20 litres with water, and pH was adjusted to 4.8. A volume ratio of one part soil sub-sample to five parts of solution was used. The solution was then mixed for 30 minutes and filtered using 0.45 µm Whatman$^{(TM)}$ filter paper into vials for analysis. Samples were quality checked by including one blank and ten control per run of 89 soil samples. Elemental analysis was then carried out on the clear extract by spectrophotometry at 880nm using the phosphomolybdate method (Murphy and Riley, 1962; Treacy et al., 2008).

5.3.4.2 Other soil chemical properties (Mehlich 3)

Mehlich 3 extractant (Mehlich, 1984) was used to determine concentrations of Al, Ca, copper (Cu), Iron (Fe), magnesium (Mg) and Manganese (Mn) in soil. The Mehlich 3 reagent (M3) comprises of 0.2M acetic acid (CH_3COOH), 0.25M ammonium nitrate (NH_4NO_3), 0.013M nitric acid (HNO_3), 0.015M ammonium fluoride (NH_4F), and 0.001M ethylenediaminetetraacetic acid (EDTA) (Mehlich, 1984). Mehlich 3 was buffered at pH 2.5, to prevent reaction of Ca with Fluorine. A 2 g sample of soil was placed in a centrifuge tube and 20 ml of reagent was added. The solution was then placed on an end-to-end shaker for 60 minutes and allowed to settle for 30 minutes. The solution was then filtered through 0.45 µm Whatman(TM) filter paper. Again, elemental analysis was carried out by spectrophotometry.

5.3.4.3 Soil pH and lime requirement

Soil pH was determined by measuring pH in a soil:water suspension of 1:2 (S.I. No. 605 of 2017, EU 2017). A volume of 10 ml of soil sub sample and 20 ml of deionised was placed in a clean beaker. The solution was stirred for 30 minutes and allowed to settle for 10 minutes twice. Soil pH was then determined using a calibrated pH meter. Lime requirement was also determined according to Pratt and Bair, (1962).

5.3.4.4 Organic matter analysis

Soil organic matter (SOM) content was determined by loss on ignition (LOI) according to S.I. No. 605 of 2017 (EU, 2017). A small quantity of soil was prepared and placed in an oven for 16 hours at 105°C. Approximately 4 g of soil was placed into a crucible (pre-weighed). The initial weight of the soil was then determined. The sample was then placed in a furnace at 500°C for 16 hours ("ashing"). The final weight of the soil was determined after the sample was cooled. The difference in weight between the initial and final weights was expressed as a percentage and represented soil organic matter for that sample.

5.3.4.5 Paddock scale nutrient advice maps

The Soil Index system for STP and STK in Ireland, which describes nutrient status and associated fertiliser recommendations (Wall and Plunkett, 2016), was applied to each soil test result, each year, so that each paddock was assigned one of four Index categories (Table 5-3). Four Index categories were also used for pH with pasture on mineral soils. This Index systems classifies anything over 6.5 pH as Sufficient/High (Table 5-3).

Table 5-3. Soil Index system and associated soil status for plant available phosphorous (STP), plant available potassium (STK) (both using Morgan's extractant), and soil pH indices and ranges used for this study). Adapted from Wall and Plunkett, (2016)

Soil Index	Soil status	STP (mg l^{-1})	STK (mg l^{-1})	pH
Index 1	Very Low	0.0 to 3.0	0 to 50	< 5.5
Index 2	Low	3.1 to 5.0	51 to 100	5.6 to 6.0
Index 3	Medium/Adequate	5.1 to 8.0	101 to 150	6.1 to 6.5
Index 4	Sufficient/High	> 8.0	> 150	> 6.5

To guide nutrient management for farmers, colour coded paddock scale maps were generated showing the Index for pH, P, and K based on soil test (Figure 5-1). In spring 2015 (baseline year), paddock scale map was developed and given to each participating farmer and their advisor. A recommended annual input rate (kg ha^{-1}) per field was generated for P, K and lime (t ha^{-1}). The rate was calculated based on the paddocks individual soil nutrient status and farm stocking rate, using standard recommended rates for pasture production in Ireland (Wall and Plunkett, 2016). In 2015, the maps were the first indication of the distribution of soil fertility at the paddock scale for participating farmers. The maps were re-generated each year according to soil sampling results.

5.3.5 Field environmental factor classification

Two environmental factors that might be expected to impact the change in soil fertility were assessed: gradient and field soil drainage class.

5.3.5.1 Gradient class

Fields were classified as sloping or level. Level was defined as < 10 degrees of slope and Gradient as >10 degrees of slope (Roberts and Watkins, 2014).

5.3.5.2 Field soil drainage class

Fields were classified as well drained (Sandy loam), moderately drained (Clay loam) and poorly drained (Clay) (CTECO, 2010; Simo et al., 2007).

5.3.6 Field management factor classification

Three sward management factors that might be expected to affect soil fertility were assessed: sward species composition, cropping and reseeding.

5.3.6.1 Sward species composition

Perennial Ryegrass (PRG) was grouped into three different proportions: 0 to 75% PRG (Low), 76 to 90% PRG (Medium), and 91 to 100% PRG (High). White clover (WC) was grouped into two levels: WC absent and WC present.

5.3.6.2 Cropping rate

Each field was classified into one of three cropping classes; Grazing Only (GO), Grazing + Harvest (G+1H), and 2 Cut Harvest (2CH).

5.3.6.3 Reseeding interval

Fields were grouped into three categories based on the number of years since a reseeding event occurred; < 5 years, 5 to 10 years, and > 10 years.

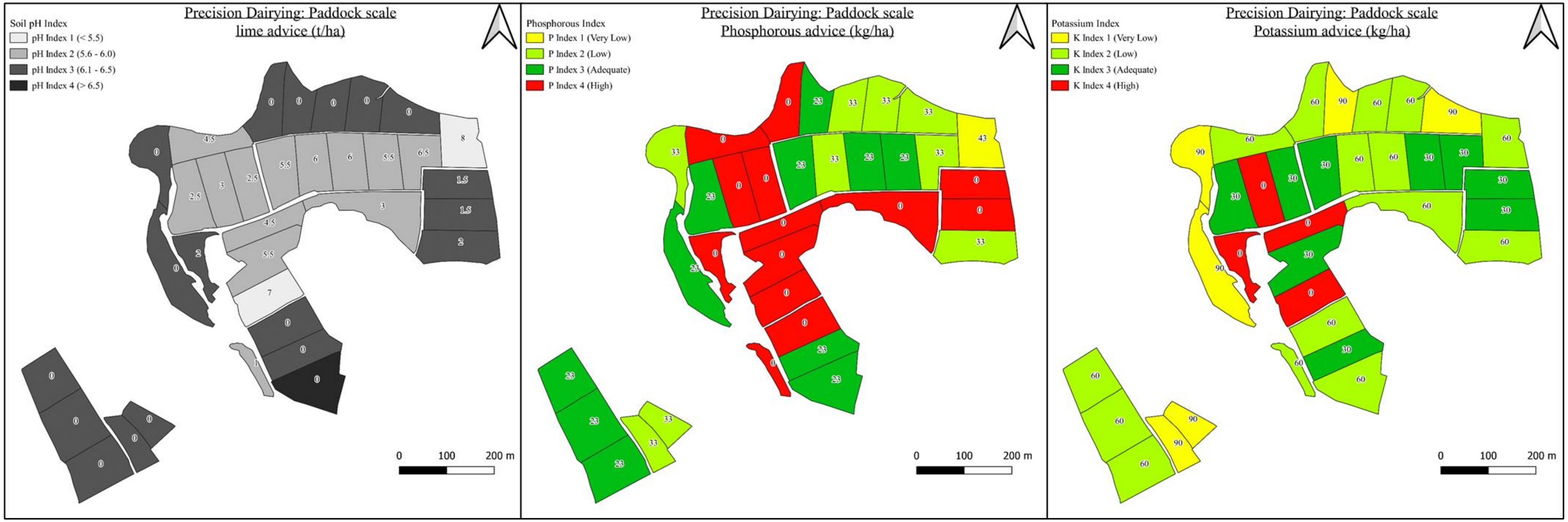

Figure 5-1. An example of the paddock scale nutrient management advice maps for lime, P and K provided to participating farmers during the study.

5.4 Statistical analysis

All data were screened for outliers and data entry errors. Measures of centrality and variation were determined in SPSS (IBM SPSS, 2016). Tests for normal distribution were carried with Kolmogorov-Smirnov where $n > 100$. Multiple stepwise linear regression was carried out to investigate the relationship between dependent and independent variables, see Equation 5-3. Scatterplots of standardised predicted values verses standardised residuals were used to test if the data met the assumptions of homogeneity of variance and linearity, and the residuals were approximately normally distributed. Where all assumptions were met, Wilkes Lambda was used to determine significant factors, where assumptions were not met, Pillas trace was used instead. A significance level of < 0.05 was used to determine the difference between means. Bonferroni post-hoc test was used to distinguish where the significant differences occurred within a group (Pairwise comparison). For data that was not normal, tests carried out would be non-parametric. Non-parametric independent tests were carried out to determine significance. Kruskal Wallis and Mann Whitney U tests were used to distinguish where the significant differences occurred. The following equation were used to investigate the second hypothesis outlined previously:

$$\textbf{Field scale}: \hat{Y}1 = \beta o + \beta 1X1 + \beta 1X2 + \beta 1X3 + \beta 1X4 \qquad \textbf{Equation 5-3.}$$

where $\hat{Y}1$ = ▲ soil fertility (soil pH, STP and STK),
X1 = soil chemical properties (SOM, Al, Ca, Cu, Fe, Mg, Mn),
X2 = environmental factors (gradient and drainage class),
X3 = sward management factors (species composition, cropping, reseeding),
X4 = nutrient management (lime, N, P and K imports and N, P and K balance).
▲ = delta, representing “change in”

5.5 Results

5.5.1 Nutrient advice, import, balance and soil fertility

Nutrient imports, exports and balance results for N, P and K are described in detail in Chapter Three and Four. A summary of paddock scale nutrient advice (P and K), nutrient imports (lime, P and K), nutrient balance (N, P and K) and soil fertility (soil pH, STP, and STK) for each farm is shown in Table 5-4. Mean advice was 19 kg P ha^{-1} yr^{-1} (SD, 9), 29 kg K ha^{-1} yr^{-1} (SD, 13) and 1,672 kg ha^{-1} yr^{-1} lime (SD, 1,145). Mean import per farm was 2,139 kg ha^{-1} yr^{-1} lime (SD, 2,128), 255 kg N ha^{-1} yr^{-1} (SD, 58), 21 kg P ha^{-1} yr^{-1} (SD, 10), and 76 kg K ha^{-1} yr^{-1} (SD, 29). Mean balance per farm was 139 kg N ha^{-1} yr^{-1} (SD, 60), 6 kg P ha^{-1} yr^{-1} (SD, 9), and 39 kg K ha^{-1} yr^{-1} (SD, 24). Mean soil pH was 6.3 (SD, 0.2), mean STP was 7.1 mg l^{-1} (SD, 2.4) and mean STK was 146.7 mg l^{-1} (SD 50.2). Note that mean soil fertility values were not weighted by field area.

Table 5-4. Mean and standard deviation for nutrient management (advice, import, balance) and soil fertility (soil pH, soil test phosphorus and potassium) for participating farms (*n* = 15)

	Advice			Import				Balance			Fertility		
Farm	**P**	**K**	**Lime**	**Lime**	**N**	**P**	**K**	**N**	**P**	**K**	**Soil pH**	**STP**	**STK**
					kg ha^{-1}							mg l^{-1}	
1	20	12	979	4,495	267	42	113	87	20	30	6.6	7.9	212.8
2	34	45	3,569	2,818	285	12	55	215	5	38	6.1	3.2	91.1
3	10	33	1,149	2,071	166	12	47	25	-7	3	6.3	10.6	121.5
4	19	42	0	1,685	244	10	68	157	-3	39	6.5	6.8	100.6
5	13	34	2,576	0	206	22	69	141	13	31	6.1	8.0	141.4
6	35	24	2,598	0	217	28	66	126	17	42	6.4	3.4	157.0
7	19	31	2,316	3,803	219	30	92	88	13	27	6.1	5.6	125.1
8	33	23	3,135	1,823	374	21	98	233	-2	67	6.0	4.0	141.9
9	14	3	1,935	0	338	28	126	162	4	90	6.3	10.5	267.4
10	11	42	286	1,567	267	7	42	202	-3	39	6.3	7.8	104.8
11	19	17	2,219	7,360	299	25	64	34	-5	3	6.3	7.5	203.5
12	15	42	1,385	0	189	21	75	146	16	52	6.1	8.8	109.3
13	26	22	0	2,071	311	20	99	161	2	33	6.5	5.7	135.6
14	12	20	2,387	0	204	7	20	167	3	19	6.4	6.1	185.3
15	7	43	537	4,242	241	28	98	122	16	72	6.2	10.4	102.8
Mean	19	29	1,672	2,139	255	21	76	139	6	39	6.3	7.1	146.7
SD	9	13	1,145	2,128	58	10	29	60	9	24	0.2	2.4	50.2

5.5.2 Paddock scale soil analysis

Soil chemical property results are shown in Table 5-5. Soil chemical properties were analysed in 2015 only (*n* = 407). SOM had a mean of 10.4% and values ranged from 5.1 to 34.1% (SD, 2.7). Mean Al was 648.3 mg l^{-1} (SD, 235.5) and ranged from 6.5 to 1248.0 mg l^{-1}. Mean Ca was 1841.9 mg l^{-1} (SD, 542.7) and ranged from 633.3 to 3312.7 mg l^{-1}. Mean Cu was 4.7 mg l^{-1} and ranged from 0.8 to 23.7 mg l^{-1} (SD, 2.5). Mean Fe was 308.6 mg l^{-1} (SD, 79.2) and ranged from 161.9 to 576.9 mg l^{-1}. Mean Mg was 201.6 mg l^{-1} (SD, 102.7) and ranged from 54.5 to 642.7 mg l^{-1}. Mean Mn was 155.1 mg l^{-1} (SD, 91.9) and ranged from 21.4 to 479.4 mg l^{-1}.

Table 5-5. Mean and standard deviation for soil chemical properties (aluminium, calcium, copper, iron, magnesium, and manganese) extracted using Mehlich 3 extractant (Mehlich, 1984), and soil organic matter determined by loss on ignition at field scale (n = 374) in 2015

	SOM	**Al**	**Ca**	**Cu**	**Fe**	**Mg**	**Mn**
	(%)	**(mg l^{-1})**					
Mean	10.4	648.3	1841.9	4.7	308.6	201.6	155.1
SD	2.7	235.5	542.7	2.5	79.2	102.7	91.9
Min	5.1	6.5	633.3	0.8	161.9	54.5	21.4
Max	34.1	1248.0	3312.7	23.7	576.9	642.7	479.4

5.5.3 Changes in paddock scale soil fertility mean during the advice period

The mean soil fertility distribution and changes over time across all fields and subzones are shown in Table 5-6. Mean result for each component of soil fertility over the four years was 6.24 for pH, 7.30 and 141.76 mg l^{-1} for soil test P and K respectively. Over the course of the study mean soil pH changed from 6.02 in 2015 to 6.33 in 2018 (Figure 5-2). The change in pH at the end of the study was a mean increase of 0.10 (SD of 0.23). Mean STP changed from 9.10 to 6.77 mg l^{-1} and mean STK changed from 151.87 to 142.90 mg l^{-1} over the four years (Figure 5-3 and Figure 5-4). STP had a mean decrease of 0.78 mg l^{-1} (SD 1.93 mg l^{-1}). STK had a mean decrease of 2.99 mg l^{-1} (SD 15.89 mg l^{-1}).

Overall, pH had increased, STP had decreased, and STK had decreased. It was found, for all fields, that the change in pH and P were significant for each time period between 2015 to 2018 ($p < 0.05$). The change in soil K was significant for each period except for 2017 to 2018 ($p > 0.05$). Therefore, the increase in soil pH and decrease in STP were statistically significant (Figure 5-2 and Figure 5-3) The decreases in STK were significant between 2015 and 2017 but the increase between 2017 and 2018 was not (Figure 5-4).

Table 5-6. Mean and change in mean soil fertility results for soil pH, soil test phosphorus and soil test potassium across all fields and subzones (Dairy, Non-Dairy and Silage) over time from 2015 to 2018

	Year	pH	STP (mg l^{-1})	STK (mg l^{-1})	Time period	▲ pH	▲ STP (mg l^{-1})	▲ STK (mg l^{-1})
All fields *n* = 374	2015	6.02	9.10	151.87	2015baseline			
	2016	6.16	6.21	144.12	2015to2016	0.14	-2.89	-7.75
	2017	6.47	7.12	128.17	2016to2017	0.32	0.91	-15.95
	2018	6.33	6.77	142.90	2017to2018	-0.15	-0.35	14.74
	Mean	6.24	7.30	141.76	Mean	0.10	-0.78	-2.99
	SD	0.20	1.26	9.90	SD	0.23	1.93	15.89
Dairy *n* = 338 (90%)	2015	6.02	9.34	155.24	2015baseline			
	2016	6.15	6.29	148.32	2015to2016	0.13	-3.06	-6.91
	2017	6.48	7.18	129.97	2016to2017	0.33	0.89	-18.35
	2018	6.33	6.91	146.94	2017to2018	-0.15	-0.27	16.97
	Mean	6.25	7.43	145.12	Mean	0.10	-0.81	-2.76
	SD	0.20	1.33	10.73	SD	0.24	2.03	18.02
Non-dairy *n* = 20 (5%)	2015	5.86	6.68	126.30	2015baseline			
	2016	6.10	5.49	105.50	2015to2016	0.24	-1.18	-20.80
	2017	6.27	6.30	110.68	2016to2017	0.17	0.81	5.18
	2018	6.17	5.56	106.61	2017to2018	-0.10	-0.74	-4.08
	Mean	6.10	6.01	112.27	Mean	0.10	-0.37	-6.56
	SD	0.17	0.58	9.61	SD	0.18	1.05	13.17
Silage *n* = 16 (4%)	2015	6.15	7.00	112.70	2015baseline			
	2016	6.31	5.55	109.53	2015to2016	0.15	-1.46	-9.11
	2017	6.52	6.87	105.50	2016to2017	0.22	1.32	8.24
	2018	6.44	5.36	108.68	2017to2018	-0.08	-1.50	-8.88
	Mean	6.35	6.19	109.10	Mean	0.10	-0.55	-3.25
	SD	0.16	0.86	2.96	SD	0.16	1.62	9.95

5.5.4 Subzones

The sample size was small for the subzones other than dairy. Dairy paddocks made up over 90% of the samples. Non-dairy and silage fields made up 5 and 4%, respectively. However, each subzone had the same outcome; pH increased and soil test P and K decreased on average (Table 5-6). Non-dairy fields had the lowest pH in 2015 (5.86) followed by dairy and silage fields (6.02 and 6.15, respectively), but each subzone had increased pH equally (0.10) on average. STP in the 2015 baseline year was highest on dairy paddocks but this subzone also had the largest decrease in STP (-0.81 mg l^{-1}) at the end of the study. Silage and non-dairy fields had a mean decrease of 0.55 and 0.37 mg l^{-1}, respectively. STK was reduced by an average 2.76 mg l^{-1} on dairy fields and 3.25 mg l^{-1} on silage fields. Non-dairy fields had the largest decrease in STK (-6.56 mg l^{-1}).

