Environmental and economic aspects of nitrogen and phosphorus use efficiency on intensive grass-based dairy farms in the South of Ireland

Thesis submitted for the degree Doctor of Philosophy

By

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Declaration

I declare that this thesis has not been previously submitted for a degree to this or any other institution and I further declare that the work is, unless otherwise stated, entirely my own work.

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Abstract

The large use of nitrogen (N) and the finite nature of global phosphorus (P) resources have led to increasing concerns about balancing agronomic, environmental and economic gains from N and P use on dairy farms. Nitrogen and P inputs, in the form of fertiliser and concentrates, are key drivers of increased herbage yields and milk saleable output on most dairy farms. However, N and P inputs typically exceed N and P outputs in milk and livestock exported off the farms. Increased N and P use efficiency (NUE and PUE) may be considered as a strategy to reduce the expenditures on the main N and P inputs on dairy farms. Data from a 3 year (2009-2011) survey were used to assess farm-gate N and P balances and NUE and PUE on 21 intensive grass-based dairy farms operating under the Good Agricultural Practice (GAP) regulations in Ireland, as well as the economic implications of NUE and PUE on 19 of these farms and the sensitivity of net profit to changes in milk and fertiliser N prices. Comparative profitability and sensitivity to changes in milk and fertiliser N prices of ten N fertilised grass (FN) and eight grass-white clover-based (WC) dairy systems were also investigated. Mean balances for the 21 farms were 175 kg N ha^{-1} and 5.09 kg P ha^{-1}, respectively, or 0.28 kg N kg MS^{-1} (milk solids), and 0.004 kg P kg MS^{-1}, respectively. Mean NUE was 0.23 and mean PUE was 0.70. Comparison to similar studies carried out before the introduction of the GAP regulations in 2006 indicates that N and P balances have significantly decreased (by 40 and 74 %, respectively) and NUE and PUE increased (by 27 % and 48 %, respectively), mostly due to decreased inorganic fertiliser input and a notable shift towards spring application of organic manures. Mean net profit was €598 ha^{-1} and was driven mainly by milk receipts and to a lesser extent by expenditure on mineral fertilisers. Net profit was indirectly related to N and P surplus and N and P use efficiency. The results of this study generally indicate that Irish dairy farms, as low input production systems, have the potential to achieve both economic (as indicated by net profit per ha) and environmental (as indicated by N and P balances per ha, N and P use efficiency and N-eco-efficiency) sustainability. The results of the sensitivity analysis indicated that milk price was the main driver for changes in net profit between 2009 and 2011 both in high and low milk price situations investigated across nine price scenarios. Net profit was similar for FN and WC (€1,274 ha^{-1}) mainly due to €148 ha^{-1} lower expenditure on mineral N fertiliser on WC. Net profit of WC was found to be comparably less sensitive than FN in low milk price situations. A wider adoption of WC on farms offers potential to meet the twin goals of a sustainable income for dairy farmers in the context of rising fertiliser N price while decreasing N surpluses on pasture-based dairy farms.

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List of abbreviations

Anim. = Animals

ARC = Agricultural Research Council

BIO = Biological Input-Output budget

BNF = Biological Nitrogen Fixation

CAN = Calcium Ammonium Nitrate

CAP = Common Agricultural Policy

CH_3COONa = sodium acetate

CH_3COOH = acetic acid

Concentr. = concentrates

CSO = Central Statistics Office

DAFM = Department of Agriculture, Food and the Marine

DAFWA = Department of Agriculture and Food of Western Australia

DAIRYMIS = Dairy Management Information System

EIO = Economic Input-Output budget

EPA = Environmental Protection Agency

ERDF = European Research and Development Fund

EU = European Union

FN = systems of dairy production based on N fertilised grass

FPCM = fat and protein corrected milk

FYM = farmyard manure

GAP = Good Agricultural Practice

GBM = Grange Beef Model

GD = number of days spent grazing

GFCM = Grange Feed Costing Model

H_2PO_4 = dihydrogen phosphate

IFSM = Integrated Farm System Model

LU = Livestock Unit

Man. = manure

MDSM = Moorepark Dairy System Model

Min. Fert. = mineral fertiliser

MRP = molybdate reactive phosphorus

MS = milk solids

NAP = National Action Programme

NFRV = mineral N fertiliser replacement value

No. = number

NUE = nitrogen use efficiency

NMR = nuclear magnetic resonance

PMP = potentially mobile phosphorus

PO_4 = phosphate

PUE = phosphorus use efficiency

REPS = Rural Environment Protection Scheme

S = price scenario

SFP = Single Farm Payments

S. I. = statutory instrument

SOM = soil organic matter

SR = stocking rate

STP = soil test phosphorus

SUP = soluble unreactive phosphorus

temp. = temperature

TP = total phosphorus

TRIO = Transfer-Recycle-Input-Output budget

TUAA = total utilised agricultural area

WC = systems of dairy production based on grass-white clover grassland

WFD = Water Framework Directive

1. Introduction

1.1. General Introduction

Irish dairy production systems tend to be relatively intensively managed compared to other Irish grassland agricultural production systems, and are pasture-based, with the objective of producing milk in a low cost system through maximising the proportion of grazed grass in the cows' diet (Shalloo *et al.*, 2004a; McCarthy *et al.*, 2007; Ryan *et al.*, 2011). Increasing the proportion of grazed grass reduces milk production costs and can increase the profitability of grass based milk production systems in Ireland and other temperate regions (Dillon *et al.*, 2005; Dillon, 2011). Nitrogen (N) and phosphorus (P) inputs, in the form of fertiliser and concentrate feeds, are key drivers of increased herbage yields and milk saleable output on most dairy farms (Aarts, 2003; Spears *et al.*, 2003a; Treacy *et al.*, 2008; Ryan *et al.*, 2011; Gourley *et al.*, 2012). However, N (Jarvis, 1993; Goodlass *et al.*, 2003; Aarts, 2003; Humphreys *et al.*, 2008) and P (Van Keulen *et al.*, 2000; Spears *et al.*, 2003b) inputs typically exceed N and P outputs in milk and livestock exported off the farms. This imbalance results in surplus N (Gourley *et al.*, 2010; Cherry *et al.*, 2012) and P (Van Keulen *et al.*, 2000; Arriaga *et al.*, 2009) that is either accumulated on (N and P surplus), or lost from, dairy farms.

As N surplus is commonly associated with excessive, inefficient N use on farms, as well as harmful environmental impacts (Leach and Roberts, 2002; Eckard *et al.*, 2004; Powell *et al.*, 2010), it is considered an indicator of potential N losses and environmental performance (Schröder *et al.*, 2003; Carpani *et al.*, 2008). Nitrogen surplus potentially accumulates in soil organic matter (SOM) (Jarvis, 1993) or is lost through denitrification, nitrate (NO_3) leaching, ammonia (NH_3) volatilisation (Pain, 2000; Jarvis and Aarts, 2000; Del Prado *et al.*, 2006) and through runoff to surface waters (De Vries *et al.*, 2001). Denitrification is naturally facilitated in Ireland, due to common anaerobic soil conditions and the generally high content of organic carbon (C) in soils (between 2 and 7 %; Dillon and Delaby, 2009) enabling activity of denitrifying bacteria. These N losses can have negative environmental impacts, such as eutrophication of surface waters, pollution of groundwater aquifers, ozone depletion, and anthropogenic climate change (in the case of N_2O emissions) (Leach and Roberts, 2002; Eckard *et al.*, 2004; O'Connell *et al.*, 2004).

The P surplus does not predict the actual losses and loss pathways, but it is considered a long-term risk indicator of the P losses (Jarvis and Aarts, 2000). However, unlike N surpluses, which are seen as an economic waste and potential environmental problem, P surpluses may be necessary on farms where an increase in soil P content is required (Culleton *et al.*, 1999). However, the P surplus may accumulate in the soil (Gourley *et al.*, 2010) or may be lost in eroded material containing particulate P or P adsorbed on to organic-rich clay soil fractions (Kurz *et al*., 2005) or through leaching (Heathwaite, 1997). These P losses can have negative environmental impacts such as eutrophication of surface waters (Clenaghan *et al.*, 2005), and pollution of groundwater aquifers (Heathwaite, 1997). In Ireland, P is the major limiting nutrient in surface waters and increased additions may result in algal blooming (McGarrigle, 2009).