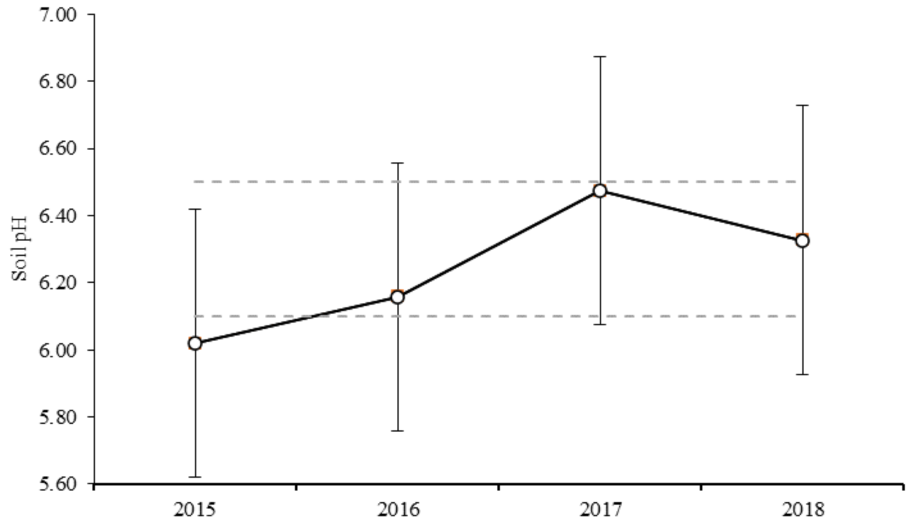

Figure 5-2. Mean pH for all fields between 2015 and 2018, optimum range for pH 6.1 to 6.5 (Index 3) indicated by dashed line. SD represented by whiskers.

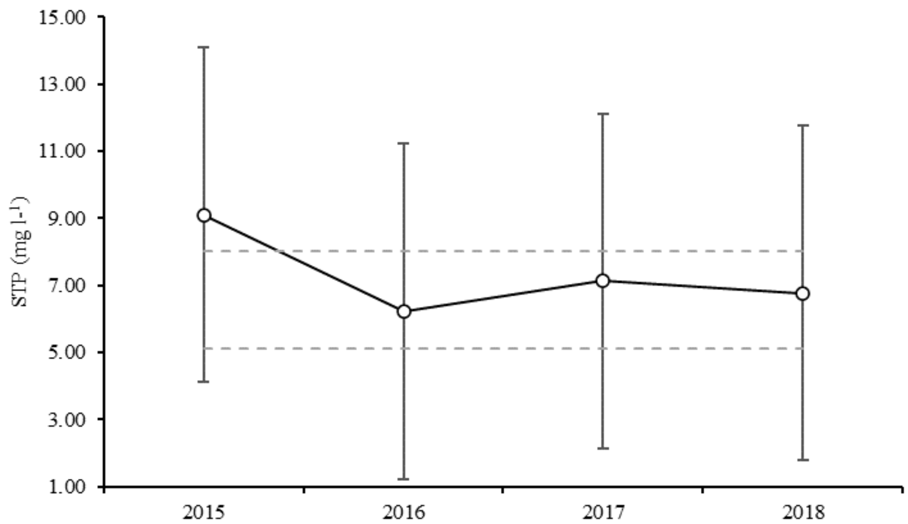

Figure 5-3. Mean STP for all fields from 2015 to 2018, optimum range for STP 5.0 to 8.0 mg l^{-1} (Index 3) indicated by dashed line. SD represented by whiskers.

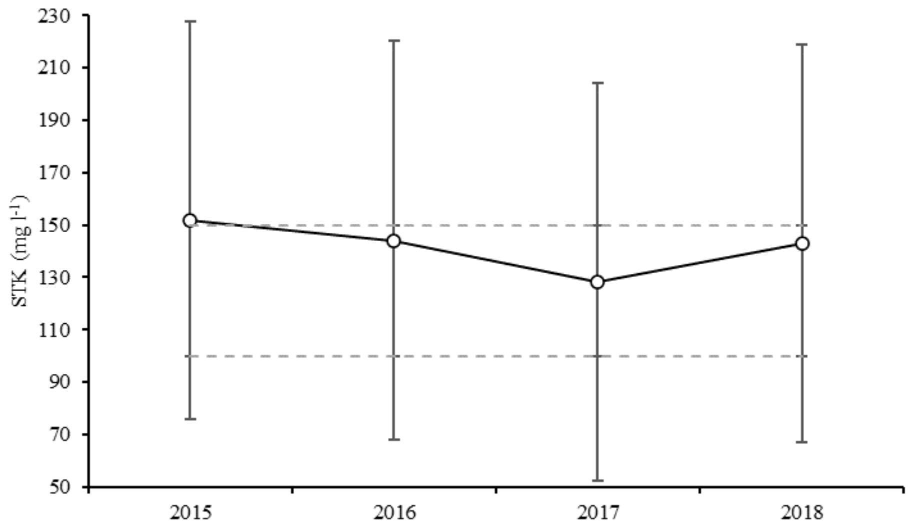

Figure 5-4. Mean STK for all fields from 2015 to 2018, optimum range for STK 101 to 150 mg l^{-1} (Index 3) indicated by dashed line. SD represented by whiskers.

5.5.5 Changes in distribution of soil index

The mean and change in mean distribution of soil fertility indices for each year is shown in (Table 5-7). The mean distribution over the study for pH was 5% of fields in Index 1, 32% in Index 2, 32% in Index 3 and 31% in Index 4. The mean distribution for STP was 18% in Index 1, 21% in Index 2, 28% in Index 3 and 33% in Index 4. The mean distribution for STK was 5% in Index 1, 29% in Index 2, 28% in Index 3 and 38% in Index 4.

Table 5-7. Mean and change in mean soil fertility index distribution for soil pH, soil test phosphorus and soil test potassium across all fields and subzones (Dairy, Non-Dairy and Silage) over time from 2015 to 2018

	pH				**Time period**	**▲ pH**			
	Index 1	**Index 2**	**Index 3**	**Index 4**		**Index 1**	**Index 2**	**Index 3**	**Index 4**
2015	10%	44%	35%	11%	2015baseline				
2016	7%	34%	37%	21%	2015to2016	-3%	-10%	3%	10%
2017	2%	15%	31%	52%	2016to2017	-5%	-19%	-6%	31%
2018	1%	32%	26%	41%	2017to2018	-1%	17%	-5%	-11%
Mean	5%	32%	32%	31%	Mean	-3%	-4%	-3%	10%
SD	4%	12%	5%	18%	SD	2%	19%	5%	21%
	STP					**▲ STP**			
2015	8%	19%	27%	46%	2015baseline				
2016	25%	22%	31%	23%	2015to2016	17%	3%	4%	-23%
2017	20%	18%	28%	34%	2016to2017	-5%	-3%	-3%	12%
2018	17%	24%	28%	31%	2017to2018	-2%	6%	0%	-3%
Mean	18%	21%	28%	33%	Mean	3%	2%	0%	-5%
SD	7%	3%	2%	10%	SD	12%	4%	3%	17%
	STK					**▲ STK**			
2015	2%	25%	28%	45%	2015baseline				
2016	5%	25%	29%	42%	2015to2016	3%	-1%	1%	-3%
2017	8%	36%	28%	29%	2016to2017	3%	11%	-1%	-13%
2018	5%	32%	27%	36%	2017to2018	-3%	-3%	-1%	8%
Mean	5%	29%	28%	38%	Mean	1%	2%	0%	-3%
SD	3%	6%	1%	7%	SD	4%	8%	1%	10%

In 2015, the distribution of pH changed from 54% of fields sampled in pH Index 1 and 2 to 33% of fields by 2018. Fields in Index 3 for pH changed from 35% to 26% between 2015 and 2018, and fields in Index 4 changed from 11% to 41% over the same period. The distribution of STP changed from 27 to 41% in index 1 between 2015 and 2018. There was virtually no change in P Index 3 fields (27 to 28%) but Index 4 fields changed from 46 to 31%. The distribution of fields in K Index 1 and 2 changed from 27 to 37%, again almost no change occurred in Index 3 (28 to 27%) but Index 4 changed from 45 to 36%. The mean changes in distribution are shown in Figure 5-5. Overall, the distribution of pH showed a decrease in Index 1, 2 and 3 and an increase in Index 4. The distribution of P and K showed a decrease in Index 4 and increase in Index 1 and 2, with no change in Index 3.

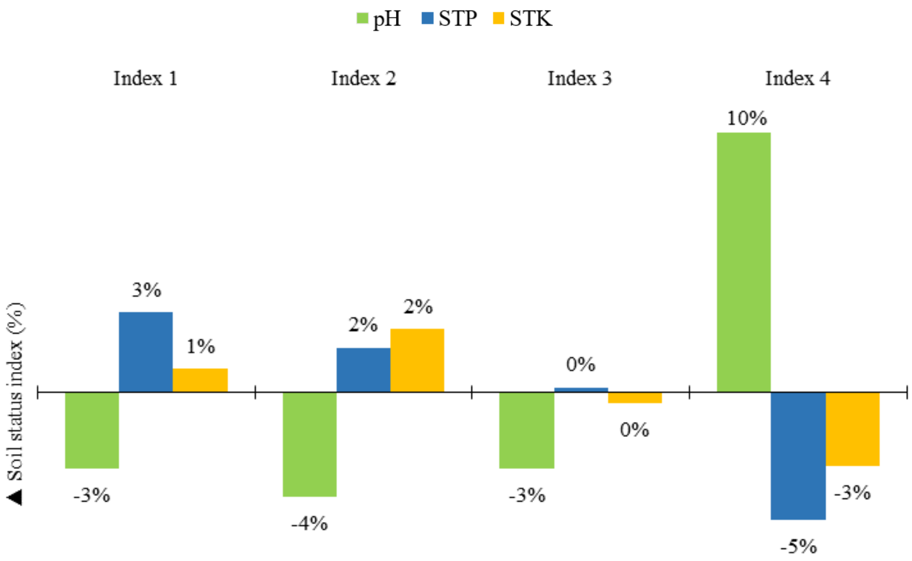

Figure 5-5. Change in proportion of fields in each Index for pH, P and K, over the study period 2015 to 2018.

5.5.6 Factors affecting change in soil fertility

There was no significant model when the dependent variables (soil pH, STP and STK) were considered jointly, Wilk's $\lambda = 0.87$, F (3,35) = 1.75, $p > 0.05$, partial $\eta^2 = 0.13$. When considered independently however, the overall regression model for change in soil pH was significant, F (62,0.031) = 3.02, $p < 0.05$, $r^2 = 0.84$. The model was also significant for STP, F (62, 4.62) = 3.02, $p < 0.05$, $r^2 = 0.89$ and STK, F (62, 351.98) = 2.183, $p < 0.05$, $r^2 = 0.79$.

5.5.7 Change in soil pH as affected by soil, environmental and management factors

Change in soil pH was significantly affected by SOM ($p < 0.05$), which also contributed the largest towards explaining the variation in the regression model ($\eta^2 = 0.16$). There was a significant interaction between gradient class, WC and reseed ($p < 0.05$, $\eta^2 = 0.13$). Within this interaction, the largest change in soil pH occurred on fields that were level, had WC present in the sward, and were reseeded between 5 and 10 years ago (0.25, Standard Error (SE) 0.06). Whereas the least change in pH was on fields with a gradient, had no clover present in the sward, and were reseeded less than five years ago (0.12, SE 0.08). An interaction between cropping, WC and reseed ($p < 0.05$, $\eta^2 = 0.12$) also contributed significantly to explaining variation in the model. For this interaction, the largest change in pH was in GO fields with WC present and were reseeded less than 5 years ago (0.25, SE 0.06). The smallest change was in G+1H fields, with no clover present, and were reseeded less than 5 years ago (0.33, SE 0.06). There was also a significant interaction between gradient class and cropping ($p < 0.05$, $\eta^2 = 0.11$). In this case, level fields used for GO had the largest change in pH (0.19, SE 0.04), while level fields used for 2CS had a negative change (-0.02, SE 0.18). Finally, mean lime import contributed significantly to explaining variation in the change in soil pH ($p < 0.05$, $\eta^2 = 0.10$).

5.5.8 Change in STP as affected by soil, environmental and management factors

The largest significant contribution to a change in STP was Al ($p < 0.05$, $\eta^2 = 0.42$) (Figure 5-6). Additionally, a significant interaction was found between WC and reseed ($p < 0.05$, $\eta^2 = 0.37$). In this interaction, the largest change in STP was on fields where WC was absent and were not reseeded in over 10 years (0.60 mg l^{-1}, SE 0.30). Whereas there was a negative change in fields where WC was absent and were reseeded in less than 5 years (-1.83 mg l^{-1}, SE 0.49). There was also an interaction between WC and cropping ($p < 0.05$, $\eta^2 = 0.24$). Within this interaction, G+1H fields where WC was absent had the smallest negative change in STP (-0.11 mg l^{-1}, SE 0.35). While 2CS fields where WC was absent had the largest negative change (-2.53 mg l^{-1}, SE 1.87). There was also an interaction between gradient class, WC and reseed ($p < 0.05$, $\eta^2 = 0.18$). For this interaction, fields with a gradient, where WC was absent and were reseeded over 10 years ago had the largest positive change in STP (1.07 mg l^{-1}, SE 0.43). Fields that were level, where WC was absent and were reseeded less than 5 years ago had the largest negative change in STP (-1.90 mg l^{-1}, SE 0.66). Mean lime import ($p < 0.05$, $\eta^2 = 0.21$), SOM ($p < 0.05$, $\eta^2 = 0.20$) Mg ($p < 0.05$, $\eta^2 = 0.18$) and gradient class ($p < 0.05$, $\eta^2 = 0.12$) were significant contributions to variation in change in STP. For gradient class, change in STP was greatest on level fields (-0.93 mg l^{-1}, SE 0.22) compared to fields with a gradient (-0.51 mg l^{-1}, SE 0.17).

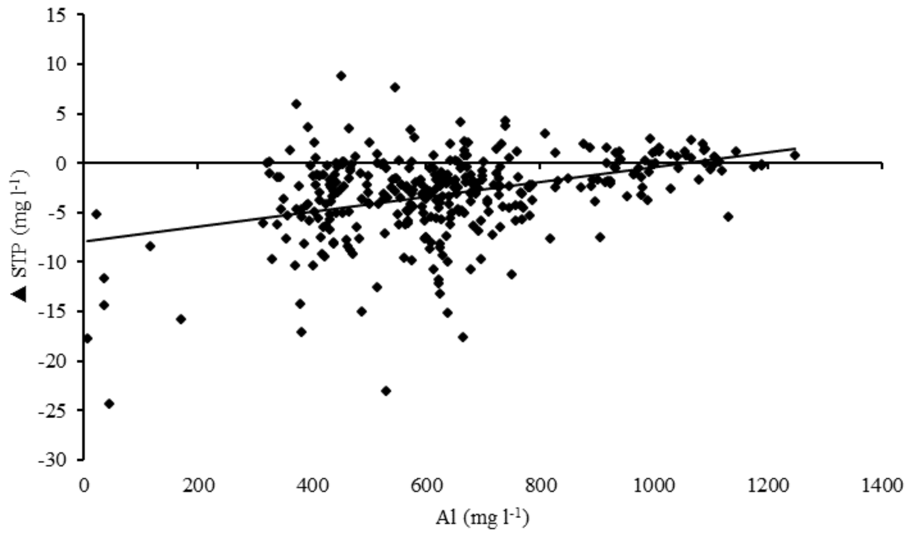

Figure 5-6. The relationship between change in STP and Al levels in the soil was significant, $y = 0.0075x - 7.9244$ $r^2 = 0.1535$, $p < 0.05$. Change in STP decreased as Al levels in the soil increased.

5.5.9 Change in STK as affected by soil, environmental and management factors

An interaction between WC and reseed was the largest significant contributor to variation in change in STK ($p < 0.05$, $\eta^2 = 0.23$). In this interaction, fields where WC was absent and were reseeded less than 5 years ago, had the largest negative change in STK (-16.04 mg l^{-1}, SE 6.67). Whereas fields with WC present and were reseeded between 5 and 10 years had the largest positive change in STK (1.49 mg l^{-1}, SE 5.03). Soil chemistry levels of Mg and Al had a significant effect on STK (p < 0.05, η^2 = 0.22 and p < 0.05, η^2 = 0.17, respectively). There were also significant interactions for STK between reseed and cropping (p < 0.05, η^2 = 0.18). Within this, 2CS fields reseeded over 10 years ago had the largest significant change in STK (14.97 mg l^{-1}, SE 24.61). While 2CS fields reseeded less than 5 years ago had the largest negative change in STK (-24.77 mg l^{-1}, SE 25.46). There was also an interaction between reseed, WC and PRG (p < 0.05, η^2 = 0.13). Within this, fields with high levels of PRG, where WC was present and were reseeded between 5 and 10 years ago, had the highest positive change in STK (18.93 mg l^{-1}, SE 9.23). Whereas, fields with low levels of PRG, where WC was present and were reseeded over 10 years ago, had the largest negative change in STK (-32.41, SE 17.60). An interaction was found between reseed, PRG and gradient class (p < 0.05, η^2 = 0.12). In this scenario, fields that were level, had low PRG, and were reseeded between 5 and 10 years ago had the highest positive change (13.93 mg l^{-1}, SE 12.72), but level fields with low PRG and which were reseeded less than 5 years ago had the largest negative change (-54.04 mg l^{-1}, SE 32.06). There was a significant interaction between PRG and WC (p < 0.05, η^2 = 0.20). Fields with high PRG levels and where WC was absent had a positive change in STK (3.41 mg l^{-1}, SE 5.73) compared to the same fields with high PRG but where WC was present had a negative change (-28.40 mg l^{-1}, SE 12.16). Gradient class also significantly contributed to variation in STK (p < 0.05, η^2 = 0.10). Fields with a gradient had a lower change in STK (-6.94 mg l^{-1}, SE 2.41) than fields that were level (-7.01 mg l^{-1}, SE 2.97).

5.5.10 Matching nutrient management to nutrient advice

For P nutrient management at paddock scale, it was found that P advice and P imports were significant different in each individual year 2015 to 2017 (p < *0.05*) (Figure 5-7). However, the paddock scale P advice mean (18 kg P ha^{-1}, SD 12) and P import mean (19 kg P ha^{-1}, SD 14) after three years were not significantly different (p > *0.05*) (Figure 5-7). This suggests that the paddock scale P advice and was adhered to overall. Over the three years, P advice rate was significantly lower in 2015 compared to 2016 (14 and 22 kg P ha^{-1}, respectively) (p < 0.05). P advice was significantly higher in 2016 compared to 2017 (22 and 18 kg P ha^{-1}) (p < 0.05). For P import, there was no significant difference between 2015 and 2016 (19 and 17 kg P ha^{-1}, respectively) but there was a significant increase in 2017 (22 kg P ha^{-1}) (p < 0.05). Finally, PBAL decreased significantly between 2015 and 2016 (7 and 4 kg P ha^{-1}) however there was no difference between 2016 and 2017 (both 6 kg P ha^{-1}). Overall, both P advice and P imports trended upwards, however PBAL trended downwards. For K nutrient management, it was also found that K advice and K imports were significantly different between each individual year, and after three years (p < *0.*05) (Figure 5-8). Mean K advice was significantly lower (31 kg K ha^{-1}) than K import (71 kg K ha^{-1}). The K advice rate was not significantly different between 2015 and 2016 (28 kg K ha^{-1}, respectively), however there was a significant increase between 2016 and 2017 (37 kg K ha^{-1}) ($p < 0.05$). For K import, there was a significant decrease between 2015 and 2016 (76 and 63 kg K ha^{-1}) ($p < 0.05$) and then a significant increase in 2017 (74 kg K ha^{-1}) ($p < 0.05$) (Figure 5-8). Finally, for KBAL, there was a significant decrease between 2015 and 2016 (52 and 34 kg K ha^{-1}) but there was no significant difference for 2017 (34 kg K ha^{-1}). Overall, K advice and K import trended downwards slightly, and KBAL also trended downwards. Additionally, lime import was significantly higher (3,745 kg ha^{-1}, SD 4,300) than lime advice (1,988 kg ha^{-1}, SD 2,385) ($p < 0.05$) at the end of the three years.