It has been emphasised that dairy production should ideally be achieved in a sustainable manner, without impairing natural capital (soils, water, and biodiversity) (Goodland, 1997). Improved nutrient use efficiency has a significant role to play in the development of more sustainable dairy production systems (Goulding *et al*., 2008). Therefore, there is an on-going debate surrounding the use of high- or low-input systems in dairy farming. The low-input systems are considered more economically and environmentally sustainable than the high-input systems (Ridler, 2008, O'Brien *et al.*, 2012) as they are less vulnerable to volatility in input and output prices (Humphreys *et al.*, 2012; Moreau *et al.*, 2012) and are associated with lower farm nutrient surpluses (Humphreys *et al.*, 2008; Ledgard *et al.*, 2009).

Among the nutrient imports in dairy production systems, N is particularly important as it is used in large quantities, between 172 and 301 kg N ha^{-1} (Groot *et al.*, 2006; Nevens *et al.*, 2006; Roberts *et al.*, 2007; Ryan *et al*., 2011; Cherry *et al.*, 2012) but with generally low efficiency (Goulding *et al.*, 2008). These high fertiliser applications may often be attributed to risk aversion to lower crop yields or to relatively low fertiliser prices. The lower the relative price of fertiliser, the greater the incentive to apply it to offset potential risk and yield uncertainty (Buckley and Carney, 2013). Also, the volume of bought-in feeds is often driven more by the desire to produce specific volumes of product rather than by the desire to make the most efficient use of inputs. Concurrently, there has been a general tendency to overlook the importance of the 'free' resource (pasture and soil nutrient supply) (O' Connell *et al*., 2004; Ridler, 2010).

In grass-based dairy production systems, there are a number of factors limiting NUE, such as N losses from manure, slurry and mineral fertiliser management and application to land (Webb *et al.*, 2005), losses from dung and urine deposited by grazing animals, the ability of grass plants to convert N from applied mineral fertiliser and manure into biomass in herbage, utilisation by animals of grass herbage grown and the biological potential of cows to convert N from concentrate feeds and herbage into milk (Powell *et al.*, 2010). More effective use of N imports in fertiliser N and concentrate feeds can potentially contribute to decreased imports and increased rates of NUE (Groot *et al.*, 2006). Increased nutrient (N and P) use efficiency (Gourley *et al.*, 2010) may be also considered as a strategy to reduce the expenditures on the main N and P inputs (mineral fertilisers and concentrate feeds) on dairy farms.

There is also great concern for efficient P use on intensive farming systems and reduction of P losses to the environment, due to the finite nature of global P resources (Simpson *et al.*, 2011; Huhtanen *et al.*, 2011; Cordell *et al.*, 2011). Besides the factors mentioned that affect the levels of NUE on grass-based dairy production systems, there are additional factors affecting PUE, including soil P-sorption capacity in relation to soil P inputs, uneven distribution of excreta leading to uneven soil P content (in grazing enterprises), and P losses resulting in accumulation of P as sparingly-available phosphate (PO_4) in the soil (Simpson *et al.*, 2011).

Losses of N and P also incur economic costs in two ways; the cost of wasted N and P inputs, at farm level, and the cost of clean-up associated with pollution caused as a result of such losses, at national level (Buckley and Carney, 2013). It has been proposed that these costs should be factored into the sale price of milk (Von Keyserlingk *et al.*, 2013).

In the European Union (EU), the Nitrates Directive (91/676/EEC) (European Council, 1991) has established guidelines in relation to farming practices to reduce NO_3 leaching that are implemented in each member state through a National Action Programme (NAP). In Ireland, these are legislated as the Good Agricultural Practice (GAP) Regulations (European Communities, 2010), first passed in 2006. Under the Regulations, farms are limited to a stocking rate (SR) of 2 livestock units (LU) ha^{-1}, or 2 dairy cows ha^{-1}. The Regulations also establish the quantity of available N and P that can be applied to grass and other crops (depending on factors such as SR, soil test P (STP), or crop type), and soil indices.

The Nitrates Directive is complemented by the Water Framework Directive (WFD) (2000/60/EC) (European Council, 2000), with the main aim to bring the water quality in all waters in EU to “good ecological status” by 2015 and no later than 2027 (Jacobsen, 2009). In Ireland, the WFD was first implemented as Water Policy Regulations (European Communities 2003), in 2003. To reduce the pollution of waters and ensure security of drinking water, these regulations established a limit of 0.03 mg Molybdate Reactive Phosphorus (MRP) litre^{-1} or 35 µg PO_4 litre^{-1} (European Communities 2009). This threshold has also a limiting effect on P use on farms.

Although explicitly aimed at decreasing N losses to water, these Regulations might be expected to lead to improved NUE and PUE on farms, as most of the measures aim to decrease losses by increasing retention of N and P within the production systems. However, most of the existing data on grassland-based dairy farm N (Mounsey *et al.*, 1998; Treacy *et al.*, 2008) and P (Mounsey *et al.*, 1998; Treacy, 2008) balances in Ireland date from the period before the implementation of the Regulations in 2006. There is only one study on NUE and PUE (Buckley *et al.*, 2013) on grassland-based dairy farms after the implementation of the Regulations. In the European context also, there are few farm-gate N (Groot *et al.*, 2006; Nevens *et al.*, 2006; Roberts *et al.*, 2007; Cherry *et al.*, 2012; Oenema *et al.*, 2012) and P (Van Keulen *et al.*, 2000; Aarts, 2003; Swensson, 2003; Nielsen and Kristensen, 2005; Raison *et al.*, 2006) balances on grassland-based dairy farms post the implementation of the Nitrates Directive.

Besides the restrictions regarding N and P use, which may negatively affect the herbage (Hennessy *et al.*, 2008; Power *et al.*, 2005) and milk (Shalloo *et al.*, 2004b; McCarthy *et al.*, 2007) yields on grassland-based dairy farms, the dairy farmers also face increasing volatility of market price received for sold milk. This is because in 2008, the "Health Check" decisions of the Common Agricultural Policy (CAP) included the expiry of the milk quota system, which is expected to take place in 2015 in Ireland. It is anticipated that this will create an imbalance between milk supply and milk demand and therefore high milk price volatility (Kelly *et al.*, 2012). In addition, increasing input prices (Soder and Rotz, 2001), as well as rising labour, machinery and animal housing costs (MacDonald *et al.*, 2008) are leading dairy farmers to search for ways to decrease milk production expenditures, and grazed grass-based dairy systems offer opportunities to reduce these expenditures during the grazing season (Soder and Rotz, 2001; MacDonald *et al.*, 2008). Strategies to reduce expenditures in grazed grass-dairy production systems include increasing resource use efficiency (Ridler, 2008; Finneran *et al.*, 2011; Patton *et al.*, 2012; Kelly *et al.*, 2013), nutrient use efficiency (Gourley *et al.*, 2010), N-eco-efficiency (the amount of milk produced per kg of N surplus) (Nevens *et al.*, 2006; Beukes *et al.*, 2012), accounting for mineral nitrogen fertiliser replacement value

(NFRV) of organic N contained in slurry (Lalor, 2008) or fixed by white clover in pastures (Humphreys *et al.*, 2012). Therefore, there has been a rejuvenated interest in grass-based dairy production systems internationally (MacDonald *et al.*, 2008) as a low-input, low-cost system that may be less vulnerable to volatility in input and product prices.

Under these conditions, work has been undertaken on grass-based dairy farms in Europe with specific focuses on N (Groot *et al.*, 2006; Nevens *et al.*, 2006; Roberts *et al.*, 2007; Treacy *et al.*, 2008; Cherry *et al.*, 2012; Oenema *et al.*, 2012) or P (Mounsey *et al.*, 1998; Van Keulen *et al.*, 2000; Steinshamn *et al.*, 2004; Nielsen and Kristensen, 2005; Huhtanen *et al.*, 2011) use efficiency and the economic impacts of implementing the Nitrate Directive (Van Calker *et al.*, 2004) and Water Framework Directive (Jacobsen, 2009). In Ireland, the economic implications of compliance with the Nitrate Directive and decoupling of single farm payments (SFP) on dairy farms were investigated by Hennessy *et al.* (2005) and those of milk quota abolition were investigated by McDonald *et al.* (2013). Crosson *et al.* (2007) investigated economic beef production systems in relation to N and P management strategies. Buckley and Carney (2013) investigated economic impacts of the management of N and P inputs from mineral fertilisers and feeds on dairy farms based on one-year data. However, none of the above studies included both economic impacts of N and P use efficiencies, economic implications of compliance with Nitrate Directive regulations, and sensitivity to volatility of milk and mineral fertiliser prices on grazed grass-based dairy farms.