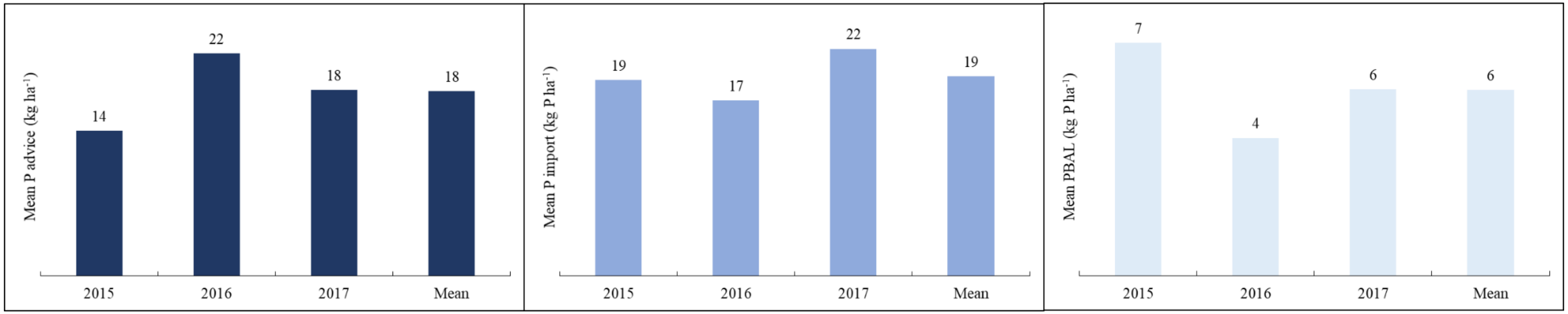

Figure 5-8. Mean P nutrient advice (application rate; kg P ha^{-1}), actual management (application rate; kg P ha^{-1}) and balance (kg P ha^{-1}) at paddock/field scale for individual three years (2015-2017) and at the end of the study (*n* = 495).

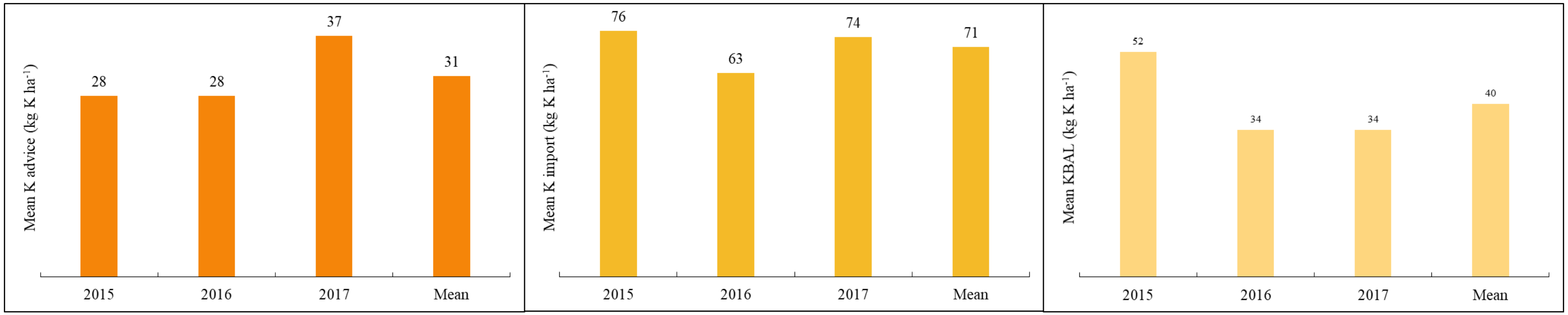

Figure 5-7. Mean K nutrient advice (application rate; kg K ha^{-1}), actual management (application rate; kg K ha^{-1}) and balance (kg K ha^{-1}) at paddock/field scale for individual three years (2015-2017) and at the end of the study (*n* = 495).

5.6 Discussion

5.6.1 Paddock scale nutrient advice, import, balance and soil fertility

It was found that paddock scale nutrient advice overall for P was adhered to, but K and lime were not strictly adhered to. This may suggest that farmers prioritised P imports over K or lime imports. For P, the mean advice rate and import rate were not significantly different (18 and 19 kg P ha^{-1}, respectively) (Figure 5-7). However, for individual years, the results suggest that there are factors that limit the capacity of a farmer to match P advice with P imports annually, for example fertiliser costs (Dillon et al., 2018). For K, it was found that K advice rate was significantly lower than K import rate (Figure 5-8) and suggests that K nutrient use is not highly influenced by factors like cost, and that there is potentially less regard given to matching K advice and imports. It can be expected that there was difficulty for participating farmers to annually omit nutrient imports on Index 4 soils for both P and K, as it is may be against individual farmer intuition, therefore reduced total import rates were likely used instead. Lime imports were not strictly matched to paddock scale advice either, however, this is likely due to the limitations of the practicalities of spreading of lime in bulk by contractors, rather than a lack of a targeted approach by participating farmers. As a result, for lime application, it is more likely that a homogenous rate of lime is used across a number of paddocks instead of individual rates per paddock. Finally, adhering to the nutrient advice was voluntary and not a requirement of the project, which may have influenced the results differently. In light of these factors however, it was still found that PBAL and KBAL surpluses decreased significantly and that soil fertility levels (STP, STK and pH), improved significantly towards optimum values during the paddock scale advice period (Figure 5-7 and Figure 5-8). Gourley et al., (2015) showed how nutrient surpluses on dairy farms lead to excessive levels of soil nutrient accumulation on pastures systems. Losses of nutrients to the environment are likely to be greater from areas with high nutrient loads and where hydrological pathways will contribute to their movement (Penn et al., 2007; Sanderson et al., 2010), and will be exacerbated by further application of fertilisers. Adding to this, Anderson et al., (1992) found that soil nutrient variability was driven by localised animal stocking rate intensity, as well as surface topography, and that accumulation of total soil P in surface soils was related to the level of organic matter loading. Furthermore, it was found that nutrient imports and balances were not significant indicators of the potential change in soil fertility, but a combination of soil chemical composition, environmental factors, sward management strategies and lime use were significant indicators. Instead, the results suggest that nutrient advice rates without information on soil characteristics (chemical composition and gradient class) and sward management (sward species composition, cropping and reseeding rate) are potentially limiting the capacity to improve soil fertility levels and spatial distribution. This is in agreement with Bacon et al., (1990) who also identified that a relationship between the magnitude, and spatial patterns of within-farm nutrient flows of a farm are related to farm management decisions after interacting with the biological processes of the soil. Also, Penn et al., (2007) showed that the use of a composite sampling strategy or an average P value for an entire field or management unit can potentially mask STP patterns and for example, result in an inaccurate estimation of P losses to surface waters. Ultimately this can limit the capacity to improve within-farm nutrient use efficiency. Overall, monitoring at the paddock scale was effective at identifying the limitations of improving soil fertility on dairy systems and may be useful as a strategy for tailoring advice rates appropriately.

5.6.2 Temporal trends in paddock scale soil fertility and changes in distribution

Long term soil fertility studies (Edmeades, 2003) highlight that, the ratio of nutrients in manures, is different from the ratio of nutrients removed by crops, which can lead to excessive accumulation of some nutrients in particular P, K and N. This can arise from the long-term use of manures (Edmeades, 2003) and is one of the factors that highlights the challenge in achieving the observed trends in this study of improved average soil pH, STP and STK (Figure 5-2, Figure 5-3, and Figure 5-4) over the course of this study. However, the trend observed in this study would be expected to lead to nutrient use efficiency benefits at the paddock scale, through better nutrient distribution, and a subsequent reduction in excessive accumulation in soil. A major positive outcome of this study, was that the Dairy subzone fields/paddocks (Table 5-6), which, for these participating farms, made up an average of 57% of the total farm area (Chapter Three) had the highest baseline STP levels, but also had the largest decrease in STP levels (Table 5-6) by the end of the study. Therefore, in this case, if sampling strategies were to focus on the dairy subzone only, there is a substantial opportunity to improve soil fertility levels for 57% of the farm area. To reliably improve the overall trend in soil fertility for pastoral systems, soil sampling strategies will need to be devised to include between paddock variability in soil nutrients, which is in agreement with Anderson et al., (1992) and Mathews et al., (1994), but also inter paddock variability as shown by Aarons et al., (2015).

The lower mean pH at the start of the study (6.02), and particularly, the low pH on the non-dairy subzone (5.86) indicates that these farms and areas within these farms, were operating at sub-optimal levels of pH, which has an economic cost and environmental risk (Oshunsanya, 2019). It also increases the risk of Al and Mn toxicity, as well as Ca and Mg deficiencies in pastures and crops, and risks sub-optimal plant-availability of a range of nutrients (Oshunsanya, 2019). It requires only a small decrease in pH to enable a large increase in Al (Brady and Weil, 2008). By the end of the study period, the mean pH was 6.33 for dairy paddocks, 6.1 for non-dairy and 6.5 for silage fields. This is close to the optimum for pastures on mineral soils (6.3). At this pH level, it is expected that the optimum response from fertiliser nutrients is achieved as well as ensuring optimum soil nutrient availability to plants (Šimek et al., 2002), which is a positive outcome for the farms in this study.

For STP, which was 9.4 mg l^{-1} (Index 4; in excess of optimum) in the baseline year of the study (Table 5-6), suggests that areas of these farms were operating at high levels of STP. High STP indicates a legacy of annual nutrient surpluses and poor distribution of P within the “farm gate” and indicates an elevated nutrient source pressure. The environmental risk associated with soils with excessive levels of STP are losses to water in run off and leaching (Schoumans and Groenendijk, 2000; Schröder and Neeteson, 2008). The longer fields are kept at high STP, the greater the risk of P entering waterways (Spears et al., 2003). Both nutrient management advice and the nitrate regulations (S.I. No. 605 of 2017, EU, 2017) are geared towards lowering P concentrations in high P (Index 4) soils so that they should drop to a lower index over time. The nitrate regulations prohibit application of fertiliser P (inorganic and organic) to Index 4 soils in most circumstances. Optimum agronomic advice would be to apply no fertiliser P to Index 4 soils until P concentrations have dropped to Index 3. The decreased mean STP from excessive levels to within the optimum range, suggests that farms had improved P management, to some degree through paddock scale advice, and that the risks associated with high STP had decreased as a result, over the study period.

The high mean STK level at the baseline year of the study (Table 5-6), indicates that there was an economic loss by over supplying K to the soil on parts of these farms. Sustained high levels of K also increase the risk of boron deficiency in plants and well as hypomagnesemia in ruminants due to high potassium intake relative to calcium and magnesium intake (Wilkinson and Stuedemann, 1979). The change to a slightly lower mean STK, within the optimum range, suggests improved K management over the study period.

Another positive outcome of the paddock scale advice was the change in distribution of soil indices, to agronomically acceptable Index 3 for P and K, and Index 3 and 4 for pH (Table 5-7). It is likely that the colour coded paddock scale maps helped influence the nutrient distribution strategy for individual farms. One challenge to achieving the distribution of nutrients in relation to soil fertility, is that the strategy in which farmers manage their grazing herds movement will influence the mean soil fertility as well as its spatial variability. For example, inter paddock variability in soil K levels, can be influenced by the tendency for dairy cows to congregate and excrete urine and manure near gateways (Sanderson et al., 2010). On top of this, the efficient of K recycling by cows can be up to 91% of their dietary K intake (Anderson et al., (1992); Kayser and Isselstein, (2005)). These contributing factors would lead to an accumulation of K anywhere where animals congregate. Penn et al., (2007) also observed hotspots of P in dairy soils, and attributed this to deposition of excreta from cows. Sanderson et al., (2010) also observed higher nutrient levels in concentrated areas, compared with unaffected parts of farm pastures. In intensive forms of rotational grazing, strip grazing should improve nutrient use efficiency in grazing systems (Oenema et al., 2003a), however it would be expected that the overall nutrient source pressure is greater also due to higher imports and rate of urine and dung deposits per area. The high Ca, Mg, Na and K concentrations in dung and urine (During and Weeda, 1973), and the luxury uptake of K by pasture (Kayser and Isselstein, 2005) can be expected to largely influence the disproportionate distribution of soil fertility levels overall. Thus, correcting nutrient imports through targeted advice, to improve soil fertility distribution, will also require the careful management of stocking rate density and grazing strategy over a long period (Petersen et al., 1956; Petersen, 1995).

When mean results of this study are compared to the large sample size study on dairy farms soil fertility in Ireland (Teagasc, 2018), for the same years (2015 to 2017), it was found that paddock scale distribution had a lower proportion of Index 1 and 2 soils for both P and K, as well as a higher proportion of Index 4 soils. Although the farms selected from this study may not be representative of national scale soil fertility it does highlight the importance of assessing soil fertility at different scales, and may be a factor in the paradox between soil fertility levels and declining water quality (Kelly et al., 2014). Notably, the proportion of Index 4 fields for both P and K were declining between 2015 and 2017 for this study but were almost static at national scale. The fact that the distribution of pH was changed from 54% of fields in Index 1 and 2 to 34% over the four years suggests that farmers responded by using the paddock scale information in the baseline year by at least decreasing the proportion of the farm area that was in sub-optimal pH status. In relation to STP and STK, it was shown that the largest changes to occur across all indices was in the decrease of Index 4 fields (-5 and -3%, respectively), which suggests there was a response from farmers to at least decrease the proportion of farm area that was in soil nutrient excess.

5.6.3 Drivers of change in soil fertility

Although paddock scale nutrient advice was not strictly adhered to (except for P), the significant influence of soil characteristics and sward management factors, indicate that achieving optimum soil fertility may actually rely on matching the advice to these environmental and management factors first. Currently, matching nutrient imports to nutrient advice based on stocking rate and estimated offtakes may not be sufficiently accurate to sustain increases in soil fertility levels. It was expected that the paddock scale colour coded maps were a driver of change but there was not quantitative assessment of this. Despite this, the following were measured drivers.

5.6.3.1 Soil chemical properties

It was found that SOM was the largest significant driver of the capacity to change soil pH, and a significant factor for change in STP also. This was likely because SOM levels influence nutrient availability, moisture levels, and aeration as well as pH buffering capacity (Brady and Weil, 2008). The higher organic matter soils (for example, as levels increase towards > 30%, which are classified as peat soils (Fay, 2007)) are expected to have a stronger “resistance” to change in soil pH (Parfitt et al., 1995). The influence of SOM can be expected as the grazed pasture-based dairy system adds organic matter through excreta application and animal deposition. In agricultural regions with high rainfall levels, soils can quickly become acidic due to the formation of carbonic acid (H2CO3), but also because of the use of ammonium-based fertilisers (nitrification process releases additional H+) (Brady and Weil, 2008). This increase in acidity emphasises the need to regularly use lime for its addition of Ca and Mg to maintain optimum soil pH on farms. The importance of SOM in grassland soils and nutrient management was also highlighted by Daly et al., (2015) who showed that soils high in organic matter, exhibited low P sorption and storage capacities, which suggests that Irish mineral soils may not be suitable for heavy applications of manure or fertiliser P.

Accumulation of total soil P in surface soils has been shown to be related to the level of organic matter loading and has also been associated with other soil characteristics that influence the release of P such as total Al, Ca, and soil pH (Anderson et al., 1992). As Al was found to be a significant contributor to variation for change in STP (Figure 5-6) this would suggest that there is a need to consider the impacts of Al “lock up” during nutrient management planning, at this scale (Hocking et al., 2000).

5.6.3.2 Environmental factors

Gradient class was a significant factor for the capacity to change in pH, P and K, on its own and/or as part of an interaction. Gradient and nutrient management are typically associated with increased soil erosion risk and increase nutrient surface run off, leading to eutrophication (particularly P) of waterways (Correll, 1998). Li et al., (2006) showed that rainfall intensity had a significant influence on the nutrient concentrations in runoff flow on sloping lands. It was also found that slope length influenced the nutrient loss of soil and that, as the slope gradient decreased, the nutrient loss also decreased, because of the increase in infiltration. Chen et al., (1999) also found that the loss of P has a strongly positive relationship with rainfall intensity. Gburek and Sharpley (1998) found that the P loads leaving from two small drainage basins with 3 to 17% and 1 to 20% of gradient respectively, were positively proportional to the rainfall intensity. Pastures tend to have lower levels of erosion than for example tillage, which can have frequent cultivations and periods of prolonged bare soil, which adds to the risk of soil loss (McDowell et al., 2006). Therefore, it can be expected that losses of nutrients can be relatively greater on fields with a gradient compared to level fields. The nutrient loss also correlates to the nutrient content in the soil. As soil P increases the particulate and dissolved forms of P, along with runoff, will increase (Sharpley et al., 1993; Sharpley et al., 1995). As a result, soil fertility requirements and nutrient management strategies should be different on fields with and without significant gradients. In order to carry out the nutrient management practices of paddock scale management units, a new management approach based on the basic knowledge of sloping lands should be designed, for example through targeted fertilisation rates (Li et al., 2006).

It was expected that soil drainage class would be a significant factor also, as shown by several other studies (Van Beek et al., 2009; McCormick et al., 2009; Withers et al., 2001; Djodic et al., 2004). However, the results of this study suggest that sward management factors have a more significant impact on change in soil fertility levels than drainage classes at paddock scale.

5.6.3.3 Management factors

It was found that sward management factors, sward species (PRG and WC), cropping and reseed all contributed significantly to changes in soil pH, STP and STK. Notable results included; soil pH and STP tending to have larger negative changes where silage harvests occurred i.e. greater pasture offtakes (and therefore nutrients); both STP and STK tended to have larger negative changes where recent reseeding had occurred (< 5 years ago), suggesting that there are nutrient losses from the soil from reseeding practices, potentially due to the change in soil structure where cultivation is carried out; and species composition of WC tended to be a greater driver of change in STK than of change in STP or pH. These results suggest that individual sward management factors for each paddock are important considerations for the improvement of soil fertility, when developing nutrient advice. Garden et al., (2001) and Sanford et al., (2003) have shown that soil fertility is a key factor that can influence differences in species composition. Although not significant in this study, soil type can be an important factor affecting species distribution too (Guppy et al., 2013). Garden et al., (2001), also highlighted that rainfall, nutrient imports and stocking rates are also important factors in species distribution. Hill et al., (2005) found that pasture species varied considerably in the response to soil P levels. The importance of accounting for sward species composition is that some species will require different nutrient management strategies, for example, a lower P import (Sanford et al., 2003). Over fertilising on some swards increases the risk of P loss and leads to expensive and unnecessary nutrient surpluses at paddock scale.

The interaction between reseeding and cropping was significant and suggests that reseeding strategies should be considered in the context of cropping rates. However, Behrendt et al., (2013) showed that when soil fertility is high, ‘tactical grazing rests’ for the sward can be more profitable than reseeding a pasture. Their results also indicated that pastures under a consistent high soil fertility management can maintain higher stocking rates and have a capacity to recover from low levels of perennial species in the sward given adequate grazing rest (Cook et al., 1978; Crofts, 1997; Garden et al., 2001). With low soil fertility and lower stocking rates, pasture reseeding can be a better decision (Behrendt et al., 2013) however, considerable thought should be given to costs of reseeding (Scott et al., 2000). Achieving optimum soil fertility levels with regard to sward management factors (species, cropping and reseeding) may be best managed on a paddock by paddock basis, which is in agreement with (Dowling et al., 2005). In relation to cropping strategies, paddocks routinely harvested for silage and or hay typically show lower soil K levels (Mathews et al., 1994). Gourley et al., (2015) describes statistically significant relationships between soil P and K levels and grazing management characteristics also. Aarons et al., (2015) showed that the frequency of feeding forage and pasture harvesting highlight that some generic management practises can exacerbate elevated soil nutrient concentration within grazed dairy farms and have direct implications to farmers and advisors.

It was expected that nutrient management (imports and balances) would have a significant contribution to change in soil fertility. However, only lime import was a significant factor, and its effectiveness for changing soil pH are well documented (Curtin and Syers, 2001; Peoples et al., 1995; Helyar and Anderson, 1970; Wall et al., 2018). Nutrient imports and balances have a significant role in the total nutrient that is being cycled in a farm system, but soil characteristics, sward management and lime determine whether the actual total nutrient import meets the individual paddock recommended nutrient total. This difference between the total nutrient import advised and the actual nutrient total reaching the soil is likely contributing to the difficulty of improving soil fertility effectively and leading to the poor distribution of soil indices. An absence of planning for the gains and losses of nutrients due to individual paddock scale characteristics (for example with a gradient, reseed rate, and cropping rate) may be a large barrier that is limiting the achievement of optimum soil fertility levels on grazed grass dairy farms.

5.7 Conclusion

Overall, during the paddock scale advice period, the soil fertility levels and the distribution of soil fertility indices improved towards more ideally optimum values, (mean paddock scale pH increased to 6.3 or Index 3 (agronomically optimum level), soil test P and K decreased to 7.3 and 139 mg l^{-1}, respectively (i.e. from Index 4, high/sufficient to Index 3, agronomically optimum). Paddock scale P and K nutrient balance surpluses decreased significantly between 2015 to 2016, from 7 to 4 kg P ha^{-1} and 52 to 34 kg K ha^{-1}, respectively. As a result, it is has been shown that there are potentially significant opportunities to further optimise P and K management on dairy farms at paddock scale, which is in agreement with Gourley et al., (2015). However, it was found that with the exception of P, advised nutrient application advice were not strictly matched by actual application rates. There are likely to be numerous reasons for mismatches between advice and actual management. For example, it may not have been controllable for farmers to completely omit nutrient applications on index 4 – high/sufficient nutrient level paddocks due to restricted organic manure storage capacity and restricted area for spreading. The factors influencing farmer decision making on organic manure applications should be explored further. Additionally, of course, it was not a requirement to follow the nutrient advice rate. Despite this, the converging trend of improving soil fertility levels and decreasing nutrient balances during the paddock scale nutrient advice period was a positive outcome for these commercial dairy farm enterprises.

It was found that changing soil fertility levels will depend, to a large degree, on matching the advice to significant soil characteristics and sward management factors on a paddock by paddock basis, rather than generating a total nutrient import rate for the whole farm, for example, based on average stocking rate nutrient load and total concentrate feed imported. The improvement in soil fertility and nutrient management strategies was shown to be influence by paddock scale soil characteristics that are outside of the farmers control, but this study showed that the change in soil fertility can be improved through farmer decision making in relation to sward management practices and lime application, in particular. Perhaps one of the barriers to improving soil fertility is that advice may not sufficiently consider the implications of these soil and sward factors. Due to the high spatial variability in soil nutrient indices at paddock scale, traditional methods of sampling (e.g. large sample areas 4 ha or greater) may not be effective for monitoring soil fertility or guiding nutrient management advice. A greater understanding of farm nutrient fluxes and expected patterns of within-farm nutrient distribution, complemented by comprehensive soil testing, might help guide more effective and environmentally beneficial nutrient management decisions. Targeted data based grouping of paddocks according to management (for example, by subzone, such as the milking platform, or silage ground, or by reseeding rate) for soil sampling could be a cost-effective approach to minimise sampling costs Gourley et al., (2004), and should be explored further.

CHAPTER SIX: HERBAGE NUTRIENT STATUS AT PADDOCK SCALE AS AFFECTED BY SOIL AND SWARD MANAGEMENT FACTORS ON GRASS BASED DAIRY FARMS

Preface: This paper has been prepared for the submission to the journal *Grass and Forage Science*. Formatting and reference style have been edited for consistency throughout the thesis. Figure and table captions have been assigned with a chapter prefix. Acknowledgements have been removed.

Keywords: Seasonality, Paddock Scale, Herbage Nutrient Status, Soil Fertility

6.1 Abstract

Natural levels of pasture growth in temperate regions (e.g. Atlantic Central Europe, New Zealand and the eastern coast of Australia) offers an advantage for ruminant and milk production systems, by enabling the potential to utilise pasture as a relatively inexpensive feedstuff. Climatic factors, such as medium/high precipitation levels and cool winter temperatures, provide long pasture growing seasons and as a result, the animal’s diet in these regions can be predominantly made up of herbage dry matter. The nutrient status of herbage therefore can affect overall productivity and profitability of the ruminant or milk production system. An understanding of seasonality patterns in pasture growth is therefore required, to enable farmers to better match nutrient concentration of the pasture, to the feed requirements of livestock. Furthermore, an understanding of the factors that can influence herbage nutrient concentration, is essential for improving decision making for livestock breeding and sward management. Herbage nutrient status studies are commonly carried out at plot/field or research farmlet scale. These studies typically control many of the environmental or management factors, and make assumptions or use coefficients to represent factors that were not measured. Fewer studies on herbage quality have been carried out across the full spatial spectrum of commercial grass-based dairy farms, where actual environmental and management factors that could affect herbage nutrient status interact. In Ireland, specialised dairy farms are typically the most intensively managed in terms stocking rate, compared to other livestock systems, and rely predominantly on a pasture diet.

The aim of this study was to characterise herbage nutrient status (herbage concentration level of nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), copper (Cu), manganese (Mn) and zinc (Zn)) at paddock scale (ca. 0.5 – 2.5 ha) on commercial dairy farms. Additionally the aim was to evaluate the impact of seasonality (spring, summer, autumn), soil fertility (soil test pH, soil test P (STP), K (STK) and Magnesium (STMg)), soil characteristics (chemical properties, drainage and gradient), nutrient management (N, P and K balance and use efficiency), and sward management (species composition, grazing rotations, harvest events and cropping rate) on herbage nutrient status. It was hypothesised that herbage nutrient status would be significantly affected by 1) season, 2) soil fertility and nutrient management, and 3) soil characteristics and sward management factors. A total of 21 commercial dairy farms were selected from the south and south east of Ireland. Repeated measure (within subject) analysis was used to identify significant changes in herbage nutrient concentration between season and year. Multiple stepwise linear regression was carried out to investigate the relationship between dependent and

independent variables. Seasonality was analysed on 21 farms (n = 552 paddocks) but all other analysis was carried out on 15 farms (n = 361 paddocks), due to data constraints.

It was shown that herbage nutrient concentration was most variable (denoted by CV: Coefficient of Variation) in spring (CV 30%), and slightly less variable in both summer and autumn (CV 26% and 27%, respectively). Mean herbage N, P, Ca and Mn were within range of their respective sufficient status level (33.0, 3.6, 5.0 g kg^{-1} and 85.0 mg kg^{-1}, respectively) in each season, but mean herbage K, Mg Cu and Zn were slightly lower than the sufficient status (26.4, 1.8 g kg^{-1} and 8.8, 28.2 mg kg^{-1}, respectively), for each season. Overall, herbage P, K and Mg increased in concentration from spring to summer and into autumn also, herbage N, Cu and Mn decreased from spring to summer, but increased in autumn, and herbage Ca and Zn increased from spring to summer but decreased in autumn. It was shown that the typical species composition for paddocks (n = 461) in this study was > 75% Perennial Ryegrass, with 10-25% White Clover, and 10-25% of other species, which were most likely to be Broad Leaf Dock, Creeping Buttercup and Annual Meadow grass. It was also shown that paddock scale herbage nutrient status has a strong relationship with soil nutrient supply and therefore, soil fertility management. STK level was the main driver of herbage K and Ca (η^2 = 0.26 and 0.11, respectively) while STMg level was the principal driver of herbage Magnesium (η^2 = 0.28) and STP level was the key driver of herbage P (η^2 = 0.35). Soil pH level was the main driver of herbage Mn and Cal (both η^2 = 0.15). Additionally, the total number of grazing rotations was significant for herbage P (η^2 = 0.15), while the total number of harvests was significant for herbage N and K (η^2 = 0.15 and η^2 = 0.22). Results of this study highlight that variability and seasonal trends in herbage concentration at the paddock scale, particularly in spring and autumn, could have implications for decision making in livestock breeding strategies and sward management. Low concentrations of herbage K, Mg, Cu and Zn in spring and autumn could risk dietary disorders or limit herbage production at these times of the year. Herbage seasonality may also have important implications for nutrient management and soil fertility, such as decision making relative to lower herbage P and K levels in spring, and higher levels in autumn. This highlights the importance of optimum soil fertility levels for spring for example, it suggests also, that there may be a need to soil or herbage sample for Cu and Zn at these times of the year to ensure they are at adequate levels. The strong relationship between soil fertility and herbage nutrient status may indicate a need to soil test regularly (e.g. 2-3 years), or potentially to take seasonal herbage samples to guide nutrient management, particularly for micro nutrients in the early and late part of the growing season. Sward management strategies i.e. grazing rotations and harvest events in the previous growing season may be important for herbage nutrient status in the following season also. Overall, the paddock scale analysis provided some insight into the seasonal variability and the factors that drive it in a commercial pasture base dairy farm scenario, and may be provide insight into supporting the development of future pasture and soil management strategies, or contribute towards validating studies at different scales such as plot, farmlet or modelling/simulation.

6.2 Introduction

Agriculture in temperate regions (e.g. Atlantic Central Europe, New Zealand and the eastern coast of Australia) offers an advantage for ruminant or milk production systems, by enabling the potential to utilise pasture as a relatively inexpensive feedstuff (Jones, 2013). Such regions typically have intensive dairy systems to economically capitalise on this advantage. Climatic factors, such as medium/high precipitation levels and cool winter temperatures (EU, 2011), provide long pasture growing seasons and as a result, the animal's diet in these regions can be predominantly made up of herbage dry matter. The nutrient status of herbage therefore can affect overall productivity and profitability of the ruminant production system. (Shalloo et al., 2018). An understanding of seasonality patterns in pasture growth is therefore required, to enable farmers to better match nutrient concentration of the pasture, to the feed requirements of livestock. The herbage nutrient concentration of any pasture should ideally meet the mineral and carbohydrate requirements of the grazing animal, in order to, minimise the risk of dietary disorder for the animal, especially when there is supplementation of the diet with concentrate feeds for example (Roche et al., 2009b; Pearson and Ison, 1997; Lambert and Litherland, 2000). Ireland is part of the high rainfall temperate region of the European Atlantic central (EU, 2011), which enables grass production almost all year round. Pasture-based dairy systems in Ireland use rotational grazing of paddocks on the farm, typically ranging from 18 to 18 days, depending on the time of the year. This rotational grazing system is used to provided sufficient herbage biomass and for managing the nutrient status of herbage consumed by the grazing dairy herd. The goal is to move the cows in a cycle (or rotation) to keep a constant supply of fresh herbage, high in nutritive value, throughout the growing season (Machado et al., 2005). In this system, cows calve in late winter or early spring and access to pasture begins in February-March, when soil temperatures and light intensity start to increasing (Jones, 2013). Concentrate supplementation of the dairy cows is provided when pasture availability or nutrient status becomes low. However, fresh grass is the main source of dairy cow nutrition until the end of the growing season. During the winter period, cows are typically dried off and fed on a diet of conserved forage (grass silage) and some concentrate supplementation is introduced into the diet again. Growing season length varies depending on regional weather differences and soil trafficability, can vary considerably from year to year. In the south and east regions of Ireland there are between 236 and 242 grass growing days while in the west and north, the mean number of grass growing days is 210 days (NFS, 2017). On average, the eastern side of the country gets between 750 and 1,000 mm of rain annually, and mean temperatures can range between 9 and 10 ^{o}C (Met Éireann, 2019). Variability in weather plays a large role in the capacity to produce quality herbage, and creates a challenge for meeting the requirements of the dairy cow consistently.

Research has shown that variation in herbage nutrient concentration i.e. nutrient status in the medium to long term can be affected by a legacy of soil nutrient management (Reijneveld et al., 2014; Frame et al., 2002), but in the short-term, can be affected by season, soil (fertility and drainage) and sward factors (species composition, cropping and reseed rate) (Saul et al., 1999; Metson et al., 1979; Shakhane et al., 2013). The degree to which these factors have been shown to influence herbage nutrient status varies between studies. For example, the significance of soil nutrient management or species composition on herbage nutrient status may be different depending on the past soil fertility levels or species composition of the grass sward. The influence of sward management factors varies too. There is increasing interest in multi-species swards as a means of reducing environmental risks associated with pasture production (Murphy et al., 2018; McCarthy et al., 2019), while also improving production performance from ruminants (Grace et al., 2018). Herbage nutrient status is expected to be influenced by changes in fertiliser regulations, and due to knock on effects, on soil fertility and grass production potential of individual fields or farm. Soil fertility and herbage nutrient status have been found to be closely correlated (Reijneveld et al., 2014) with most studies carried out at plot/field or farmlet scale (Roche et al., 2009a; Lawrence et al., 2017) and subsequently modelled under specific physical, chemical or environmental conditions in an effort to reflect actual dairy systems (Jouven et al., 2006; Phelan et al., 2014). These studies typically control for many of the environmental or management factors by making assumptions in the analysis. On working farms there are fewer controllable variables and many factors can interact and effect herbage nutrient status. It's more difficult for modelling approaches to accurately predict the true relationship between soil fertility and herbage nutrient status. Farmlet studies, typically carried out on research farms, are more similar to working farms but they often have legacy effects of research advice and they only use a portion of the farm systems, which does not capture the full management constraints and practicalities of the whole farm system. In Ireland, specialised dairy farms are typically the most intensively managed in terms stocking rate, compared to other livestock systems, and rely predominantly on a pasture diet. This study aimed to investigate the relationship between soil fertility and herbage nutrient status across the full spatial extent of the farm system at the paddock scale (ca. 0.5 – 2.5 ha) and to identify the effects of soil and sward management, and season on herbage nutrient status. The objective of this study was to characterise the herbage nutrient status across the entire growing season, but more discretely to measure the variation in herbage nutrient concentration in spring, summer and autumn. It was also set out to understand how soil (fertility and drainage) and sward management factors (number of species, cropping rate and reseeding rate), affect herbage nutrient status at the paddock scale. The hypotheses were as follows; 1) Herbage quantity and nutrient status will differ significantly between seasons, 2) herbage nutrient status will be significantly affected by soil fertility levels 3) herbage nutrient status would be significantly affected by soil and sward management factors.

6.3 Materials and Methods

A summary of the materials and methods used is provided here. Further details of materials and methods can be found in Chapter Three, Four, and Five.

6.3.1 Farm selection and farm characterisation

A total of 21 farms were selected but only 15 had sufficient data for all aspects of analysis, these 15 are characterised below. The farms were located in the south and south east region of Ireland across counties Cork, Tipperary and Wexford. These regions are typically associated with long annual growing seasons (~242 days) and a high proportion of well drained soils (NFS, 2017). Average Total Utilisable Agricultural Area (TUAA) was 53.1 ha (includes total grassland and crop area), which was slightly below the national average of 56.0 ha for dairy enterprises in Ireland (NFS, 2017). Average stocking rate was 2.9 LU ha^{-1}, which was higher than the national average of the previous three years: 1.75 LU ha^{-1} in 2012 to 2014 (Butler et al., 2017). Average milk solid production (fat plus protein) was 806 kg ha^{-1}, which was above the national average (649 kg ha^{-1}) for the 2012 to 2014 period. Concentrate feed imported onto the farm was, on average, lower on these farms (847 kg cow^{-1}), compared to the national average (926 kg cow^{-1}) (Butler et al., 2017). Ten out of the fifteen farms in this study were on average above the 170 kg organic N ha^{-1} restriction on stocking rate (SR) and were, therefore, in nitrate derogation conditions. Some farms crossed over the derogation threshold during the study period. Ten farms were on well drained soils (Sandy loam), three farms on moderately drained (Clay loam), and two farms on poorly drained (Clay).

The grassland area of each farm was subdivided into Dairy, Non-Dairy and Silage subzones, within which, the individual fields and paddocks were monitored. ArcGIS® software (ArcGIS, 2016) was used to map individual paddock, field and farm boundaries. Bing maps aerial photography was used as a base-map (Bing™ Maps, 2005). Paddocks on the Dairy sub-zone ranged from 0.7 to 2.6 ha and paddocks and fields on the Non-Dairy and Silage subzones ranged from 1.1 to 6.5 ha.

6.3.2 Meteorological data

Meteorological data was acquired from one centrally located station (Moorepark Food Research Centre) to get a representative weather pattern for all farms across both regions, the south and south east of Ireland, for 2015 and 2016. Annual and seasonal weather data, precipitation (mm), soil temperature (°C at 100mm depth), and growth rate (kg DM ha^{-1}) were all acquired from this station (Met Éireann, 2019). For the purposes of this study, seasons were defined as Spring (February to April), Summer (May to July), Autumn (August to October) and winter (November to January), to coincide with a typical grass growing season in these regions (February to October) and the events of a rotational grazing system, Table 6-1.

Table 6-1 Grass management targets relative to mean growth rates during a typical growing season. Adapted from O'Donovan and McEvoy (2016).

Growth (kg DM Day^{-1})	Month	Event	Season
9.0	Feb 15th	Cows out to grass by day only	Spring
37.6	Mar 15th	Cows out day and night	
88.2	May 10th	Grass supply exceeds demand	Summer
68.5	June to August	Maintain grass cover per cow	
65.0	Aug 15th		Autumn
51.0	September 1st		
37.1	September 15th	Peak grass cover achieved	
30.0	October 1st		
26.8	October 15th	First paddock closed	
15.0	November 1st	Supplement introduced	Winter
8.5	Nov 15th		
2.7	November 22nd	Cows housed day and night	

6.3.3 Herbage sampling regime

Herbage sampling was carried out between mid-March and April-end (spring), mid-June and July-end (summer), and mid-September to October-end (autumn). No herbage sampling was carried out in winter. For sampling methodology, an adaption of the approach taken by Sheridan et al., (2008) was used. A 0.5 m X 0.5 m quadrat [see Appendix 1A: Figure 9-1] was semi-randomly placed three consecutive times in a diagonal line, moving down the long axis of each paddock, to get a representative sample. Three individual samples were cut from each paddock, using a battery powered grass shears (Bosch ASB 10.8 V, GrasstecGroup), at a height of 40-45 mm from the soil surface, to simulate grazing height residual (Tuñon et al., 2014).

6.3.4 Herbage characteristics

Individual samples were weighed (wet) first, and then pooled, to make a 100 g sub sample to represent the whole paddock. The sub sample was dried at 40°C for 48 hrs to determine the sub sample dry weight. Dry matter content (DM%) was calculated as dry weight as a proportion of the 100 g sub sample (Equation 6-1). Herbage cover (kg DM ha^{-1}) per paddock was calculated using the DM% and the average weight of the individual herbage samples (Equation 6-2).

$$Herbage\ dry\ matter\ (DM\%)\ = \frac{Herbage\ dry\ weight}{Herbage\ wet\ weight}$$ **Equation 6-1.**

$$Herbage\ cover\ (kg\ DM\ ha) = \ weight\ of\ herbage\ (g)\ X\ DM\%\ X\ 40$$ **Equation 6-2.**

Differences in grazing status, and associated herbage cover and sward height, are shown in Table 6-2. Optimum grazing status covers range from 1,300-1,600 kg DM ha^{-1} or 81-120 mm. A target grazing rotation length of 18-21 days is typically used to maintain grass quality and cow performance in the May to July period (Kennedy et al., 2006).

Table 6-2. Target grazing status and associated ranges in herbage cover and sward height, as identified for rotational grazing management in Ireland (Kennedy et al., 2006)

Grazing status	Herbage Cover	Sward Height
	(kg DM ha^{-1})	(mm)
Grazed out	< 200	40-50
Recovering/immature	201 to 1,300	51-80
Optimum	1,301 to 1,600	81-120
Silage	> 1,600	120+

The dried herbage sub sample was milled through a 0.2 mm sieve and analysed for nutrient concentration using a LECO Truspec and Elemental Macro Cube instrument (LECO Corporation, 2020) for herbage N, and HNO3 digestion followed by ICP-OES for the remaining elements. The major (N, K, Ca, P, and Mg) and minor element (Mn, Zn, Cu) concentrations in the herbage were determined for each paddock. Optimum herbage nutrient concentration level for herbage major and minor elements are shown in Table 6-3.

Table 6-3. Optimum ranges for herbage nutrient concentrations. Adapted from McDonald (2011). *Adapted from Wall and Plunkett (2016)

Element	Low	Normal	High
	g kg^{-1} DM		
Nitrogen* (N)	< 20	30 - 40	> 45
Potassium (K)	< 12	15-30	> 35
Calcium (Ca)	< 2.0	2.5 – 5.0	> 6.0
Phosphorus (P)	< 2.0	2.0 – 3.5	> 4.0
Magnesium (Mg)	< 1.0	1.2 – 2.0	> 2.5
	mg kg^{-1} DM		
Manganese (Mn)	< 30	40 – 200	> 250
Zinc (Zn)	< 10	15 – 50	> 75
Copper (Cu)	< 3.0	4.0 – 8.0	> 10

6.3.5 Field environmental factor classification

A number of environmental factors that might be expected to impact herbage nutrient concentration were assessed: soil chemistry, gradient and field soil drainage class. Only a summary is provided here, further details are described in Chapter Five.