1.2. Aims and Objectives

1. **Nitrogen balance and use efficiency on twenty-one intensive grass-based dairy farms in the South of Ireland (Chapter 3)**

i. to assess farm-gate N balances and use efficiencies on 21 intensive grass-based dairy farms operating under the Nitrate Regulations in Ireland and compare these to pre-Regulations studies to investigate the impact of the Regulations;

ii. to identify the factors influencing NUE on these farms;

iii. to explore potential approaches to increase NUE and decrease N surpluses on these farms.

2. **Phosphorus balance and use efficiency on twenty-one intensive grass-based dairy farms in the South of Ireland (Chapter 4)**

i. to assess farm-gate P balances and use efficiencies on 21 intensive grass-based dairy farms operating under the Nitrate Regulations in Ireland and compare these to pre-Regulations studies to investigate the impact of the Regulations;

ii. to identify the factors influencing PUE on these farms;

iii. to explore potential approaches to increase PUE and decrease P surpluses on these farms.

3. **Economic impacts of nitrogen and phosphorus use on nineteen intensive grass-based dairy farms in the South of Ireland (Chapter 5)**

i. to assess the economic impacts of N and P farm-gate balances and use efficiencies on 19 intensive grass-based dairy farms;

ii. to assess economic implications of compliance with the Nitrate Directive regulations on these farms;

iii. to assess the sensitivity of net profit of these farms to volatility in milk and fertiliser prices.

4. **An economic comparison of systems of dairy production based on N fertilised grass and grass-white clover grassland in a moist maritime environment (Chapter 6)**

i. to evaluate the potential of white clover to replace fertiliser N and contribute to the profitability of pasture based systems of dairy production in the context of recent fertiliser N and milk prices

ii. to assess the comparative sensitivity to the volatility of milk and fertiliser prices of eight white clover-based and ten fertilised grass-based dairy production systems.

1.3. Thesis layout

The thesis contains seven chapters including list of references at the end of each chapter. Following the general introduction, Chapter 2 contains a literature review explaining climatic conditions of Ireland, Irish dairy production systems, N and P cycling in grassland soils, as well as strategies to control N and P losses from dairy farms. Chapter 3 investigates N balance and use efficiency on 21 intensive grass-based dairy farms. Chapter 4 investigates P balance and use efficiency on 21 intensive grass-based dairy farms. Chapter 5 assesses the economic performance and economic impacts of N and P use on 19 intensive grass-based dairy farms. Chapter 6 compares the profitability of eight white clover-based and ten N fertilised grass-based dairy production systems. Chapter 7 includes a general discussion.

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2. Literature review

2.1. Grassland in Ireland

2.1.1. Climate

Ireland has a west maritime climate, with mild moist winters and cool cloudy summers. Maritime air associated with the Gulf Stream contributes to mild temperatures (Finch and Gardiner, 1993). Therefore, Ireland benefits from climatic conditions allowing grass growth also during winter, albeit at lower rates (Ryan *et al.*, 2010). The air temperatures range between 5.1º C in January and 14.7º C in July, and annual rainfall ranges between 800 and 1,200 mm evenly distributed throughout the year. In addition, soil moisture deficit does not impede grass growth during the summer (Humphreys, 2008).

The Irish soils have naturally low suitability for agricultural activities. Main soil groups encountered in Ireland are shown in Table 2.1.

Table 2.1. Main soil groups in Ireland (Gardiner and Radford, 1980)

Physiographic division	Main soil group (%)	Description	Suitability
Mountain and hill	Brown Podzolic (80%)	-sandy loam -excessively drained -low nutrient status -acidic	Cropping and pastures
Rolling lowland	Lithosols (80%)	-bedrock	Rough grazing
Drumlins and flat to undulating lowland	Gleys (85%)	-clay loam -impeded drainage	Unsuitable for cultivation

2.1.2. Effect of temperature on grass growth

Grass growth rates are directly influenced by air and soil temperature. In Ireland, grass growth rates were recorded of 18.3 kg DM ha^{-1} day^{-1} at 8.2 ^{0}C and of 35.9 kg DM ha^{-1} day^{-1} at 15.6 ^{0}C (Brereton and Hope-Cawdery, 1988). At a soil temperature of 6 ^{0}C the grass starts growing (Ryan *et al.*, 2010), reaching rates of between 60 and 80 kg DM ha^{-1} day^{-1} at temperatures between 15 and 20 ^{0}C (Treacy *et al.*, 2008).

2.1.3. Annual herbage production

In Ireland, herbage production is mainly influenced by N fertilisation rates. Herbage yields of 5.64 t DM ha^{-1},on average, can be achieved with zero fertiliser N application, from N supplied by soil, while fertilisation N rates between 250 and 330 kg N ha^{-1} can increase herbage production up to between 9.85 and 16.66 t DM ha^{-1} (Ryan, 1974; Hennessy *et al.*, 2008).

Generally, herbage production is highly seasonal on Irish grasslands. For example, herbage yields can reach 2.45 t DM ha^{-1} in spring compared with 1.10 t DM ha^{-1} in autumn. The higher yields in spring are mainly due to accumulated daily temperatures during previous autumn which help sward tillering in spring (O'Donovan *et al.*, 2002).

2.2. Systems of dairy production in Ireland

The system of milk production that has developed in Ireland is a 'hybrid', using grazed pasture, pasture silage and concentrate on an annual basis. This is primarily due to the seasonal nature of grass production and the constraints in grass utilisation due to soil and climatic conditions (Shalloo *et al.*, 2004a).

In Ireland, climatic conditions enable an extended grazing season from February to November (Humphreys *et al.*, 2009a). Therefore, Irish dairy production systems highly rely on grazed grass for the animals' diet and are unique in Europe because the calving period (from January to April) typically matches the beginning of grass growth (Dillon *et al.*, 2005). The proportion of grazed grass in the diet of dairy stock is hence maximised (Humphreys *et al.*, 2009a), allowing for maximum amount of milk to be produced from grazed grass and reducing requirements for concentrate feeds post-calving (Dillon *et al.*, 1995). Increasing the proportion of grazed grass in animals' diets, particularly in early spring, reduces milk production costs and can increase the profitability of grazed grass based-dairy production systems in Ireland (Dillon, 2011).

During the grazing season, grazed grass commonly accounts for 60 to 75 % of the diet of dairy livestock, which is supplemented by silage (23 %) and concentrates (10 %) when drier years occur (963 mm $year^{-1}$) and in the winter time (Humphreys *et al.*, 2009a).

Due to low dry matter (DM) content of grass, in Ireland the milk yields are typically low, between 3,500 and 6,500 kg milk cow^{-1} year^{-1} (Humphreys *et al*, 2009) going up to almost 6,800 kg milk cow^{-1} year^{-1} under experimental conditions, with supplementation (concentrates) levels of up to 1,445 kg DM cow^{-1} year^{-1} (McCarthy *et al.*, 2007).

2.3. Stocking rate in dairy production systems in Ireland

After the implementation of GAP Regulations, in 2006, the stocking rate (SR) was limited at 2 up to 2.9 dairy cows per ha, under derogation conditions (European Communities, 2010a). On experimental farms, SR varied between 2.44 and 2.90 LU ha^{-1} (Horan, 2009; Coleman *et al.*, 2010; Ryan *et al.*, 2012; McCarthy *et al.*, 2012). The objective of most of the research work was to investigate increased productivity of dairy production systems through increased SR associated with increased grazed grass in animals' diet while possibly decreasing concentrate and mineral fertiliser inputs. The results showed that at lower SRs (2.44 LU ha^{-1}; Ryan *et al.*, 2012) there was lower grazed grass intake, of 2,950 kg DM cow^{-1}, compared with 4,051 kg DM cow^{-1} (McCarthy *et al.*, 2012) at higher SRs, of 2.90 LU ha^{-1}. On the other hand, concentrate intake decreased from 408 kg DM cow^{-1} (Ryan *et al.*, 2012) to 236 kg DM cow^{-1} (McCarthy *et al.*, 2012), but fertiliser N rate stayed around 245 kg N ha^{-1} at both SRs. However, milk yield was 5,186 kg cow^{-1} at the SR of 2.44 LU ha^{-1} (Ryan *et al.*, 2012), similar to 5,286 kg cow^{-1} at the SR of 2.90 LU ha^{-1} (McCarthy *et al.*, 2012). This is because at higher SRs, milk production per cow is reduced due to reduced daily herbage allowance and intake associated with increased grazing severity, and the inability of the animal to select greater quality herbage from within the sward (McCarthy *et al.*, 2011).