6.3.5.1 Soil chemistry

Soil chemistry analysis included, soil organic matter (SOM), Aluminium (Al), Calcium (Ca), Copper (Cu), Iron (Fe), Magnesium (Mg) and Manganese (Mn). SOM was determined by loss on ignition (LOI) (Ball, 1964) and soil elements were determined by Mehlich 3 extraction (Mehlich, 1984).

6.3.5.2 Gradient class

Fields were classified as sloping or level. Level was defined as < 10 degrees of slope and Gradient as >10 degrees of slope (Roberts and Watkins, 2014).

6.3.5.3 Field soil drainage class

Fields were classified well drained (Sandy loam), moderately drained (Clay loam) and poorly drained (Clay) (CTECO, 2010; Simo et al., 2007).

6.3.6 Field management factor classification

Field or sward management factors that might be expected to affect herbage nutrient concentration were assessed: nutrient management, soil fertility, sward species composition (including no. of species, and their proportion in the sward), cropping rate, and reseeding interval. Only a summary is provided here, further details are described in Chapter Four and Five.

6.3.6.1 Nutrient management

Nutrient imports, exports and balance results for N, P and K are described in detail in Chapter Three and Four.

6.3.6.2 Soil fertility

Each field and paddock were soil sampled for nutrient analysis in Winter 2014, 2015, 2016 and 2017. Each management unit (paddock/field) was sampled in a "W" shaped pattern. A stainless-steel soil sampler with a 100 mm length sampling cone was used to take 20 soil cores from each sampling area and were boxed and labelled accordingly. Samples were dried at 40 °C and sieved through a 2mm sieve. Morgan's extract (Peech and English, 1944) was used to measure the soil test plant available Phosphorus (STP) and soil test Potassium (STK) levels using the method described by Byrne, (1979). Soil analysis was quality checked by including one blank and ten control per run of 89 soil samples. Soil pH was measured in a 1:2 soil to water ratio and determined using a Mettler Toledo glass calomel electrode.

6.3.6.3 Sward species composition

A herbage survey was carried out in 2017 on each field to estimate the proportions of herbage species that make up the swards full composition. Using a 0.5 m X 0.5 m quadrat, it was subdivided into subsections by string (5 X 5) [see Appendix 1A: Figure 9-3]. Each subdivision represented approximately 4% of the area inside the quadrat. When the quadrat was laid down, the proportion of area (i.e. number of subdivisions) taken up by each species identified was then recorded, e.g. *Lolium Perenne* predominantly occupied 3 subdivisions, or ca. 30% of the sample area. This was repeated three times for each individual paddock. The plant species inside the quadrat were identified using Feehan et al., (2012) and Fossit, (2000). Additionally, a category was assigned to each species using the DAFOR scale (Dominant, Abundant, Frequent, Occasional, Rare) (Kent et al., 2001). For this study, the proportions of Perennial Ryegrass *(Lolium Perenne)* and White clover (*Trifolium Repens)* species were included in the statistical analysis. Perennial Ryegrass (PRG) was grouped into three different proportions: 0 to 75% PRG (Low), 76 to 90% PRG (Medium), and 91 to 100% PRG (High). White clover (WC) was grouped into two levels: WC absent and WC present. Furthermore, the total No. of species in the sward per paddock was determined.

These were grouped as: One to Two *spp.* (species)., Three to four *spp*., and Greater than four *spp*.

6.3.6.4 Grazing rotations, harvest events and cropping rate

Each field was classified into one of three cropping classes; Grazing Only (GO), Grazing + Harvest (G+H), and 2 Cut Harvest (2CH). The annual mean number of defoliations/rotations was 7 for the south east region of Ireland in 2013 (O'Leary et al., 2013). Harvests take place where planned or when pasture yield exceeds the feed demand of the dairy herd. The total number of grazing rotations and total number of harvests, which are recorded by the farmer, were included as explanatory variables. Harvesting is carried out where high grass covers occur (e.g. grass dry matter at >1,600 kg DM ha^{-1}) i.e. beyond the leafy vegetative stage, where nutritive value is at its highest (Kilcher, 1981).

6.3.6.5 Reseeding interval

Fields were grouped into three categories based on the number of years since a reseeding event occurred; *< 5 years*, *5 to 10 year*s, and *> 10 years*.

6.3.7 Statistical analysis

Data were firstly screened for outliers and data entry errors. All analysis was carried out using SPSS statistical software (IBM SPSS, 2016). Data were tested against the assumption of normality. Where data was not meeting the assumption; data was not transformed but non-parametric tests were carried out instead. Measures of centrality and variation were then determined. Repeated measure (within subject) analysis was used to identify significant changes in herbage nutrient value between season and year. Multiple stepwise linear regression was carried out to investigate the relationship between dependent and independent variables. Scatterplots of standardised predicted values verses standardised residuals were used to test if the data met the assumptions of homogeneity of variance and linearity and the residuals were approximately normally distributed. Where all assumptions were met, Wilkes Lambda was used to determine significant factors, where assumptions were not met, Pillas trace was used instead. A significance level of < 0.05 was used to determined difference between means. Bonferroni post-hoc test was used to distinguish where the significant differences occurred within a group (Pairwise comparison). The following equation was used to investigate the relationships between soil and herbage nutrient concentrations:

$$\textbf{Field scale}: \hat{Y}1 = \beta o + \beta 1X1 + \beta 1X2 + \beta 1X3 + \beta 1X4 \qquad \textbf{Equation 6-3.}$$

where $\hat{Y}1$ = herbage nutrient concentration (N, K, Ca, P and Mg, Mn, Zn, Cu),
X1 = soil fertility (soil pH, STP and STK),
X2 = nutrient management,
X3 = environmental factors; soil chemical properties (SOM, Al, Ca, Cu, Fe, Mg, Mn), gradient class (Gradient and Level) and soil drainage class (WD, MD, PD), and
X4 = management factors; sward species (PRG and WC), grazing rotations, cropping (GO, G+H, 2CH) and reseed rate ($<$ 5 years, 5 to 10 years, and $>$ 10 years).

6.4 Results

6.4.1 Growing conditions

6.4.1.1 Annually

Mean precipitation (mm) and soil temperature (°C at 100 mm depth) per month for 2015 and 2016, is shown in Figure 6-1. The temperature threshold for initial pasture growth, ~6°C (Davidson, 1969; Peacock, 1975) is also shown in Figure 6-1. In 2015, total precipitation was 1,209 mm, and mean soil temperature was 11 °C. In 2016, total precipitation was 987 mm and mean soil temperature was also 11°C. There was 55 days where soil temperature was below the 6°C threshold for pasture growth in 2015, indicating there was at most 310 growing days in total. There was 77 days below 6°C in 2016, indicating there was at most 288 growing days.

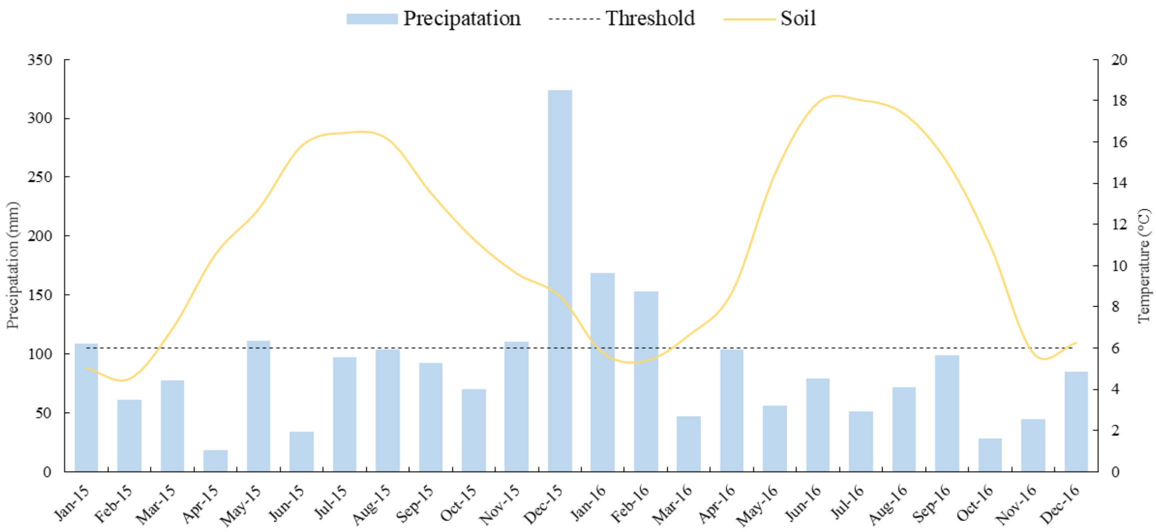

Figure 6-1. Growing conditions (precipitation and soil temperature) over the study period of 2015 and 2016 at Teagasc Moorepark Research Farm.

A mean pasture growth rate (kg DM ha^{-1}) per month for 2015 and 2016 at Teagasc Moorepark Research Farm Figure 6-2. A 5-year average growth rate as a baseline reference (PBI, PastureBase Ireland Database) is shown also. Mean pasture growth rate for 2015 and 2016 was 41 kg DM ha^{-1}, which was slightly higher than the 5-year mean (39 kg DM ha^{-1}). Overall, precipitation, soil temperature and growth rate were not significantly different between 2015 and 2016 ($p > 0.05$).

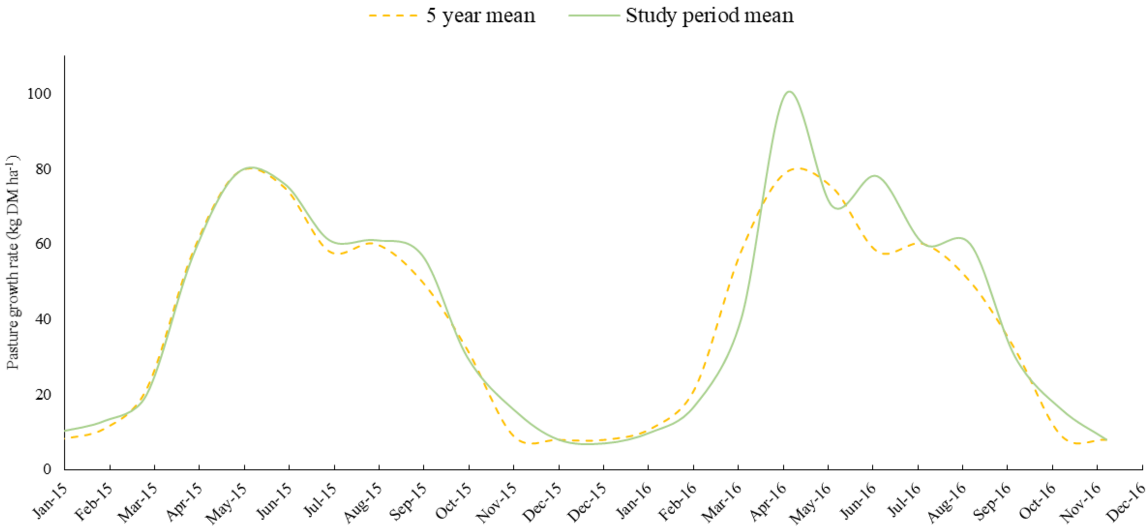

Figure 6-2. The 5 year mean daily growth rate and the daily growth rate over the study period of 2015 and 2016 at Teagasc Moorepark Research Farm.

6.4.1.2 Seasonal

Growing conditions per season are shown in Table 6-4. Spring 2015 had lower sum total precipitation (157 mm) and higher growth rates (30 kg DM ha^{-1}) compared to spring 2016 (305 mm and 22 kg DM ha^{-1}, respectively). However, summer and autumn 2015 had higher total precipitation (242 and 266 mm) and lower growth rates (72 and 49 kg DM ha^{-1}, respectively) compared to 2016 (186 and 199 mm, and 83 and 50 kg DM ha^{-1}, respectively). However mean precipitation, soil temperature and growth rate were not significantly different between seasons (p < 0.05).

Table 6-4. Growing conditions (precipitation and soil temperature) and growth rate per season (at Teagasc Moorepark Research Farm)

	Precipitation (mm)		**Soil (°C)**		**Study period mean (kg DM ha^{-1})**		**5-year mean (kg DM ha^{-1})**	
	2015	**2016**	**2015**	**2016**	**2015**	**2016**	**2015**	**2016**
Spring	157	305	7	7	30	22	30	30
Summer	242	186	15	17	72	83	71	71
Autumn	266	199	14	14	49	50	47	47
Mean	222	144	12	13	50	52	49	49
SD	57	139	4	5	21	30	20	20

6.4.2 Herbage characteristics

6.4.2.1 Cover, dry matter and distribution

Mean 2015 and 2016 herbage cover (kg DM ha^{-1}) and DM% are shown in Table 6-5. Spring had the highest mean cover and DM (871 kg DM ha^{-1} and 23%). Summer and autumn had the same mean DM (both 19%), but a higher mean cover was found for summer (848 and 684 kg DM ha^{-1}, respectively). There were significant differences between seasons in herbage nutrient concentration and DM (p < 0.05). Variation in cover was greatest in spring followed by summer and autumn (CV 65%, 58% and 63%, respectively) and variation in

DM% was also largest in spring (CV 30%) but summer and autumn were the same (CV both 21%).

The overall annual mean cover and DM% was 801 kg DM ha^{-1} and 20% DM after the two years.

Table 6-5. Mean, standard deviation and coefficient of variation per season for herbage cover and DM% for 2015 and 2016

	Herbage cover (kg DM ha^{-1})			Herbage DM (%)				
	Spring	Summer	Autumn	Spring	Summer	Autumn	Annual	
Mean	871	848	684	23	19	19	801	20
SD	569	498	436	7	4	4	501	5
CV	65	58	63	30	21	21	62	25

The distribution of herbage cover relative to grazing status is shown in Table 6-6. Overall, 84% of samples were taken when the paddock was in a recovering/immature grazing status (201 to 1,300 kg DM ha^{-1}). Only 6% of samples were taken when the paddock was in optimum grazing status (1,301 to 1,600 kg DM ha^{-1}) and 7% of samples were taken when or ready for silage harvest (> 1,600 kg DM ha^{-1}). There were only 3% of samples taken when the paddock was in a grazed-out stage (< 200 kg DM ha^{-1}). This pattern of distribution was relatively similar between each season (Table 6-6).

Table 6-6. Distribution of grazing status of herbage per season at time of sampling

Grazing status	Herbage cover (kg DM ha^{-1})	Sward height (mm)	Distribution (%)			
			Spring	Summer	Autumn	Annual
Grazed out	< 200	40-50	2%	2%	6%	3%
Recovering/immature	201 to 1,300	51-80	82%	84%	85%	84%
Optimum	1,301 to 1,600	81-120	7%	7%	5%	6%
Silage	> 1,600	120+	10%	7%	4%	7%

6.4.3 Sward species composition

The sward species identified in the herbage survey are listed in Table 6-7. The results are illustrated in Figure 6-3. The total number of species per sward ranged from 1 to 7. PRG was recorded as dominant (> 75%), the most frequently (351 records). There were much fewer records of abundant (51-75%), frequent (26-50%), and occasional (11-25%) for PRG (83, 21 and 6 records). WC was recorded as rare (1-10%) most frequently (147 records), followed by fewer records for occasional and frequent (87 and 29 records). BLD was also recorded most frequently as rare (163 records), followed then by occasional and frequent (9 and 3 records, respectively). All 10 other the species had fewer than 100 records, and were predominantly recorded as rare, making up only 1-10% of the paddock species composition at any time. Therefore, a typical species composition for a paddock was > 75% PRG, with 10-25% WC, and 10-25% of other species and they were most likely BLD, CB or AMG.

Table 6-7. Herbage species (code, Latin name and common name) identified during field scale herbage survey 2017

No.	Code	Latin name	Common name
1	PRG	*Lolium perenne*	Perennial rye grass
2	YF	*Holcus lanatus*	Yorkshire Fog
3	AMG	*Poa annua*	Annual Meadow Grass
4	WC	*Trifolium repens*	White Clover
5	CC	*Stellaria media*	Common Chickweed
6	BLD	*Rumex obtusifolius*	Broad leaf Dock
7	D	*Taraxacum officinale*	Dandelion
8	WT	*Carduus crispus*	Welted Thistle
9	N	*Urtica dioica*	Nettle
10	DFCB	*Geranium molle*	Doves foot Cranes Bill
11	CB	*Ranunculus repens*	Creeping Buttercup
12	HB	*Cardamine hirusta*	Hairy Bittercress
13	RDN	*Lamium perpereum*	Red Dead Nettle
14	SP	*Capsella bursa-pastoris*	Shepherds Purse
15	RP	*Plantago lanceolata*	Ribwort Plantain
16	RC	*Trifolium pratense*	Red clover
17	CME	*Cerastium fontanum*	Common Mouse ear

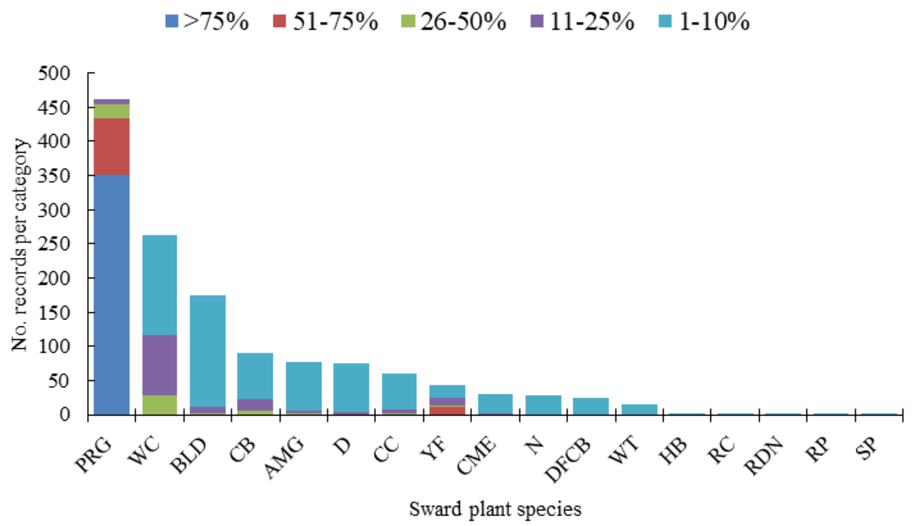

Figure 6-3. Occurrence and prominence of sward herbage species at paddock scale.

6.4.4 Herbage nutrient status

The proportion of herbage samples and their nutrient concentration status are shown in Table 6-8. Overall, 66% of herbage samples were within Sufficient nutrient concentration status. It was found that 27% of samples were in Low nutrient status 8% were in High nutrient concentration.

Table 6-8. Proportion of herbage samples with low, sufficient, or high nutrient concentration status

Nutrient status	Proportion of samples within optimal range (%)								
	N	P	K	Ca	Mg	Cu	Mn	Zn	Mean
Low	28	25	26	22	40	17	30	28	27
Sufficient	65	61	68	65	54	74	68	72	66
High	7	14	6	13	6	9	2	-	8

6.4.5 Temporal variation and trends in herbage nutrient concentration

Herbage nutrient concentration results per season and annual mean are shown in Table 6-9 and Figure 6-4, Figure 6-5, Figure 6-6, Figure 6-7, Figure 6-8. Nutrient concentration in plants was most variable in spring (CV 30%) and slightly less variable both summer and autumn (CV 26% and 27%, respectively) (Table 6-9). Mean herbage N, P, Ca and Mn were within range of their respective sufficient status level in each season, but mean herbage K, Mg Cu and Zn were slightly lower than the sufficient status, for each season.

Table 6-9. Mean, standard deviation and coefficient of variation of herbage nutrient concentration at field scale between 2015 and 2016

		N	P	K	Ca	Mg	Cu	Mn	Zn	
				(g kg^{-1})				(mg kg^{-1})		
	Sufficient status*	**30-40**	**3-4**	**30-60**	**3-6**	**3-6**	**10-20**	**30-100**	**30-60**	
Mean	Spring	34.0	3.4	23.9	4.9	1.6	9.2	91.2	27.0	
	Summer	31.3	3.6	27.0	5.2	1.9	7.9	84.6	27.6	
	Autumn	33.8	3.8	28.2	4.8	1.9	8.4	87.8	26.8	
	Annual Mean	33.0	3.6	26.4	5.0	1.8	8.8	85.0	28.2	
SD	Spring	5.9	0.7	5.4	1.3	0.3	2.8	59.9	10.8	
	Summer	6.0	0.5	5.6	1.2	0.4	1.9	54.7	5.7	
	Autumn	5.4	0.6	5.9	1.1	0.4	2.2	56.8	7.2	
	Annual Mean	5.8	0.6	5.7	1.2	0.4	2.8	62.5	8.2	CV mean
CV (%)	Spring	17	20	23	27	20	31	66	40	30
	Summer	19	15	21	23	18	24	65	21	26
	Autumn	16	15	21	23	20	27	65	27	27
	Annual Mean	18	17	22	24	19	27	65	29	38

Each herbage nutrient was significantly different between seasons ($p < 0.05$), except for Mn. Concentrations of the different nutrients in herbage showed quite different temporal patterns through the year. Herbage N concentration was significantly higher in spring and autumn (34.0 and 33.8 g kg $^{-1}$, respectively) compared to summer (31.3 g kg $^{-1}$). Herbage P was lowest in spring (3.4 g kg^{-1}) compared to summer and autumn (3.6 and 3.8 g kg $^{-1}$, respectively). Similarly, herbage K followed the same trend, lowest in spring (23.9 g kg^{-1}) compared to summer and autumn (27.0 and 28.2 g kg^{-1}, respectively). Herbage Ca was significantly higher in summer (5.2 g kg^{-1}) compared to spring and autumn (4.9 and 4.8 g kg^{-1}, respectively). Herbage Mg was significantly lower in spring (1.6 g kg^{-1}) compared to summer and autumn (1.9 g kg^{-1}). Herbage Cu was highest in spring (9.2 mg kg^{-1}) and decreased in summer and increased slightly in autumn (7.9 and 8.4 mg kg^{-1}). Herbage Mn followed the same pattern, highest in spring (91.2 mg kg^{-1}) lowest in summer and slightly higher in autumn (84.6 and 87.8 mg kg^{-1}, respectively), but these differences for Mn were not significant ($p > 0.05$). Finally, herbage Zn was significantly higher in summer (27.6 mg kg^{-1}) compared to spring and autumn (27.0 and 26.8 mg kg^{-1}, respectively). In summary, herbage P, K and Mg increased in concentration from spring to summer and into autumn also. Herbage N, Cu and Mn decreased from spring to summer, but increased in autumn. Herbage Ca and Zn increased from spring to summer but decreased again in autumn.

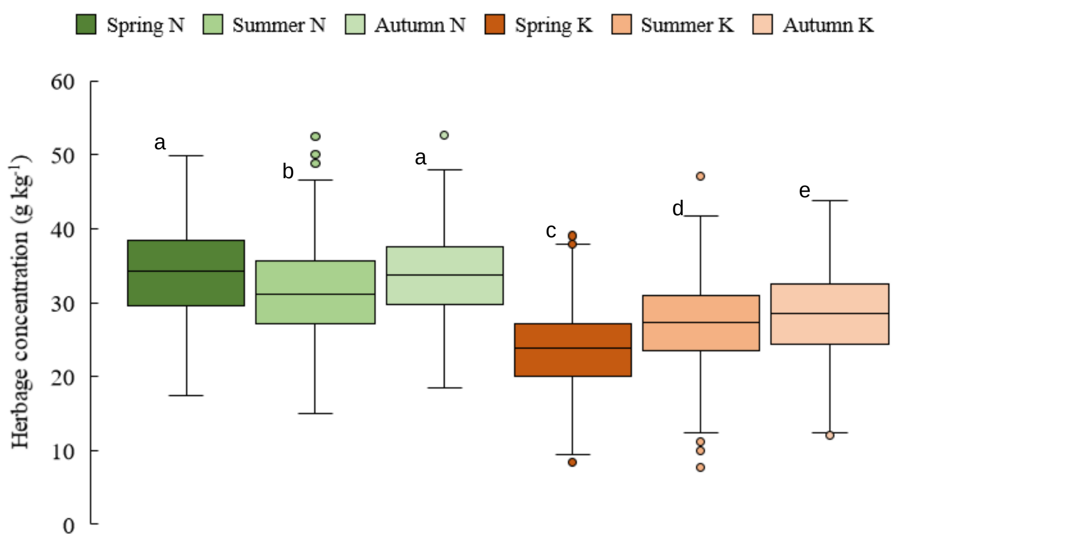

Figure 6-4. Mean herbage N and K nutrient concentration for spring, summer and autumn 2015 and 2016.

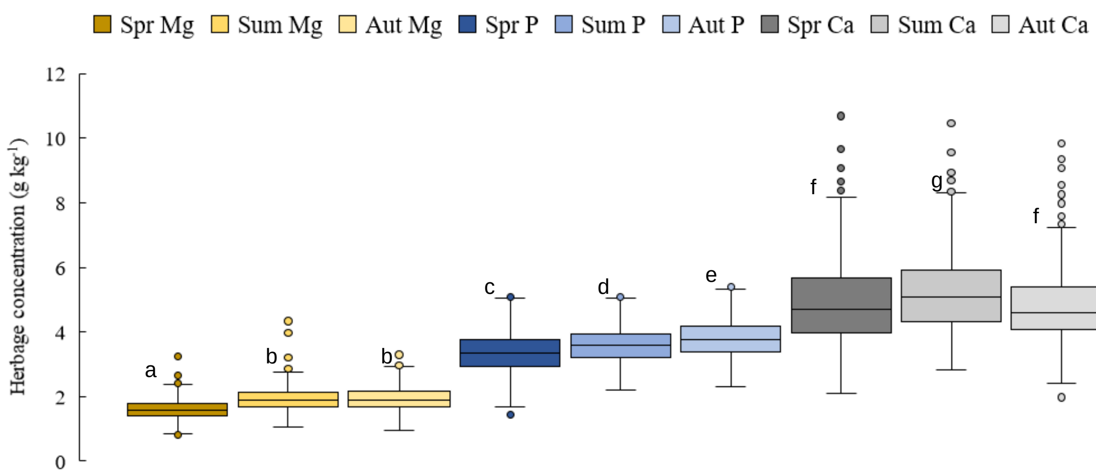

Figure 6-5. Mean herbage Mg, P and Ca nutrient concentration for spring, summer and autumn 2015 and 2016.

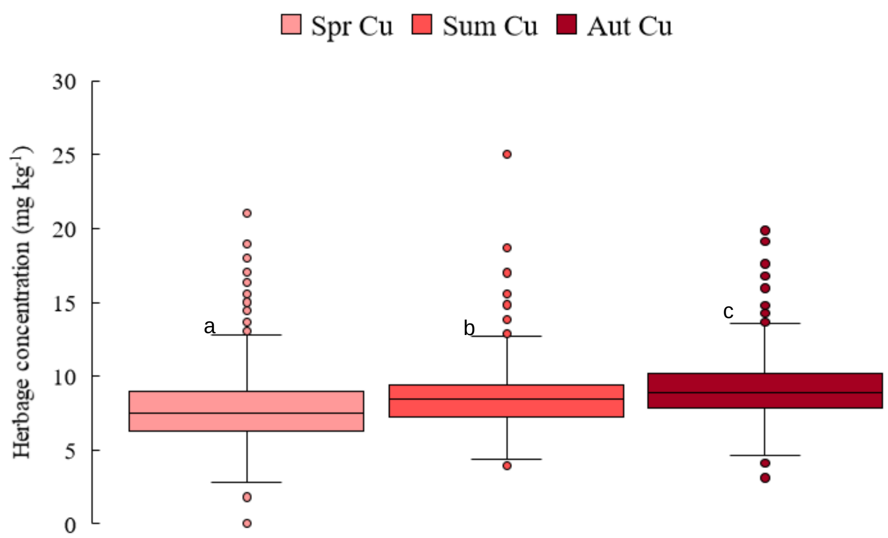

Figure 6-6. Mean herbage Cu nutrient concentration for spring, summer and autumn 2015 and 2016.

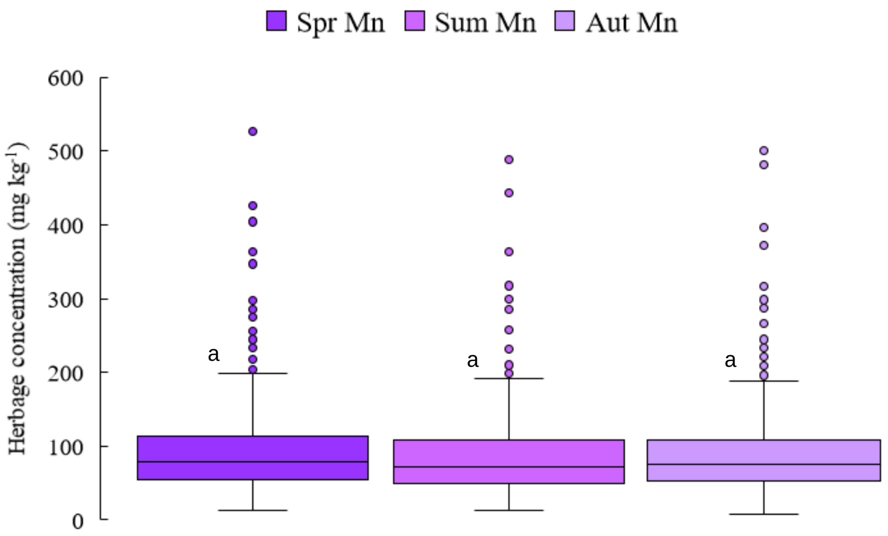

Figure 6-7. Mean herbage Mn nutrient concentration for spring, summer and autumn 2015 and 2016.

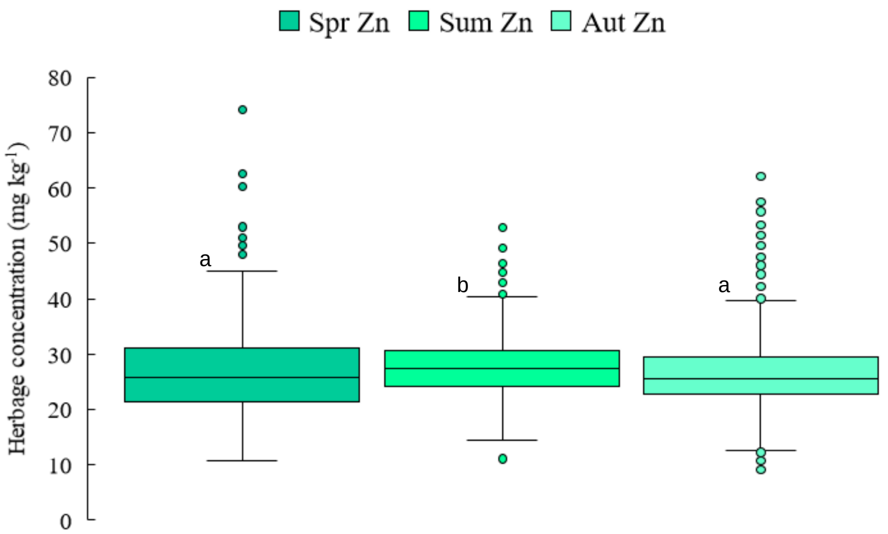

Figure 6-8. Mean herbage Zn nutrient concentration mean for spring, summer and autumn 2015 and 2016.

6.4.6 Soil fertility, environmental and management factors

There were 15 farms with sufficient data for the full analysis of soil, environmental and management factor.

6.4.6.1 Soil fertility

Soil pH was lowest in the first year of the herbage study but increased in the second year (Table 6-10). Both STP and STK had decreased between years but STMg had increased.

Table 6-10. Mean soil fertility levels (soil pH, soil test phosphorus and potassium) across participating farms in 2015 and 2016

	Soil fertility			
	pH	**STP**	**STK**	**STMg**
2015	6.0	9.0	145.7	180.4
2016	6.2	6.1	142.2	212.2

A number of significant relationships were found between herbage nutrient concentration (herbage N, P, K, Ca, Cu, Mg, Mn and Zn) and soil fertility (soil pH, STP, STK and STMg) ($p < 0.05$). The significant model for herbage K had the largest amount of variation explained by all soil fertility components (η^2 = 0.44), followed by herbage P (η^2 = 0.39), Mg (η^2 = 0.34), Ca (η^2 = 0.28) and Mn (η^2 = 0.23). Less variation was explained by soil fertility for herbage N (η^2 = 0.13), Zn (η^2 = 0.10), and Cu (η^2 = 0.02). Overall, only six herbage nutrients had variation greater than η^2 = 0.10 explained by soil fertility. It was found that STK was the main driver of herbage K and Ca (η^2 = 0.26 and 0.11, respectively). Furthermore, STMg was the main driver of herbage Mg (η^2 = 0.28) and STP was the main driver of herbage P (η^2 = 0.35). Finally, soil pH was the main driver of herbage Mn and Ca (both η^2 = 0.15).

6.4.6.2 Soil characteristics

Soil chemistry and drainage were significant factors for herbage nutrient concentration, but gradient class was not. Soil Al had a significant effect on herbage Mg (η^2 = 0.10), as soil Al increased, herbage Mg increased also. Soil Fe had a significant effect on herbage Ca (η^2 = 0.14), as soil Fe increased, herbage Ca increased also. Soil Cu had a significant effect on herbage Cu and Zn (η^2 = 0.15 and 0.17), as soil Cu increased, herbage Cu and Zn increased also.

Additionally, soil drainage class was significant for herbage Mn, Ca and N (η^2 = 0.18, 0.13 and 0.12, respectively). For this result, herbage Mn was significantly lower on well drained (WD) (78.1 mg kg^{-1}) compared to moderately drained (MD) and poorly drained (PD) (105.6 and 142.4 mg kg^{-1}), herbage Ca was significantly lower on WD (0.50 g kg^{-1}) compared to MD and PD (0.54 and 0.61 g kg^{-1}, respectively) and herbage N was significantly higher for WD (3.4 g kg^{-1}) compared to MD and PD (3.1 and 3.0 g kg^{-1}, respectively), but MD and PD were not significantly different.

6.4.6.3 Nutrient management

Nutrient balance and nutrient use efficiency significantly affected herbage P, K, Mg, Ca and Cu ($p < 0.05$) concentrations. Potassium balance (KBAL) was significantly related to both herbage P and K ($\eta^2 = 0.21$ and 0.12) concentrations (increasing effect), while nitrogen and phosphorus use efficiency (NUE and PUE, respectively) was significantly related to herbage Ca (both $\eta^2 = 0.13$) concentration (high NUE decreased herbage Ca but high PUE increased Ca) and phosphorus balance (PBAL) was significantly related to herbage Mg ($\eta^2 = 0.15$) concentration (high PBAL increased herbage Mg). These relationships are likely driven by the soil fertility status also. It was also found that total K application significantly affected herbage P and K ($\eta^2 = 0.16$ and 0.12) concentration (K application increased herbage values). Total N application significantly affected herbage Cu and Mg ($\eta^2 = 0.14$ and 0.12) concentration (high N had increasing effect for herbage Cu, decreasing effect for Mg). Total P application significantly affected herbage Mg concentration also ($\eta^2 = 0.15$) (high P had a slight decreasing effect on herbage Mg).

6.4.6.4 Sward management

In terms of sward management, it was found that the total number of grazing rotations had a significant effect on herbage P ($\eta^2 = 0.15$) concentration, while the total number of harvests had a significant effect of herbage N and K ($\eta^2 = 0.15$ and 0.22) concentrations. Additionally, it was found that only herbage N and K were significantly impacted by sward management factors ($p < 0.05$), however, their contribution to explaining variation in herbage nutrient concentrations was small overall.

For herbage N, there were significant interactions found between No. of species, cropping and reseed ($\eta^2 = 0.06$) and also between PRG and reseed ($\eta^2 = 0.05$). In this first interaction, the highest herbage N was found in paddocks with three to four species per sward species, used for 2CH and were reseeded over ten years ago (3.72 mg kg^{-1}, Standard Error (SE) 0.23). While herbage N was lowest in paddocks with more than four species per sward, used for GO, and were reseeded less than five years ago (2.93 mg kg^{-1}, SE 0.20). For the second interaction, the highest herbage N was in paddocks with medium levels of PRG composition and were reseeded over ten years ago (3.42 mg kg^{-1}, SE 0.05), and the lowest herbage N was in paddocks with low levels of PRG composition and were reseeded in fields less than five years ago (3.09 mg kg^{-1}, SE 0.09).

For herbage K, a significant interaction was found between PRG, WC and reseed, as well as between No. of species and reseed ($p < 0.05$). Herbage K was highest in paddocks with low levels of PRG, where WC was present and were reseeded less than five years ago (3.00 mg kg^{-1}, SE 0.20). Whereas lowest herbage K was on paddocks with high levels of PRG, where WC was absent and were reseeded less than five years ago (2.43 mg kg^{-1}, SE 0.19). For the second interaction, the highest herbage K was in paddocks that had three to four species per sward and were reseeded over ten years ago (2.85 mg kg^{-1}, SE 0.09). The lowest herbage K was in paddocks with three to four species per sward but were reseeded between five and ten years ago (2.42 mg kg^{-1}, SE 0.08).

6.5 Discussion

It was found that many environmental and management factors influence herbage nutrient concentration annually and seasonally, at the paddock scale on commercial dairy farms. It was also found that there are significant relationships between herbage nutrient status and soil fertility at this scale.

6.5.1 Seasonality in herbage characteristics

The significant variation in sward cover and DM% between seasons is likely to be as a result of a combination of the plant progression from vegetative to reproductive development stage, and the sward management strategy of the previous autumn, as well as the differences in seasonal growing conditions (soil temperature and rainfall) (Table 6-4, Figure 6-1). Plant development is associated with the differences in the way plants respond to their environment stimulus, which ultimately have important consequences on dry matter production (Jones, 2013). Furthermore, the sward management strategy impacts dry matter production also, for example, each week in delaying the closing of paddock grazing in autumn can have a 77 kg DM ha^{-1} reduction in spring grass accumulation (O'Leary et al., 2013). It is likely that the pasture covers were influenced by stocking rate changes which increase as the spring continues i.e. having sufficient cow numbers calved early enough to utilise spring grass (O'Leary et al., 2013). Overall, the timing of spring grazing (early or late) is expected to increase the herbage nutrient status (yield and nutrient concentration) and even in subsequent grazing rotations leading into the summer and autumn (O'Donovan et al., 2002) and ultimately improve animal performance through higher milk solids and weight gains (McEvoy et al., 2008). Grazing events are expected to stimulate the plant into the tillering process (Kays and Harper, 1974) and the nutrients from the excreta of grazing animals is expected contribute to the herbage concentration levels also (Whitehead, 2000).

It was found that the majority of paddocks were sampled when in a recovery/immature grazing status (84%). On these commercial farms where the paddock as the experimental unit, the seasonal sampling regime was executed during an active grazing rotation, so the samples over the entire farm capture the full range of grass growth corresponding to the sward cover levels across the different paddocks. The herbage cover profile in a typical rotation, is expected to range from high (pre-grazing) to very-low (post grazing) sward covers, depending on the time since the paddock was last grazed. As well as this, there may be paddocks taken out of the rotation for silage production e.g. G+1H fields. As a result, at each time of sampling for this study, a profile of the range in herbage cover and nutrient concentration from paddocks was captured. As only one paddock would be actively grazed at any time by the dairy herd, a snapshot of the grass growth stage within the rotational grazing system was captured during each sampling event (corresponding to season) and the corresponding nutrient profile was expected to represent a range of different post grazing sward covers and nutrient status present.