The studies of Ryan *et al.* (2012) and McCarthy *et al.* (2012) indicated that there is potential to reduce concentrate inputs, and therefore expenditures, on Irish grass-based dairy production systems. However, attention needs to be paid to the balance between feed supply and demand as influenced by SR, because an imbalance will result in either underfeeding of the herd or waste of excess feed (McCarthy *et al.*, 2011).

Stocking rate is particularly low (between 1.4 and 2.1 cows ha^{-1}) on grass/white clover swards (Humphreys *et al.*, 2008; Keogh *et al.*, 2010; Humphreys *et al.*, 2009b), compared with mineral N fertilised swards (2.25 cows ha^{-1}; Humphreys *et al.*, 2008;

Humphreys *et al.*, 2009b). This is because higher SRs are not supported by the former due to lower herbage production, of 10.81 t DM ha^{-1}, on average, compared with the latter (12.61 t DM ha^{-1}, on average; Humphreys *et al.*, 2008; Humphreys *et al.*, 2009b).

2.4. Models

In the field of animal research, models are used for investigating components, systems and management (Shalloo *et al.*, 2004b). Modelling the efficiency of N and P use in dairy stock (Van Keulen *et al.*, 2000; Steinshamn *et al.*, 2004) is an example of component research. In system research, modelling is aimed at characterising and understanding the interactions occurring between components at the system level. In management research, models are used to investigate the effects of management options on output (production, returns, and risk) of the system (Shalloo *et al.*, 2004b).

Simulation modelling and linear programming or optimisation are the two techniques commonly used to model agricultural systems (Crosson *et al*., 2006). Simulation models of agricultural systems are developed to accurately describe the evolution of the systems. They provide the opportunity to explore difficult relationships that cannot be explored in any other way. Optimisation models aim at optimising some criteria subject to a set of constraints (Shalloo *et al*., 2004b) and may lead to identification of optimal systems (Crosson *et al*., 2006).

2.4.1. Simulation models

In Ireland, farm simulation models have a role as direct extension and management tools, i.e. evaluating alternative production systems (Shalloo *et al.*, 2004b). A brief description of simulation models developed in Ireland is presented below.

Due to prevalence of grazed grass-based production systems in Ireland (Dillon *et al.*, 2005), a grazing model was required to initiate management interventions, such as removal of baled silage or feeding of silage, and to explore the effects of alterations in the sward height at which grazing was terminated on farm performance. This model simulates the change over time in the frequency distribution of exposed herbage strata types and the distribution of cows across this range of herbage strata (Brereton *et al*., 2005).

The Moorepark Dairy System Model (MDSM) (Shalloo *et al.,* 2004b) was constructed to allow investigation of the effects of varying biological, technical, and physical processes on farm profitability. This farm simulation model was developed to examine aspects of seasonal grass-based systems of production using minimal concentrate for animals' diet. This model used budgetary simulation and stochastic modelling of a milk production system. The budgetary simulation incorporated the biological (milk yield and composition, bodyweight, nutritional requirement, fertiliser), physical (land, labour, buildings), and economic (costs, valuations, profit and loss account, balance sheet) processes on a simulated typical Irish dairy farm. Stochastic budgeting, using Monte Carlo simulation, was used to determine the influence of variation in milk price, concentrate costs, and silage quality on farm profitability (Shalloo *et al.,* 2004b).

The Integrated Farm System Model (IFSM) is a simulation model incorporating the various farm processes that control animal performance and nutrient flows in livestock production systems. It can be used to evaluate the long-term performance and environmental impact of beef production systems. Land use, inorganic fertilisation rates, and animal production details must be specified by the model user. A least cost-ration can be determined by linear programming. Based on the diet fed, the quantity and nutrient (N, P, K) contents of the manure produced was determined. The environmental impact was assessed through manure N and P losses (Crosson *et al*., 2007).

Geary *et al*. (2010) developed a processing-sector model simulating milk collection, standardization, and product manufacture. The model is a mathematical representation of the process of conversion of milk into dairy products, accounting for all inputs, outputs, and losses involved in dairy processing. Within the model, the production of each of the dairy products was simulated (cheese, butter, whey milk powder, skim milk powder, fluid milk, and casein). The key model inputs of volume and composition of milk intake and product portfolio and its composition were used in the simulations. Processing costs were simulated, and the return from raw milk and its individual component values was calculated.

Ryan *et al.* (2011) developed an N balance model capable of investigating scenarios relevant to grass based milk production systems and an investigation into the effects of autumn closing date and spring turnout date on winter and spring herbage production was completed. The N balance model evaluated annual N use efficiency, N surpluses and N losses to the environment of contrasting grass based milk production systems. The scenarios investigated show that systems using different system boundaries return different N surpluses and therefore different N use efficiencies. At systems level, up to 80 % of N was not utilised. To ensure adequate herbage mass for grazing in early spring swards should be closed before November. Increasing the number of days spent grazing increased the quantity of N exported by 5 % and reduced total N loss to the environment by 8 %. Increasing stocking rate by 0.5 LU ha^{-1} and maintaining N fertiliser application rate reduces surplus N per kg MS produced by up to 41 %. Reducing N input by an average of 23 % reduced N surplus, increased N use efficiency and reduced N loss by an average of 29 %, 25 % and 18 %, respectively. Nitrogen surplus estimates proved to be a good predictor of NO_3 leaching loss with an average of 23 % of N surplus being leached into ground water. The study showed that there is great potential from grass based milk production systems to optimise economic return through increasing grazing season length and increasing stocking rate while minimising the risk of N losses to the environment.

However, a major limitation of computer simulation technique is limited confidence in the simulation results. Only through extensive evaluation of the model can confidence be gained in the results and recommendations derived from such modelling technique (Ryan *et al.*, 2011).

2.4.2. Optimisation models

The Grange Beef Model (GBM) is a linear programming model designed to identify financially optimal beef production systems in Ireland within a given range of resource and economic parameters. It is constructed around a typical beef cow herd, including beef cows, replacement heifers, calves, stockers, and finishing animals. Nutritional needs of each group are described in terms of energy requirements and intake capacity. Budgets are formulated for each on-farm activity. These budgets assign a cost or revenue to each activity and, based on these, the program identifies the optimal farm gross margin (Crosson *et al.*, 2007).

2.4.3. *Application of models*

Simulation models are best evaluated by comparing simulated results with actual similar data or direct measurements. For example, Brereton *et al.* (2005) evaluated the grazing model by comparing the modelled time series of herbage mass and height with the changes measured during grazing by steers on four perennial ryegrass swards. Shalloo *et al*. (2004b) compared actual data from 21 dairy herds and simulated data to determine the reliability of key model outputs.

In case of GBM and IFSM models, prior to using both models, it was necessary to ensure that IFSM accurately replicated GBM results in terms of animal performance and forage yields on Irish beef farms. Therefore, a component-based comparison was undertaken. The forage yield and response to N fertiliser were first compared. Then GBM was solved to find the financially optimal system in the policy and market environment prevailing in 2005. The IFSM was subsequently run using the resulting optimal system parameters predicted by GBM in terms of land use, fertilisation rate, and animal production. Animal intake and total feed use predicted by the two models were then compared (Crosson *et al*., 2007).

Geary *et al*. (2010) evaluated the milk processing sector by comparing the impact of variation in milk composition (specific to Irish Holstein-Friesian, Jersey, and New Zealand Holstein-Friesian) on the volume of product produced, processing costs, product sale value, component value of milk, and net value of milk. Also, two product portfolio scenarios were investigated using the model to demonstrate the change in processing costs, product sale value, component values of milk, and the net value of milk as the Irish product portfolio changed from 2000 to 2008.

2.4.4. *Nutrient budget models*

Nutrient budgets are commonly used in agriculture to characterise nutrient management and quantify the magnitude of nutrient flows. More often, nutrient budgets are applied at the farm-scale (Cherry *et al*., 2012).

Watson and Atkinson (1999) differentiated three basic approaches in N budget studies: (i) the economic input-output (EIO) budget, based entirely on farm information on the quantities of N purchased and sold over the farm-gate. This approach allows nutrient budgeting only by using information from farm records. Any calculated surplus of inputs over known outputs is assumed to be lost from the farms; (ii) biological input-output (BIO) budget, which, in addition to information on purchases and sales of nutrients over the farm-gate, includes inputs from symbiotic N_2 fixation (via legumes) and atmospheric deposition. The fate of possible N surplus is assumed to be similar to the surplus occurring in EIO budgets; (iii) transfer-recycle-input-output (TRIO) budget includes all the information from the BIO budget, but takes also into consideration the major internal soil N fluxes (mineralisation and immobilisation) largely predicted from the literature. It therefore allows for a build-up or decline in soil N. The TRIO approach has also the ability to predict the internal cycling of N within a farming system. This helps assessing the reliance of the system on internal nutrient sources, and therefore the sustainability of the system. The purpose of the study dictates the choice for one of the three budgeting approaches (Oenema *et al*., 2003).

Farm scale nutrient balances can take the form of farm-gate or whole farm balances (Buckley *et al.,* 2013). In most livestock farm-scale budgets, the common inputs of nutrients were in the form of atmospheric deposition, purchased mineral fertilisers, feedstuffs, bedding materials, livestock, and imported manure, whereas the common sources of nutrient exports were in saleable product (crops, milk, meat) or exported manure (Mounsey *et al*., 1998; Aarts, 2003; Nielsen and Kristensen, 2005; Groot *et al*., 2006; Nevens *et al*., 2006; Roberts *et al.,* 2007; Treacy *et al.*, 2008; Oenema *et al.*, 2012). Cherry *et al.* (2012) excluded atmospheric deposition, considering it as an input which is beyond farmers' direct control. The N sourced from biological N fixation (BNF) by white clover was included as an input by Oenema *et al.* (2003), Nevens *et al.* (2006) and Cherry *et al.* (2012), for example, while Aarts (2003) excluded it on the grounds that it is not a farm-gate input. Cherry *et al.* (2012) agreed with Aarts (2003), but they included the N supplied through biological nitrogen fixation as an input because they considered planting of legumes as a strategic nutrient management decision. The N losses and N flows in the soil are not typically included in dairy farm-gate N budgets (Goulding *et al*., 2008).

Nousiainen *et al.* (2011) created the dairy farm nutrient management model Lypsikki, based on three sub-models: (1) soil and crop, (2) dairy herd and (3) manure management. The model was constructed using Microsoft Excel spread sheets. The model uses empirical relationships between input (nutrients for plant growth and dairy cows) and output (crop or milk yield) variables. The input variables needed to run the model are: arable land area for different crops, nutrient import in fertilisers and their allocation for the crops, soil types and STP value, number of animals, production and reproduction parameters of the cows, and composition and nutritive value of feeds. As an output, the model predicts the whole-farm nutrient (N, P) utilisation and surplus (kg ha^{-1}) for one year time period.

Steinshamn *et al.* (2004) assessed N and P use efficiency on organic dairy production systems. They quantified the transfer of N and P to and from the farm and internally between fields, feed stores, animals, and slurry stores. This is because they considered the improvement of internal cycling of nutrients as a way to reduce N and P inputs.

Ryan *et al*. (2011) developed a model to evaluate whole-farm N balances and quantify N use efficiency by identifying surpluses and potential losses associated with contrasting levels of production within grass-based dairy production systems. Differently than other authors, he included N needed to rear dairy replacements as an input and N immobilised in the soil or lost through volatilisation, denitrification, or leaching as outputs.

2.5. Nitrogen in dairy production systems

2.5.1. Nitrogen cycle

Nitrogen is a macronutrient highly important for both terrestrial as well as aquatic ecosystems' productivity (Antikainen *et al.*, 2005) and a key input for pasture systems (Roberts *et al.*, 2007). Nitrogen is required highly by the plants because it plays a major role in photosynthesis in plant leaves (Eickhout *et al.*, 2006). Thus, the organic N in leaves is found mainly in the form of carboxylases, the main photosynthetic enzymes (Parsons and Chapman, 2000). As a component of plants' dry matter (DM) in temperate swards, N accounts for more than 30 g kg^{-1} DM (Humphreys *et al.*, 2003).

Grazed grass-based dairy farms are characterised by numerous N transformations and associated losses that occur during the production process (Roberts *et al*., 2007). These N transformations take place within the N cycle. This is a natural biogeochemical cycle (Abrol and Raghurma, 2007) comprising atmospheric, soil, plant and animal pools of N (McNeill and Unkovich, 2007) altered by management practices in the agro-ecosystems. Within this cycle, at farm level, N is being recycled between the N pools of the agro-ecosystems and within the wider environment. The components of the N cycle on a dairy farm are illustrated in Fig. 2.1. The atmospheric N_2 can be fixed by leguminous plants via BNF (Ledgard *et al.,* 1999) or it can be deposited on soil as NH_4 via rain drops (Asman *et al.,* 1998). Mineral N is taken up by the grass plants which are consumed by grazing dairy livestock (Van Keulen *et al.*, 2000). Ingested N will be converted into milk and meat protein or excreted as dung and urine (Janzen *et al.*, 2003). The N in dung and urine will be returned to the soil either through direct deposition at grazing or via spreading of slurry and manure. The N released from dung and urine will be taken up by plants or immobilised in the soil organic matter (SOM), the excess N being lost to the atmosphere (via volatilisation, denitrification or nitrification) or water (via leaching or overland flow) (Watson, 2001). Nitrogen is also exported off-farm in the sold milk (milk protein) and livestock (meat protein) (Janzen *et al.*, 2003).

Some of the N exported or lost from the system is replaced by atmospheric deposition and BNF. To maintain levels of herbage production, however, the exported or lost N is also replenished through imported mineral N fertilisers, concentrates and forages via the farm-gate (Van Keulen *et al.*, 2000; Aarts, 2003). Nitrogen cycling on grazed grass-based dairy production systems is typically more complicated than other production systems because of the numerous opportunities for N to "escape" from the system, such as losses from manure, slurry and mineral fertiliser management and application to land (Webb *et al.,* 2005), losses from dung and urine deposited by grazing animals, the limited ability of grass plants to convert N from applied mineral fertiliser and manure into biomass in herbage, limited utilisation by animals of grass herbage grown and the biological potential of cows to convert N from concentrate feeds and herbage into milk (Powell *et al.,* 2010). All these losses may negatively impact on farm profitability through decreased production and financial returns from the input costs with fertiliser and feeds (Roberts *et al.*, 2007).

There is, therefore, a need to identify ways to make more efficient use of nitrogen on grazed grass-based dairy production systems (Ryan and Fanning, 1995; Ledgard, 2001). For this purpose, understanding and quantifying nitrogen flows on dairy farms may lead to improved N management and reduced potential for N losses to the wider environment (Scholefield and Fisher, 2000; Spears *et al.,* 2003a).