6.5.2 Sward species composition

The low number of different species per sward (1 to 7) and the fact that swards were made up of more than 75% perennial ryegrass is typical of commercial dairy farms. As a result, species diversity is generally low in this study but, it is a typical sward species composition in intensive dairy systems in Ireland (O'Donovan and Delaby, 2005; Cashman et al., 2016; Tuñon et al., 2014). Monocultures and the focus on a single characteristic such as yield, is the traditional approach to the management of sward species composition on many intensive dairy farms. The grazing systems in many European countries have developed towards this single species monoculture of high yielding, high value grass species, that are managed at high levels of fertiliser nitrogen and stocking rate or with frequent harvesting (Wilkins et al., 2002; Connolly et al., 2009). Research has highlighted the benefits of including multi-species sward on grazing systems for ruminant production (Egan et al., 2017; Murphy et al., 2018), however the adoption of such strategies are still limited at commercial farm level.

6.5.3 Seasonality in herbage nutrient status

The greater variation in herbage nutrient concentration in spring compared to summer and autumn, was likely driven by the same reasons that influence cover and DM% - vegetative state of the growing plant and sward management strategy and growing conditions. Additionally, as light intensity and air temperature increases with solar radiation, growth is stimulated (McCall and Bishop-Hurley, 2003), herbage nutrients are expected to change as the plant matures. In temperate zones, the strongest relationship between these factors in determining the growth of pastures occur in spring (Skinner et al., 2008). The nutritional value of pastures is influenced by the species composition, and plant growth stage, at the time of grazing. The changes in herbage nutrient status can be influenced by several factors, such as increasing maturity, soil nutrient content, frequency and intensity of grazing and utilisation system (Martínez-Ballesta et al., 2010; Wang and Schjørring, 2012) which has been shown in this study also. Furthermore, environmental factors exert the largest influence on forage quality by altering leaf/stem ratios and also causes modifications in plant development and changes in chemical composition of plant parts (Buxton, 1996). Therefore, variation in herbage nutrients is likely to be greatest in spring, same as cover and DM%, as was found in this study.

A number of studies found seasonal variation in herbage concentrations due to either plant development/maturity or seasonal environmental differences (Demanet et al., 2015; Martz et al.; Murphy et al., 2013). As a result, there tends to be variations in the trends of some herbage nutrients.

In one study, herbage P concentration showed a seasonal pattern of variation, with maximum levels in late autumn, and winter, and minimum levels in summer (Metson and Saunders, 1978a). This pattern is in agreement with the findings of this study which found that herbage P increased from summer to autumn.

It has also been shown that herbage K showed high levels had occurred in late winter, with a peak in early spring, and the lowest levels in early summer (Metson and Saunders, 1978a), similar to the results of this study. Monthly fluctuations were attributed to the effect of livestock urine, but they found that trends in K were at lower levels in early summer, and peaks in early autumn, and again in winter (Metson and Saunders, 1978a).

Stewart and Holmes, (1953) studying the effects of N application on herbage, noted that Mg levels tended to increase as the season advanced. Other studies found that trends in Mg were seasonally similar to those for Ca, but were much less pronounced (Metson and Saunders, 1978a). For example, McNaught et al., (1968) carried out Mg fertiliser trials on clay loam plots and found substantial seasonal variations in Mg and Ca levels. Maximum values were found in summer and minimum in late winter to early spring. This is in agreement with the herbage Ca results (Figure 6-5) of this study but the herbage Mg levels actually further increased in autumn from summer (Figure 6-5). Reith, (1954) reported an increase in the Mg concentration of herbage (on control "no-Mg" plots) from 0.18% in late spring to c. 0.27% in autumn. Todd, (1961), likewise found increases in grass Mg from early spring to autumn, and also in clover Mg, from late spring to summer, but the increase was smaller for clovers. Jones, (2013) also found Mg in mixed herbage, grasses, and clovers to be lowest in spring and early summer, increasing to maximum values by autumn, similar to the trends in this study (Figure 6-5). For herbage Ca, Metson and Saunders, (1978a) found that the highest values occurred in summer and the lowest in mid-winter, while Reay and Marsh, (1976) found a sharp summer maxima for Ca for both grass and clover. McNaught et al. (1968), found marked seasonal variations in Ca levels, with maximum values in summer and minimum in late winter to early spring.

For herbage Zn and Mn, Reay and Marsh (1976) found that no significant seasonal differences were found for Zn or Mn in ryegrass plots. However, Metson et al., (1979) found that "unusually" high summer Zn levels, in another plot study, but also that there was still small seasonal effects in grass and clover. Furthermore, Zn levels in that study were observed as part of a clover study, were not consistently higher than in grasses, which differs from typical trends in Zn (Fleming, (1963); Reay and Marsh, (1976)).

Murphy et al., (2013); Enriquez-Hidalgo et al. (2016) and McFeely (1978) have shown that a characteristic of grass-clover pastures, is that in general, high N levels occur between late autumn and early spring. Metson and Saunders (1978b) also found that N levels tend to peak during the late winter and early spring which is in agreement with the results of this study (Figure 6-4). It was also indicated in that study that urinary return of N and K under grazing, can enhance the normal seasonal increases in N and K.

Adams and Elphick (1956) determined Cu levels in several pasture species on varying soil types in New Zealand. It was found that there were significant differences among species and among soils types. Seasonal trends in plant Cu were irregular which was partly attributed to climatic factors, but mature stemmy ryegrass was significantly lower in Cu than young leafy material. It was concluded that Cu was high in immature pasture with a high clover component. The results for Cu in this study (Figure 6-6) indicate that herbage Cu was high in spring but decreased in summer, which was likely due to the stemmy ryegrass which would be more common at that time of the year, due to higher growth rates in late spring and early summer.

6.5.4 Herbage nutrient status as affected by soil fertility

The results for this study found that the seasonal differences for Mn were not significant, similarly, Adamson et al., (1994) found that seasonal effects for Mn in pasture were distinct but that the overall variations between sites made it difficult to identify any general pattern. It has also been shown that Mn levels in grasses tend to exceed the levels in clover (Reay and Marsh, (1976)), and so clover presence or absence may be masked by perennial ryegrasses for example. Furthermore, Cuesta et al., (1993) indicated that fluctuations in herbage Mn values may be attributed to soil pH and oxidation-reduction effects.

The results of this paddock scale analysis found that herbage K, P, Mg, Ca and Mn had a stronger relationship with soil fertility levels than herbage N, Zn and Cu. Furthermore, STK and soil pH were drivers of more than one herbage nutrient, as opposed to STP and STMg which only significantly influenced herbage P and Mg, respectively.

In many cases, import of K fertilisers have been shown to increase herbage K concentration of pasture (McFeely, (1978); Roche et al., (2009b)). McNaught and Dorofaeff (1968) indicated that the K concentration of herbage was lower on light soils than in herbage from heavy soils. It has also been shown that surplus imports of K can result in decreased levels of calcium, magnesium, and sodium in herbage (Hemingway, 1999; Metson and Saunders, 1978a), some of which can be linked to the cation exchange capacity of the soil according to these studies.

Phosphorus imports on various forage crops have resulted in increased phosphorus content of the herbage (Duru and Ducrocq, 1996; Brockman et al., 1970). Phosphorus imports on forage crops have been shown to increase protein content which is likely due, to some extent, to the changes in botanical composition of the plant (Celebi et al., 2011). The alteration of soil pH through lime imports has been shown to increase calcium content of herbage (Riggs et al., 1995) and cause changes in the composition of forage also (Haynes, 1983).

Overall, variation in the level of soil fertility and environmental factors, like soil chemistry and soil drainage class, and the ratio of nutrients in the soil are grown is expected to cause variation in the chemical composition of the crop in several ways. As described already, botanical composition of mixed herbage and the leaf/stem ratio may cause differences as well as excreta from the rotations of grazing animals. It can be expected that in some cases, one nutrient may be so plentiful that luxury consumption occurs, resulting in the forage containing an abnormally large proportion of that nutrient (Blake et al., 2000). Additionally, various species react independently to differences in soil fertility (Anderson et al., 1992) and so the results of the relationships of soil fertility with herbage nutrient concentration may vary with the species and proportions of species in the stand.

6.5.5 Sward management factors

It was found that the total number of grazing rotations was significant for herbage P and the total number of harvests was significant for herbage N and K. The number of grazing rotations, is expected to increase the tillering rate and increase the yield and, is one of the main strategies for improving herbage utilisation, as explored by (Tuñon et al., 2014). In that study it was shown that, with increased grazing severity (post grazing height and frequency), there was increased tiller density in PRG swards. This may increase the demand/requirement or the capacity of the plant to take up P, so that the additional tillers can be produced. It is likely that the higher number of rotations, brings with it, a longer period of exposure to P from animal excreta.

For herbage N and K, it was found that the number of harvests caused a significant decline in the herbage concentration of these nutrients. Firstly, harvest remove much more of the plant than grazing does, i.e. remove much more of the stem and all of the leaves, likely leaving the plant with a bigger effort required to recover. Typically, organic manure is then recycled to the plant via machinery. Herbage N and herbage K are required in larger quantities than herbage P, and so it can be expected that larger quantities are removed during harvest also. Results suggest that with increased number of harvests, the herbage N and K in the plant declines. This could be as a result of the plant not receiving sufficient nutrient returns before each subsequent harvest to replenish the herbage N and K levels. Or that the plant is not capable of making a recovery to sufficient state before a subsequent harvest.

The results for herbage N indicate that there was a higher level of herbage N in 'older' swards (> 10 years since reseeded), or that have fewer no. of species (three to four) or that have a higher PRG composition (medium PRG), and used for silage harvests, compared to fields reseeded recently (< 5 years), have higher no. of species or low levels of PRG (low

PRG) and used for Grazing Only. It is likely that the older swards are less efficient at utilising N (Shalloo et al., 2011), and therefore require high N applications to achieve a target yield. Moreover, there is a lower requirement for N in more diverse species swards (Adamvics and Gutmane, 2018), and that grazing fields has a lower demand for N than silage (Čop et al., 2009). The outcome indicates that potentially lower N demand can be achieved through more frequent reseeding (up to 5 years), which is in agreement with (Shalloo et al., 2011), through diversity in swards or lower PRG levels and through appropriate cropping rate practices.

Highest herbage K was in fields with low levels of PRG and where WC present while the lowest herbage K was in fields with high PRG levels and where WC was absent. However, reseed rate was an important factor here also, with higher herbage K found in paddocks with older swards (reseeded > 10 years ago), compared to those more recently reseeded (five to ten years ago). Additionally, it was found that only herbage N and K were significantly impacted by sward management factors ($p < 0.05$) and that, sward management contributed little to explaining variation in herbage nutrient concentration overall.

6.6 Conclusion

There are implications for animal breeding, soil fertility, and sward management strategies due to the seasonal variation in herbage nutrient status throughout the growing season for these farms. Firstly, low concentrations of herbage K, Mg, Cu, and Zn in spring and autumn, may have implications for animal dietary requirements but also herbage production potential, due to the limiting factors of low K for example, at these times in the year. Ideally herbage nutrient status should be sufficient at each season of the year, particularly in the early spring when animals need to meet their nutritional requirements for successful breeding. To ensure herbage nutrient status and quantity is sufficient in spring, it may require important management decisions to be made in autumn of the previous growing season. The low number of total herbage species per sward was expected on these intensive dairy systems, as it is common practice to have a perennial ryegrass dominated sward which typically generate high yields. However, there may be opportunity to widen the variety of species to include species with deeper rooting systems than perennial ryegrass to provide more reliability in the supply of nutrients each season. There may be implications however for herbage yield and potentially animal productivity e.g. milk composition where changes are made to the variety of sward species. Furthermore, the impact of sward reseeding frequency, number of harvests and number of defoliations on herbage nutrient status has implications for timely management strategies. The frequency of reseeding can be high on intensively stocked farms (e.g. 10% of the farm area every 1-2 years) but there are economic and environmental costs (carbon release) from reseeding events that must be considered also. The strong link between soil fertility levels and herbage K, P Mg, Ca, and Mn compared to herbage N, Zn, and Cu, highlights the importance of regular soil testing (e.g. 2 to 3 years) for maintaining appropriate nutrient supply. Moreover, it highlights the importance for soil testing for nutrients other than soil P and K which are commonplace, and that there may be a need to test, and utilise, information on soil Mg, Ca, and Mn levels to ensure appropriate levels are being supplied to the sward. This consideration of a wider soil test extends to potentially testing for soil Al, Fe and Cu levels too, which were found to affect the availability of herbage Mg, Ca, Cu and Zn. Building on this further, consideration of the soil drainage class may be important for maintaining herbage nutrient status and potentially indicate the need to nuance nutrient strategies. For example, well drained fields had the lowest herbage Ca and Mn levels, compared to moderate or poorly drained fields, whereas herbage N levels were actually highest on well drained fields compared to less well drained fields. It must also be noted that nutrient balances/surpluses indirectly affected different herbage nutrients i.e. potassium balance and herbage P, and phosphorous balance and herbage Mg. This is likely attributed to soil fertility where the plant production is limiting by the soil nutrient that is suboptimal for growth. This may have implications for nutrient management strategies and indicates that specific applications of specific nutrients in spring or autumn for example, to ensure adequate supply to the sward at these times of the year.

Overall, the paddock scale analysis provided some insight into the seasonal variability and the factors that drive it in a commercial pasture base dairy farm scenario. Ultimately, this study should provide information that should be used to support the development of future pasture and soil management strategies and inform advice relating to best practices carried out by knowledge transfer services. Additionally, the information could contribute towards validating results from studies at different scales such as plot, farmlet or modelling/simulation.

CHAPTER SEVEN: GENERAL DISCUSSION AND CONCLUSIONS

7.1 Overview

The aim of this research thesis was to evaluate the role of scale in nutrient management of grazed grass dairy farms. This could enable strategies and best practices that would improve production and environmental sustainability of Irish dairy farms. This research was carried out on 21 commercial dairy farms, in the south and south east of Ireland.

In total, 15 of the farms provided sufficient data to carry out a full suite of analysis for Chapters Three, Four and Five. Only data from all 21 farms were used for the analysis of herbage characteristics in Chapter Six. Commercial dairy farms were selected as the experimental unit, so as to allow the investigation of different scales, from sub-field to farm. This was to allow the characterisation of variability across scales, within real farming systems. It was theorised that the use of commercial farms as experimental units would enable the results to be informative, and potentially more applicable than, for example plot studies. Evaluating commercial farms may be more reflective of, or have a high capacity to represent the real relationship between indicators and variables. The outcomes of this study suggest that use of commercial farms was informative of those relationships, and it is expected that their applicability will be high also, however there were some limitations to their use too. The selected farms were not representative of the national Irish dairy industry as a whole, but did represent intensive grazed grass dairy systems for the south and south east regions of Ireland i.e. high stocking rate (mean 2.7 LU ha^{-1}), milk yield (mean 5,561 l cow^{-1}) and concentrate feed per cow (mean 1,038 kg cow^{-1}) (Teagasc, 2017; NFS, 2017).

The literature review highlighted that there is a need to further reduce the risk of nutrient losses from the agri-environment, to receptors such as air, water and atmosphere (GHG's). The nutrient transfer continuum conceptual framework (Haygarth et al., 2005a; Lemunyon and Gilbert, 1993; Smith et al., 2013) was used to explain how there is a need for an improved nutrient budgeting tool at the source level. This would ideally address the spatial distribution of within farm nutrients as an approach to reducing the risk of nutrient losses. The literature review identified that there was a knowledge gap in the linkages between actual nutrient losses at field scale, and losses estimated at various other scales, including plot, farm, research farm, modelling studies. A simultaneous assessment of farm and field scale was proposed. The aim of this was to help bridge the gap in knowledge between farm and field scale nutrient losses, so that an appropriate nutrient budgeting strategy or tool could be developed. A synthesis of key results and outcomes from each chapter is described below.

7.1.1 Characterisation of subzones and the use of commercial farms as experimental units

The methodology was the same for Chapter Three and Four, to investigate N, P, and K cycling and management across multiple scales on the selected dairy farms. Firstly, the paddock system, which was an established management unit already on these dairy farms, was fully characterised in a GIS. This process involved delineating the total grassland area into subzones respective to animal type (dairy, non-dairy) and cropping type (Grazing Only, silage harvest etc.). The boundaries of each scale were digitally mapping. Secondly, nutrient management data (import and exports) at farm, field and paddock scale were gathered and monitored for three years (2015, 2016 and 2017). Nutrient balance and use efficiency estimations were used as indicators of nutrient management sustainability at these different scales. The use of subzones was effective in the differentiating of within-farm nutrient flows, which potentially improves the capacity to target parts of the farm with specific nutrient management practices, as explored in the following sections.

7.1.2 Evaluating the linkages between farm and field scale nutrient management

In Chapter Three, it was identified that farm scale indicators of nitrogen (N) management did not accurately reflect the variation (denoted by CV: Coefficient of Variation) in nitrogen flows, pressures and loss potential hotspots, which occurred at field and paddock scale i.e. inside the farm boundary. This has important implications for the distribution and perception of N source pressure and efficiency of N management on dairy systems. For example, while farm scale nitrogen balance (NBAL) (mean 165 kg ha^{-1}; CV 34%) appeared to be a good indicator of the average N source pressure across a farm area, it did not reflect the large variability in actual NBAL at the field scale (mean 139 kg ha^{-1}; CV 77%), or its spatial distribution. These results were in agreement with Di et al., (1998) and Van Leeuwen et al., (2019). These studies suggested that NBAL at the field and soil scale better reflect actual N cycling between the soil and plant, and the losses to the wider environment, compared to farm scale.

In Chapter Four, there was also a significantly larger mean and smaller range in phosphorous (P) balances at farm scale, i.e. between farms (mean 9 kg P ha^{-1}; CV 58%), compared to field scale, i.e. within farm (mean 6 kg P ha^{-1}; CV 294%). Although there wasn't a significant difference for potassium (K) balances and use efficiency between farm (mean 23 kg K ha^{-1}, CV 86%), and field scale (mean 39 kg K ha^{-1}, CV 179%), in Chapter Four, the difference in variation was still larger at field scale.

It was identified that the variation in nutrient flow was driven by nutrient management decisions related to nutrient imports strategies (inorganic and organic fertiliser applications), which is in agreement with Lanyon (1994) and Oenema et al., (2011). The main drivers of N, P and K nutrient flows for both scales, were imports via inorganic fertiliser and exports via milk sold. Inorganic fertiliser made up 59% or more, of total imports of P and K, and 77% or more, of N. N exports via milk sold made up 57% or more of the total, P exports via milk made up 70% or more, and K exports via milk made up 50% or more. There was also considerable variation in inorganic fertiliser imports. Values ranged from CV 28% to 35% for N, CV 55% to 93% for P, and CV 83% to 111% for K. The large variation in nutrient management at farm and field scale that was shown in these chapters was also highlighted by others who have conducted similar studies (Buckley et al., 2016a; Mihailescu et al., 2014, 2015; Van Leeuwen et al., 2019).

In each evaluation of nutrient N, P and K cycling, it was highlighted that when farm scale indicators were observed in isolation, they do not accurately reflect the variability in nutrient management indicators at field scale. This also affects the perception of nutrient source pressure from the perspective of policy makers and stakeholders. It highlights the importance of considering which scale these indicators of nutrient use are being observed from, so that policy can be correctly guided to address issues correctly.

It was also shown that the delineated management subzones or sub-units (dairy, non-dairy, grazing only, silage harvest) were informative for identifying the distribution in nutrient pressures within farm boundaries in a grassland-based system. As found in the literature review, some studies don’t typically distinguish between subzones (Gourley et al, 2012b; Aarons et al., 2015; Dungait et al., 2012; Withers et al., 2001). The advantages of doing so are expressed by Aarts et al., (2000). They showed that it was a way of further partitioning nitrogen pressures spatially within the farm boundary, and could be utilised to improve spatial targeting of efforts in nutrient management advice. For example, the Dairy and Non-Dairy subzones had similarly high N source pressure (144 and 150 kg N ha^{-1}, respectively) and low use efficiency (UE) (57% and 37%, respectively), and so targeting of these fields for precision nutrient advice would have greater benefit than silage fields, which had low source pressure and high efficiency (27 kg N ha^{-1} and 86%). For P, it was found that the silage subzone had the greatest nutrient source pressure (15 kg P ha^{-1}), followed by Non-Dairy and Dairy (13 and 5 kg P ha^{-1}, respectively). Although some of this could be attributed to soil fertility build up strategies, or to replace high offtakes via harvest events, the lowest UE was actually on the Non-Dairy fields (47%) compared to Dairy and Silage fields (104% and 74%, respectively). Non-Dairy fields, therefore, likely have the greatest risk to environmental losses for P. The distribution of K source pressure showed that Dairy fields had the highest (41 kg K ha^{-1}), followed by Silage and Non-Dairy fields (15 and 7 kg K ha^{-1}), but Dairy fields also had the lowest efficiency (89%) compared to Silage and Non-Dairy (102 and 104%).