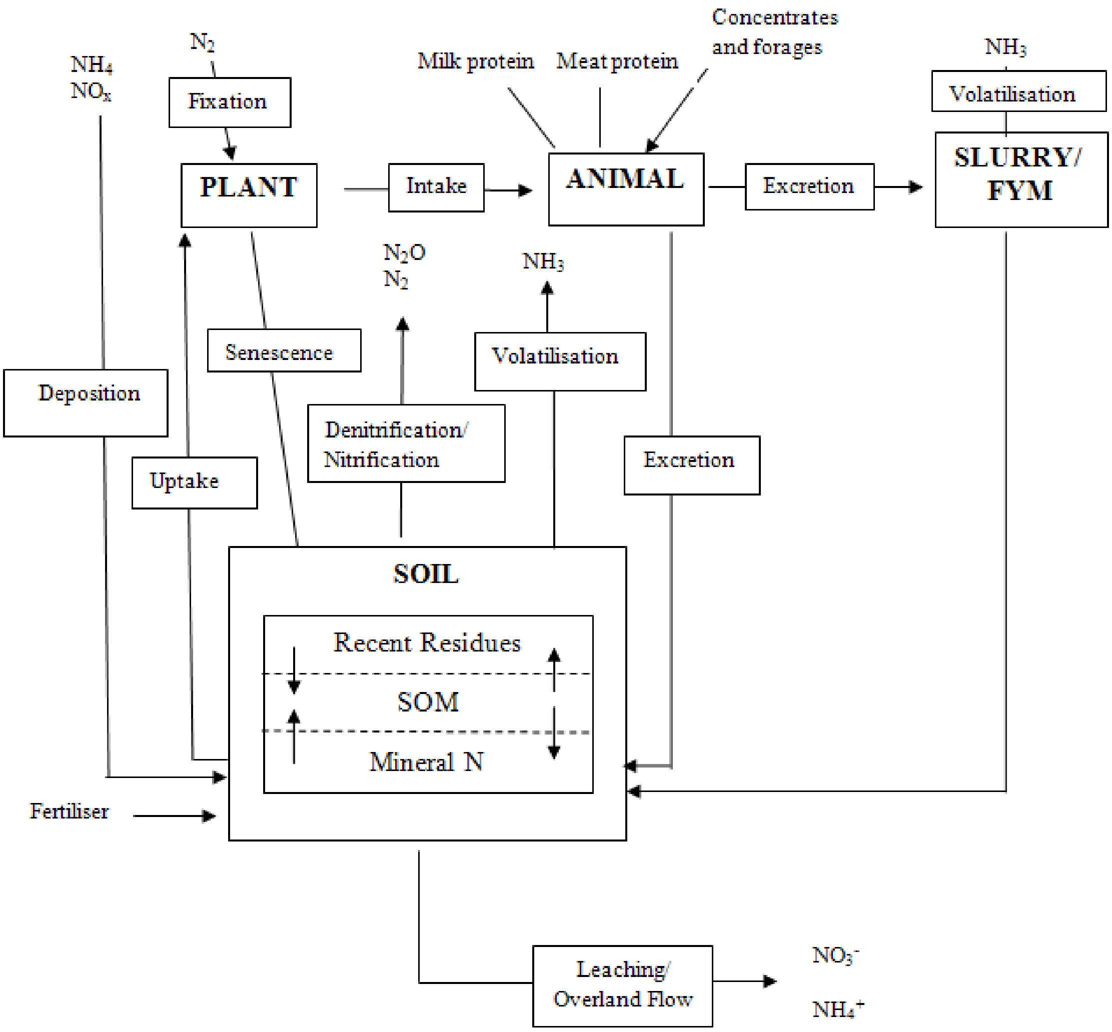


Fig. 2.1. Simplified N cycle in dairy farming systems (adapted from Jarvis and Aarts, 2000)

2.5.2. Nitrogen loss pathways

The potential N losses resulting from N cycling on dairy farms can lead to environmental damage (Aarts, 2003; O'Conell *et al.*, 2004). More precisely, emissions of nitric oxide (NO) and nitrite (NO_2), as products of denitrification, can contribute to ozone depletion; NH_3 volatilisation and subsequent redeposition in NH_4 form can lead to soil and surface water acidification; nitrous oxide (N_2O), resulting from denitrification process, is a harmful greenhouse gas and NO_3 from urine or mineral fertiliser applied to land may leach into groundwater or may reach water courses through run-off (Leach and Roberts, 2002; Verbruggen *et al.*, 2005), contributing to anthropogenic eutrophication of surface waters and pollution of groundwater aquifers. The quantity of N lost to water or atmosphere, in the form of these compounds, is related to factors such as land use, on-farm management of N resources (fertilisers, feeds), soil type and climate (O'Connell *et al.*, 2004).

On dairy farms, N losses may occur due to low N utilisation by ruminants (20 %; Watson and Foy, 2001), the timing and form of N applied to soils (in fertiliser, dung, urine, slurry or manure) potentially not matching plant requirements, and therefore leading to losses of unused N (Aarts, 2003; Gourley *et al.*, 2007). Overall, N losses from dairy farms have been found to be as high as 5 kg for every kg of N in dairy products (Clark and Harris, 1996).

Nitrate leaching, denitrification, and ammonia volatilisation are further discussed below, as they are associated with the most damaging contaminants of water and air.

2.5.2.1. Nitrate leaching

Dairy production systems represent a source of N that may be lost via NO_3 leaching (Del Prado *et al.*, 2006), potentially contaminating ground water and rivers (Eckard *et al.*, 2004) and contributing to anthropogenic eutrophication of water bodies. The main source of NO_3 in dairy production systems is urinary N originating from an excess of rumen-degradable N compared with rumen microbial requirements or from an unbalanced amino-acid supply in relation with animals' requirements. Urinary N is rapidly converted to NH_3 and easily volatilised or leached (Peyraud and Delaby, 2006). Nitrate leaching is the process during which NO_3 exceeding plant demand is moved down the soil profile below the rooting zone (Watson, 2001). This mobile anion can be moved through the soil by percolating water (Burkart *et al.*, 2006; Major *et al.*, 2009) because it is negatively charged, the same as the clay and organic soil particles. Therefore, it is not retained in the soil and it is leached to subsoil (Watson, 2001). Nitrate in soil water which is leached below the root zone may be diluted or denitrified before it gets into the surface or ground waters (Leach *et al.*, 2004). However, the NO_3 concentration in the ground waters should not be higher than 50 mg NO_3-N litre^{-1} (European Communities, 2010a).

In grazed grass-based dairy production systems, N leaching is influenced by several factors: urine deposition, SR, N fertiliser source, timing of N applications, and grazing management. The main source of readily leachable NO_3 is the urine deposited by the grazing livestock on grasslands (Decau *et al.*, 2004). This is because urine is deposited in localised urine patches (O'Connell *et al.*, 2004) containing the equivalent of 1,000 kg mineral N ha^{-1} (Cuttle *et al.*, 2001). This N is readily leachable because it exceeds grasslands' need for N; an intensively managed grassland requires approximately 350 kg mineral N ha^{-1} $year^{-1}$ (Humphreys *et al.*, 2002).

Tyson *et al.* (1997) found that higher SR is associated with greater amount of N lost as NO_3 due to increased amount of urine deposited on grasslands. The same author also showed that the use of ammonium nitrate fertiliser is associated with high losses of NO_3-N. This is because of NO_3 supply to soil, which is readily leachable when drainage events occur (Watson *et al*., 1992; Eckard *et al*., 2004). Other authors (Wachendorf *et al.*, 2006) found that incorrect timing of N application and excessive amounts of manure enrich the soil in N beyond the assimilation capacity of grass plants. As a result, the NO_3-N not taken up by grass plants passes in the soil solution and may leach to groundwater and water bodies.

Grazing management can also influence the amount of NO_3 lost through leaching. In a study conducted on perennial ryegrass and grass-clover swards in Jutland, continuous grazing resulted in leaching losses of 37 to 44 kg NO_3 ha^{-1} for both sward types compared with 25 and 26 kg NO_3 ha^{-1} respectively under continuously cut grass (Eriksen *et al.*, 2004). This was because continuous grazing allowed localised deposition of high rates of N in urine and dung resulting in increased N leachate.

2.5.2.2. Ammonia volatilisation

The NH_3 emissions occur from the soil surface if the concentration of NH_3 at the soil surface is higher than in the ambient air above the soil. This happens when the soil surface has an alkaline pH and is highly concentrated in ammonium (NH_4^+) after organic manure or N fertilisers application, for example; when the pH is alkaline the hydroxide (OH^-) ion in the soil abstracts a proton (H^+) from NH_4^+ and generates NH_3. Under dry conditions, NH_3 escapes into the atmosphere (Huijsmans, 2003). There, NH_3 is mixed into the clouds and then changed back to NH_4 via reactions with the H^+ from the cloud droplets (Asman *et al.*, 1998). This NH_4 can return to the soil surface as wet deposition through rain drops or as gaseous and particulate forms (Janzen *et al.*, 2003). Wet deposition is known as a contributor to contamination and eutrophication of natural habitats as well as to soil acidification (Marks *et al.*, 1999; Watson and Foy, 2001).

In dairy farming, important sources for NH_3 volatilisation are deposition of urine by grazing animals and mineral fertiliser N application (Watson, 2001). This is because urea in either urine or mineral fertiliser N is hydrolysed by the bacterial enzyme urease within a few hours of application resulting in the production of NH_4 (Whitehead, 1995). This NH_4 is further converted to NH_3 in the soil solution (Ryan and Fanning, 1995), which evaporates when dry conditions occur. In Ireland, this may happen between May and July, when evapo-transpiration exceeds rainfall (Humphreys *et al.*, 2004).

Manure is also associated with NH_3 emissions when it is exposed to air (Asman *et al.*, 1998). The NH_3 emissions are directly related to the equilibrium between NH_4 and NH_3 in the manure. This equilibrium is influenced by the pH of the manure. The higher the pH, the higher the NH_3 proportion in the manure (Huijsmans, 2003) and the greater the potential for NH_3 emissions. When the equilibrium is disturbed by volatilisation of NH_3, NH_4 converts to NH_3 until a new equilibrium is established (Kroodsma *et al.*, 1993).

Significant NH_3 emissions occur during animal housing, manure storage and after manure spreading on land (Leach *et al.*, 2004; Webb *et al*, 2005). On European dairy farms, the animals are commonly housed during winter in naturally ventilated slatted floor houses, the manure mixed with urine running between the slats into a pit (Kroodsma *et al.*, 1993). A more common term is "slurry", which is known as the mixture of manure, urine and dirty water resulting from washing the floors or rainwater in case of uncovered storage (European Communities, 2010a). During housing of animals, an important factor influencing NH_3 emissions is the duration of slurry exposure to air (Gilhespy *et al.*, 2006). Therefore, fast removal of slurry from the houses to storage is recommended (Jarvis and Aarts, 2000). The slurry removed from livestock houses is stored either in concrete, steel or wooden tanks or in lagoons. The larger surface area to volume ratio of the lagoons compared with the tanks enables greater potential for NH_3 emissions (Webb *et al.*, 2005). In the case of tanks, using coverage or not makes a considerable difference in terms of NH_3 emissions. Thus, 100 % of the mineral N contained in the slurry can be lost through NH_3 volatilisation from uncovered tanks compared with 1 % when using a lid and 60 % when using straw to cover the tank (Sommer *et al.*, 2003). The different rates of NH_3 emissions depend on the extent to which the different covers allow contact between the slurry and the ambient air.

Slurry spreading techniques have different impacts on NH_3 volatilisation depending on the exposure of slurry to air. More precisely, in the event of surface application, the slurry is applied mostly on top of the grass, therefore having reduced contact with the soil and being largely exposed to air. Comparatively, band application and injection of slurry into the soil reduce the contact with ambient air and increase contact with the soil, reducing NH_3 emissions by 74 and 92 %, respectively (Huijsmans, 2003). When slurry is applied to land, its DM content determines its infiltration rate into soil and therefore on NH_3 emissions. More exactly, thicker slurry, with high DM content, infiltrates slowly in the soil especially under dry conditions when it cakes on the soil surface, leading to high NH_3 emissions through exposure to air (Hennessy *et al*., 2009). Sommer and Olesen (1991) found that NH_3 volatilisation increased with higher DM content in slurry. The NH_3 emissions ranged from 19 to 100 % of total N in slurry being directly linked to slurry DM content varying from 0.9 to 16 %.

2.5.2.3. Denitrification

Denitrification is the process whereby firstly the NH_4 coming from mineralised plant residue, slurry or mineral fertiliser N applied to land is nitrified by the soil aerobic bacteria (*Nitrosomonas europea)* into NO_3. Under anaerobic conditions, in anaerobic soil pockets, sub-soil horizons or anaerobic zones such as wetlands, for example, this NO_3 can be reduced to nitrite (NO_2) by anaerobic soil bacteria (*Nitrobacter winogradskyi)* (Ryan and Fanning, 1995). This NO_2 can be reduced to NO (nitric oxide), nitrous oxide (N_2O) and dinitrogen (N_2), the final product of complete denitrification, by soil anaerobic bacteria (Pain, 2000). Even if N_2 is environmentally benign, N_2O is a powerful greenhouse gas (USEPA, 2002) contributing significantly to climate change and ozone depletion (Watson and Foy, 2001). The reactions illustrating the two processes are as follows:

$NH_4^+ \rightarrow$	$NO_3^- \rightarrow$	$NO_2^- \rightarrow$	$NO \rightarrow$	$N_2O \rightarrow$	N_2
ammonium	nitrate	nitrite	nitric oxide gas	nitrous oxide gas	dinitrogen gas

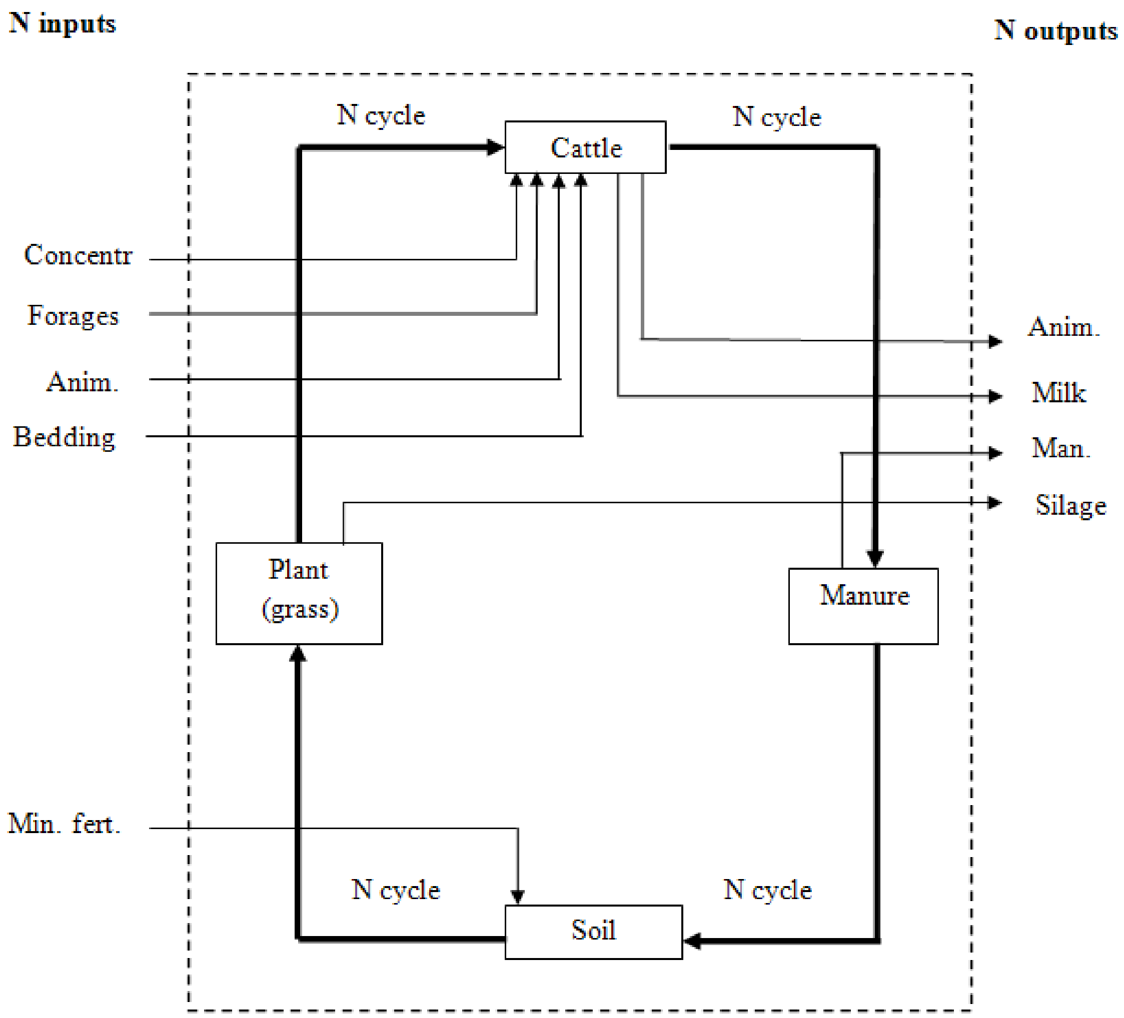
(Source: De Klein *et al.*, 2001)

Denitrification is associated with water saturated soil and subsoil conditions and high contents of organic matter (Fraters *et al.*, 2002). These conditions facilitate growth of denitrifying anaerobic soil bacteria reliant on the C from the organic matter as a source of energy (Aulakh *et al.*, 1992; Ryan and Fanning, 1995; Watson, 2001). On livestock farms, the grazing animals tread and compact the soil, contributing to generation of anaerobic conditions in the soil (Jarvis, 2000), which facilitates occurrence of denitrification. Heavy textured, poorly drained soils are mostly associated with denitrification occurrence, but also drained soils, whenever soil pores become anaerobic (Jarvis, 2000). Watson *et al.* (1992) found high rates of denitrification (154 kg N ha^{-1}) as a result of high N fertilisation rates (500 kg N ha^{-1}) coinciding with low soil moisture content. This was facilitated by the high amount of NH_4 provided by calcium ammonium nitrate fertiliser, easily nitrified in the soil under aerobic conditions and then denitrified in the lower soil profile under anaerobic conditions. However, after introduction of the Nitrates Directive (European Council, 1991) such high N application rates are no longer permitted.