Furthermore, it was found that much of the variation in nutrient indicators found at field scale was associated with factors that are controllable by farmer decision making, e.g. sward management decisions (sward species composition, cropping, reseeding), as opposed to non-controllable environmental factors (slope, soil drainage class, soil chemistry). The results highlighted the need to individually tailor N, P and K nutrient management strategies to field or paddock characteristics. For example, fields with a high proportion of perennial ryegrass in the sward (91 to 100% PRG), were shown to have the lowest N use efficiency (61%), the lowest K use efficiency (87%), but a considerably higher P use efficiency (99%). Interestingly, in terms of cropping strategy, it was found that Grazing + Harvest, has the highest nutrient efficiencies in each case for N, P and K (83%, 124% and 172%, respectively). The large nutrient capture of P and K would be expected to have implications for soil fertility and production goals if it continued in the long term. Fields where a Grazing Only strategy was carried out, had the lowest nutrient use efficiencies for N and K (44% and 58%, respectively), but were reasonably high for P (93%). Finally, fields that were reseeded between 5 and 10 years ago had the highest N and K use efficiency (59% and 104%, respectively), and reasonably high P use efficiency (93%) also, compared to older pastures (> 10 years).

In terms of environmental factors, nutrient use efficiency was highest in each case for N, P and K, on well drained soils (59%, 107% and 99%, respectively). Whereas poorly drained soils had the lowest N, P and K use efficiency (46%, 72% and 57%, respectively). While gradient class wasn't significant for N or K, it was found that P use efficiency was significantly higher on fields with a slope. The identification of the differences in nutrient indicators, relative to these paddock characteristics, is expected to closely reflect the actual relationships between nutrient cycles and these management and environmental factors.

These factors contributed significantly to the observed variation at field scale. These results also highlight opportunities to improve nutrient use efficiency, potentially through improved field scale decision making. This has implications for the knowledge transfer (KT) services, which can influence such field scale decision making. This is something which farm level nutrient management regulation has limited capacity to address and should be an opportunity for KT services. There are also many barriers to farmer practice adoption at field/paddock scale that KT will be challenged to address, such as cost, data collection quality and confidence, skill level or confidence in the benefits of technology (Greiner et al., 2009; Higgins et al., 2019; Kelly et al., 2014). Future KT and policy efforts to improve nutrient balance and use efficiency on such specialised grass-based dairy farms, need to be cognisant of the spatial variability of nutrient management within the farm boundary (field/paddock scale), and measures might be most effective if they can be tailored to this spatial scale.

Overall, it was shown that this variability in nutrient source pressure at field/paddock scale, is practically invisible to policy makers, advisors and farmers when farm performance is assessed at farm scale and in isolation. The current maximum soil sample area for soil nutrient analysis under the Nitrate Regulations in Ireland is 5 ha. Results from this study would suggest that this may be too large an area to effectively target nutrient management spatially, particularly on the dairy subzone (mean paddock area 1.4 ha) of these farms, where nutrient loadings, for both N and P, were highest. Results would suggest that the dairy subzone ("milking platform"), in particular, might be targeted for nutrient management improvements in order to reduce nutrient source pressures and improve efficiencies.

7.1.3 Precision paddock scale nutrient advice

Chapter Five focused on quantitative changes in soil fertility over the time. The capacity to change soil fertility properties – soil pH, soil test P (STP) and soil test K (STK), in the context of soil characteristics and sward management factors were evaluated. Participating farms were encouraged to better match the scale of nutrient advice to nutrient management, through annual colour coded soil test results, based on the full spatial distribution of their farms. It was found that, in general, paddock scale nutrient advice for P was adhered to, but K and lime were not strictly adhered to. Despite this, during the paddock scale advice period, soil test values and the distribution of soil indices improved towards more optimal values: soil pH increased from 6.0 to 6.3 or Index 3 (Medium/Adequate), while STP decreased from 9.1 to 6.7 mg l^{-1}, and STK decreased from 151.9 to 142.9 mg l^{-1}, respectively, from Index 4 (Sufficient/High). It is likely that the colour coded paddock scale maps helped influence the nutrient distribution strategy for individual farms. Additionally, nutrient management (balance and use efficiency) significantly reduced in the initial year of advice: P and K surplus balances decreased significantly between 2015 and 2016 (7 to 4 kg P ha^{-1} and 52 to 34 kg K ha^{-1}). Interestingly, each subzone had the same outcome in terms of changes in soil fertility i.e. soil pH increased and STP and STK decreased on average. Furthermore, the dairy subzone or 'milking platform' exhibited the highest STP levels at the start of the study, but also exhibited the largest decrease in STP levels compared to the other subzones, at the end of the study.

It was found that soil organic matter level had the strongest relationship to change in soil pH (η^2 = 0.16), whereas the soil Aluminium level had the strongest relationship to change in STP (η^2 = 0.42). Soil Aluminium level was also a significant driver of change in STK (η^2 = 0.17), as was soil Magnesium level (η^2 = 0.22). Interestingly, the interaction between white clover (*Trifolium repens*) and reseed rate, was one of the largest drivers of change for both STP and STK (η^2 = 0.37 and η^2 = 0.23, respectively). A theme emerged that STP and STK tended to have large negative changes where recent reseeding had occurred (< 5 years ago). In the evaluation of environmental and management factors overall, it was indicated that sward management factors were having a greater influence on the capacity to change soil fertility levels than soil characteristics such as chemistry or soil drainage class. As a result, it has been shown that there are potentially significant opportunities to improve P and K management and reduce nutrient source pressure on dairy farms at paddock scale, which is similar to the findings of Gourley et al., (2015). The improvements observed in this study do not, of course, represent the full range of potential improvements possible within such dairy systems; merely what was achieved by the farmers in this study over this time period. Results would also indicate that improvements in nutrient source pressure (farm/field balance, soil P status) can be achieved quite quickly. However, as Murphy et al., (2015), demonstrated for a grassland catchment in southwest Ireland dominated by similar specialised dairy farms, improvements at the source end of the nutrient transfer continuum can take significant time to lead to improvements in water quality; a significant time lag, particularly for P (Cai and Ofterdinger, 2016; Fenton et al., 2011).

The outcomes of this chapter also highlighted the importance of sampling strategy (number of samples per area and full spatial distribution) when aiming to improve or address soil fertility issues, the idea of which was in agreement with Penn et al., (2007). The need for sampling strategies that capture between paddock variability in soil nutrients is also noted by Anderson et al., (1992) and Mathews et al., (1994). Such a spatially targeted nutrient management approach in dairy systems could eventually be extended to within paddock variability as shown by Aarons et al., (2015). The high spatial variability among soil indices at paddock scale found in this study indicate that traditional methods of sampling (larger scale, e.g. > 4 ha), may not be of practical value for monitoring or planning accurate and optimised levels of nutrient application to soils. A greater understanding of farm nutrient fluxes and expected patterns of within-farm nutrient distribution, complemented by comprehensive soil testing, may help guide more profitable and environmentally beneficial nutrient management decisions. This study goes some way towards establishing this level of understanding for the grazed grass systems of milk production characteristic of southern Ireland.

7.1.4 Characterising paddock scale nutrient status on rotational grazing systems

The objective of Chapter Six was to evaluate the variation in seasonal herbage nutrient concentration and identify how soil, environmental and sward management factors affect herbage nutrient status. It was found that many of the seasonal patterns in herbage nutrients on commercial farms, were similar to that which were identified in plot and research farm studies (O'Donovan and Egan, 2019; Machado et al., 2005; Demanet et al., 2015b). For example, it was found that spring had the most variation in herbage characteristics (cover, DM% and herbage nutrient concentration), which was in agreement with Kennedy and O'Donovan, (2014); Metson et al., (1979); Murphy et al., (2013). The significant variation in cover and DM% in spring was driven by a combination of the plant progression from vegetative to reproductive development stage, growing conditions (soil temperature, light intensity and rainfall), and the sward management strategy of the previous autumn. As shown by O'Leary et al., (2013), the sward management strategy impacts dry matter production, each week in delaying the closing of paddock grazing in autumn can have a 77 kg DM ha^{-1} reduction in spring grass accumulation. These management strategies, such as the timing of spring grazing (early or late), and number of rotations, are expected to increase the nutrient status (yield and nutrient concentration) leading into the summer and autumn (O'Donovan et al., 2002). This ultimately has importance implications for animal performance i.e. milk solids and weight gains (McEvoy et al., 2008). The greater variation in herbage nutrient concentration in spring compared to summer and autumn, was likely driven by the same factors that influenced herbage cover and DM% - vegetative state of the growing plant and sward management strategy and growing conditions. Skinner et al., (2008), highlighted that in temperate zones, the strongest relationship between these factors in determining the growth of pastures typically occur in spring.

The results of this paddock scale analysis found that herbage K, P, Magnesium (Mg) Calcium (Ca) and Manganese (Mn) had a stronger relationship with soil fertility levels compared to herbage N, Zinc (Zn) and Copper (Cu). Furthermore, soil test K and soil pH were drivers of more than one herbage nutrient, as opposed to soil test P and soil test Magnesium (STMg), which only significantly influenced herbage P and Mg respectively. It was found that STK was the main driver of herbage K and Ca (η^2 = 0.26 and 0.11, respectively). Soil test Mg was the main driver of herbage Mg (η^2 = 0.28), and STP was the main driver of herbage P (η^2 = 0.35). Finally, soil pH was the main driver of herbage Mn and Ca (both η^2 = 0.15). This has implications for soil fertility management and maintenance of herbage nutrient status throughout the seasons each year. Soil fertility management typically focuses on P and K levels, however, as shown, the maintenance of soil pH and soil K is important for multiple herbage nutrients as well as animal performance and production, which is in agreement with Reijneveld et al., (2014).

Variability in herbage nutrient concentration and soil fertility are expected to be found at smaller scales than the paddock, as shown by Aarons et al., (2015). However, the paddock scale may be sufficiently small enough to start addressing within farm nutrient flows, so that variability in herbage and soil nutrients can be minimised. The intensity of grazed grass dairy system puts a high demand on nutrient flows in the soil–plant–animal cycle. And so management strategies require substantial fine tuning to ensure efficiency is adequate. The identification of the key drivers of nutrient management, soil fertility and nutrient status at paddock scale, may help to guide new management strategies in the future.

7.1.5 Limitations of the research

The data collection method could have been designed to be more efficient than what was carried out. Seasonal and annual farm visits as well as follow up phone calls to the participants were used to collect the primary data. This was a rigorous but time-consuming process due to the range in formats and styles of records that had to be standardised each month and year. The limitations can be inherent in the data collection process and therefore can create uncertainties. These factors of uncertainty are inherent in farm and field size changes due to purchased or renting land, annual animal numbers and sales, grazing date accuracy. This emphasises the difficulty of individual farmers collecting the information and shows how the approach overall is informative to the research and policy makers primarily. An alternative approach would be to select farmers that are equally skilled technologically so that records could be submitted electronically or a log book for each participant with a standardised template to strictly follow. A more stringent selection process may have also increased the sample size for the study. For example, there were 21 farms selected but only 15 provided sufficient data for full analysis, with only partial or none of the required data submitted by 6 farms. Furthermore, the research aims and objectives were broad and the reliance on a large number of variables to be collected correctly for the full analysis to be carried out meant that where a single piece of data was missing, the full analysis could not be completed. This also limited the sample size of fields and farms that could be fully evaluated as a result. The full analysis for Chapters Three, Four and Five required a large number of details to be submitted correctly, e.g. nutrient management records (N, P, K and lime imports) and sward management details (number of defoliations/rotations and number of harvests, total yield) which was based on the farmers individual capacity to regularly carry out grass measurements.

As described by Schröder et al., (2003) and Öborn et al., (2003), there is a need for systems that streamline data collection and ensure adequate screening of data, so as to reduce uncertainties. There are international indicator database systems available through the Organisation for Economic Co-operation and Development (OECD) (Eurostat, 2013; OECD, 1999b; OECD, 2015), which could be developed further to include data from additional scales. More recently, nationally representative farm scale nutrient balance benchmarks have been established (Thomas et al., 2020) but generally there is a lack of standardised paddock scale databases both nationally and internationally from which further monitoring and analysis can be carried out.

7.1.6 Future research

One aspect that was not considered as part of this research was the economic costings of, for example, the soil sampling regime, and the economic benefits of improved soil fertility or increased nutrient efficiency. Mihailescu et al., (2015); Mihailescu et al., (2014); Buckley et al., (2016b) and Adenuga et al., (2018b) have carried out an economic costings and efficiency benchmarks of dairy farms in Ireland, and it would be expected that insight could be gained from carrying out economic analysis of the value of paddock scale nutrient management planning.

Another aspect which may provide further insight would be to estimate the potential nutrient losses from nutrient surpluses of individual paddocks using coefficient values from inventory databases (Aarts et al., 2000) or nutrient loss models (Khalil et al., 2019; Bragina et al., 2019).

The paddock scale data from this project may also be used for validating existing Irish farm models or to calibrate international farm models to an Irish scenario. There are already some dairy farm models and components of models (paddock scale grass growth prediction) available e.g.; Shalloo et al., (2004b); Ruelle et al., (2019); Fitzgerald et al., (2005). It is likely that national grass measuring technologies like the PastureBase Ireland database (Hanrahan et al., 2017) and national nutrient planning software like NMP online (Wall and Plunkett, 2016; Murphy et al., 2015), which are software programmes capable of recording data at paddock scale, but operate independently, could be synchronised. The combination of these two software packages could be developed together so that data on paddock scale nutrient balance variation could be estimated and monitored long term. It is not unusual to merge components of models or multiple models together to function as a single farm system simulator. This has been done with 'MINas' in the Netherlands (Van den Brandt and Smit, 1998; Hanegraaf and den Boer, 2003) and with 'Lypsikki' in Finland (Huhtanen et al., 2011; Nousiainen et al., 2011) and with models such as 'DairyMod', 'EcoMod', and 'OVERSEER(TM)' for Australian and New Zealand grass-based dairy systems (Johnson et al., 2008; McCall and Bishop-Hurley, 2003; Bryant and Snow, 2008; Wheeler et al., 2003). The development of such modelling capacity for Irish dairy systems, and other farm systems, could be considered to support the development of more precise paddock/field scale nutrient management and the development and evaluation of policy options to reduce nutrient losses from farms to the wider environment.

7.1.7 Contribution to knowledge

It was identified that there was a knowledge gap in the linkages between farm and field scale nutrient management on specialised grazed grass dairy farms. This research contributed to bridging that knowledge gap, by appraising that the role of scale, is important for revealing that 1) there is significant variation and poor distribution in nutrient cycling between these scales, 2) precision nutrient advice at paddock scale can lead to improvements in soil fertility levels and nutrient distributions, and 3) nutrient status can be maintained through appropriate and more spatially precise soil fertility management and soil characteristic consideration.

7.1.8 Concluding remarks

The national dairy production targets of the Food Harvest 2020 and Food Wise 2025 plans (Food Harvest 2020, 2010; FoodWise 2025, 2015) have been met, well ahead of the expected target, as shown by the national level post milk quota analysis (NFS, 2017). Additionally, production targets around Europe were achieved to some extent by the increased number of dairy farms since the milk quota was abolished in 2015 (CSO, 2016; EU, 2018). However, the largest proportion of production, in Ireland in particular, came from intensification of existing dairy systems, by increasing stocking rate or increasing overall farm size, as reported by NFS, (2017). It is important now that these specialised dairy systems that have intensified production output, also intensify nutrient management strategies, to minimise the risk of nutrient losses to various environmental receptors. Overall, this research has shown that there are significant opportunities to improve nutrient management sustainability on this grass-based dairy system. This can be achievable through better spatial matching of nutrient management to the factors that control nutrient use efficiency and potential nutrient losses, at the field/paddock scale. On the most intensive (stocking rate, nutrient loading) areas of these farms (the ‘milking platform’) the paddock scale offers a management unit at a reasonably small spatial scale (mean 1.4 ha) that might effectively capture much of the spatial variability in environmental factors that control nutrient cycling and losses. As these paddocks are already an integral part of farm management in these systems (grass and grazing management, for example), adoption of improved and more precise nutrient management practices might, therefore, be considered more likely, at this scale.

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Appendix 1A – Fieldwork equipment

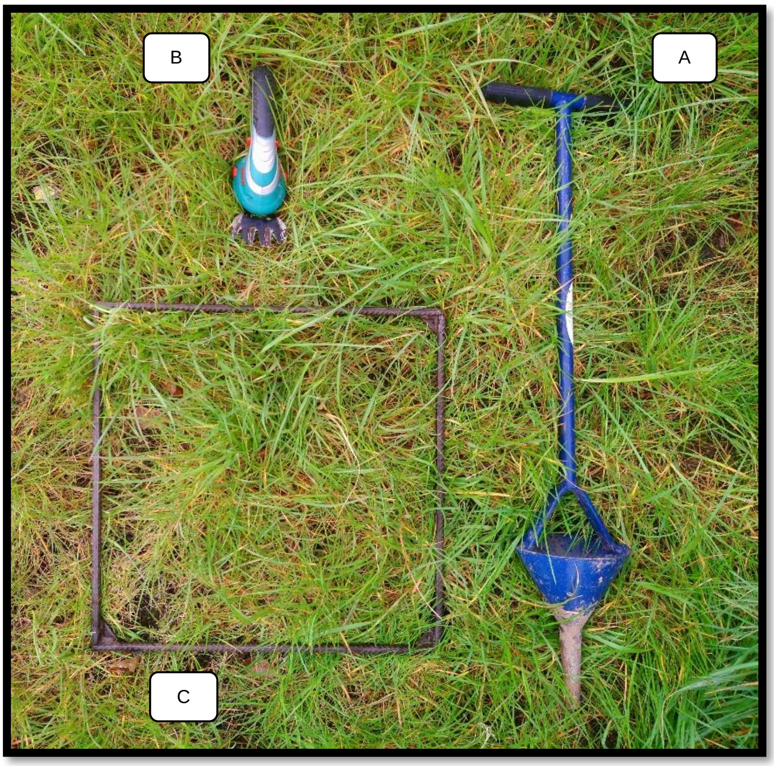

Figure 9-1. A) Stainless-steel soil sampler with a 100 mm length sampling cone, B) Battery grass shears (Bosch ASB 10.8 V), used for herbage sampling, C) Quadrat (0.5 m X 0.5 m), used for herbage sampling.

Figure 9-2. D) Stainless steel soil auger used for soil texture/drainage class assessment.

Figure 9-3. E) Tray used to hold soil auger sample for assessment F) Quadrat (0.5 X 0.5 m, with string subdivisions) used for herbage species composition survey.

Appendix 1B – Coefficient values

Table 10-1 Calculation of organic stocking rates based on annual nutrient excretion rates for livestock, as specified in NAP regulations S.I. No. 605 of 2017 (EU, 2017) and S.I. No. 65 of 2018, EU, (2018).

Livestock type	Nitrogen (kg year^{-1})	Phosphorus (kg year^{-1})
Dairy cow	85	13
Suckler cow	65	10
Cattle (0-1 year old)	24	3
Cattle (1-2 year old)	57	8
Cattle (>2 year old)	65	10

Table 10-2 Average available N, P, and K in cattle slurry and farmyard manure applied to soils with medium or high P and K index (3 and 4) in either spring or summer adapted from Wall and Plunkett, (2016).

Organic manure	Collected(3) as	Nitrogen (kg t^{-1})		Phosphorous (kg t^{-1})	Potassium (kg t^{-1})
Cattle slurry(1)	Tonnes ha^{-1}	0.7 (Spring)	0.4 (Summer)	0.5	3.5
Farmyard manure(2)	Tonnes ha^{-1}	1.4		1.2	6.0

(1). Assuming all slurry was applied as splashplate.

(2). Assuming a dry matter of 20%.

(3). Every calendar year 2015 to 2017.