In Irish soils, which are predominantly poorly drained, denitrification competes with leaching process for the available NO_3 in the soil (Jordan, 1989). This means that the existing NO_3 in the upper soil profile can be either leached in the sub-soil or nitrified by aerobic bacteria and subsequently denitrified in the sub-soil. However, the generally high (between 3 and 6 %, Gardiner and Radford, 1980) content of organic C facilitating development of denitrifying bacteria makes the Irish soils more prone to N losses through denitrification than leaching (Dillon and Delaby, 2009).

2.5.3. Farm-gate nitrogen balance and use efficiency on dairy farms

A dairy farm-gate balance, as a form of nutrient budget, accounts for N inputs onto a farm (mineral fertiliser N, concentrate feeds, forage, bedding material, livestock, and manure) and N outputs (sold milk, livestock, crops, and manure) off a farm (Fig. 2.2.). These data are obtained through direct measurements on farms or, more often, from farm records (Goulding *et al.*, 2008). A farm-gate balance is calculated as the difference between the N inputs and N outputs (Humphreys *et al*., 2003) via farm-gate. The result of the balance can be deficit (N exports > N imports) or surplus (N imports > N exports). In the case of surplus N, this may accumulate on farm, in SOM or plant biomass (Gourley *et al.*, 2010) or it may be lost to groundwater or air (Eckard *et al.*, 2007; Schröder *et al*., 2011). Therefore, N surplus is associated with excessive, inefficient N use, and negative environmental impacts, being considered an indicator of potential N losses and environmental performance (Schröder *et al*., 2003; Carpani *et al*., 2008; Powell *et al*., 2010).



(Adapted from Nevens *et al*., 2006)

Fig. 2.2. Farm-gate N balance on a dairy farm: inputs, outputs and N flows between the farm compartments (Anim.=Animals; Man.=manure; Concentr.=concentrates; Min. fert.=mineral fertiliser). Dashed line is the farm boundary.

It has been emphasised that dairy production should ideally be achieved in a sustainable manner, without impairing natural capital (soils, water, and biodiversity) (Goodland, 1997). Therefore, improved nutrient use efficiency has a significant role to play in the development of more sustainable dairy production systems. However, in dairy production systems, N is used in large quantities, between 172 and 301 kg N ha^{-1} (Groot *et al.*, 2006; Nevens *et al.*, 2006; Roberts *et al.*, 2007; Ryan *et al.*, 2011; Cherry *et al.*, 2012) but with generally low efficiency (Goulding *et al.*, 2008). In Europe, levels of N use efficiency (NUE; proportion of N imports recovered in agricultural products (Ryan *et al.*, 2012)) of between 0.17 and 0.38 have been recorded (Mounsey *et al.*, 1998; Groot *et al.*, 2006; Nevens *et al.*, 2006; Raison *et al.*, 2006; Roberts *et al.*, 2007; Treacy *et al.*, 2008; Cherry *et al.*, 2012; Oenema *et al.*, 2012). In grass-based dairy production systems, in particular, the numerous opportunities for N losses during N transfers between soil, pasture, animals, and manure components (Ledgard, 2001), as outlined in section 2.5.1, have limiting effects on N use efficiency.

In this context, farm-gate N balances are a useful tool for farmers, scientists and policy-makers to: (i) understand N flows and identify potential N losses (Watson and Atkinson 1999); (ii) understand factors affecting, and develop strategies to control, potential N losses (Gourley *et al.*, 2007; Beukes *et al.*, 2012); and (iii) increase farmers' awareness of environmental regulations on farms and implementation of these regulations to control N losses to the environment (Oenema *et al.*, 2003; Carpani *et al.*, 2008).

In the EU (European Union), the Nitrates Directive (91/676/EEC) (European Council, 1991) has established guidelines in relation to farming practices to reduce NO_3 leaching that are implemented in each member state through a National Action Programme (NAP). In Ireland, these are legislated as the GAP Regulations (European Communities, 2010a), first passed in 2006. Under the Regulations, farms are limited to a SR, of 170 kg organic N ha^{-1}, equivalent to 2 LU ha^{-1}, or 2 dairy cows ha^{-1}. The Regulations also establish the quantity of available N that can be applied to grass and other crops (depending on factors such as SR or crop type), the volume of slurry and slurry storage required (depending on factors such as local rainfall and stock type and number) and closed periods in winter months during which spreading of organic and inorganic fertilisers is restricted (depending on location in the country), as well as other measures on farm yard and field management aimed at minimising N losses to water. Farmers can apply for derogation to stock at up to 250 kg organic N ha^{-1} (2.9 LU ha^{-1}), subject to more stringent requirements, and this derogation is principally taken up by the more intensive dairy farms (European Communities, 2010).

Although explicitly aimed at decreasing N losses to water, these Regulations might be expected to have improved NUE on farms, as most of the measures aim to decrease losses by increasing retention of N within the production systems. However, most of the existing data on dairy farm N balances in Ireland date from the period before the implementation of the Regulations in 2006 (Mounsey *et al.*, 1998; Treacy *et al.*, 2008) and only one study (Buckley *et al*., 2013) after the implementation of the Regulations. Ryan *et al.* (2011) and Ryan *et al.* (2012) examined N balances and use efficiencies in Irish dairy production systems but these were based on modelling and experimental studies. In the European context also, there are few farm-gate N balances on grassland-based dairy farms post the implementation of the Nitrates Directive (e. g. Groot *et al.*, 2006; Nevens *et al*., 2006; Raison *et al.*, 2006; Roberts *et al*., 2007; Cherry *et al.*, 2012; Oenema *et al*., 2012).

2.5.4. Lowering nitrogen surpluses and losses

Reductions in N surpluses and losses are generally considered to contribute to increases in NUE in dairy production systems (Schröder *et al*., 2003; Oenema *et al*., 2003; Steinshamn *et al*., 2004; Ryan *et al.*, 2011; Nousiainen *et al*., 2011). A number of strategies have been identified as contributors to reductions in N surpluses and losses on farms and these are detailed in the next three sections.

2.5.4.1. Slurry management

Slurry is a valuable source of N (Powell and Wu, 1999; Chadwick *et al.*, 2000) on farms but this N is very susceptible to being lost through NH_3 volatilisation. This is because N is present in slurry as NH_4 and NH_3, the latter being highly volatile (Huijsmans, 2003). If managed to control N losses through NH_3 volatilisation, slurry can be used to replace mineral N fertilisers (Lalor, 2008).

Slurry management refers to collection from livestock houses, storage and application to land. Ammonia emissions may occur at any stage of the slurry management but mostly during livestock housing and after applying it to land (Leach *et al*., 2004; Webb *et al*., 2005).

The practices for reducing NH_3 volatilisation associated with slurry are well documented. Some of them are discussed below: slurry removing practices, timing of application, land application techniques, slurry acidification. During animal housing period, the practices regarding removal of slurry to storage influence the amount of NH_3 lost through volatilisation. For example, flushing the slates once every two hours can decrease NH_3 emissions by 70 % (Kroodsma *et al.*, 1993). Immediate scraping of the manure from the floor helps decreasing of NH_3 emissions from 61 to 37 % of mineral N, i.e. NH_4 and NH_3, in the slurry (Gilhespy *et al.*, 2006). These practices aim at decreasing the contact of slurry with ambient air because of the high volatility of NH_3.

At application to land, a number of techniques has been proposed to decrease NH_3 losses. Timing and rate of application are important in the first place. Vellinga and Hilhorst (2001) found that when applied in the second part of March, slurry N is efficiently utilised by the swards. This is because NH_3 from the slurry is rapidly taken up by the growing grass plants. Also, the cool and damp conditions in spring inhibit NH_3 volatilisation and enhance the infiltration of slurry into the soil (Humphreys *et al.*, 2004; Hennessy *et al.*, 2009). Thus, the coincidence of high plant demand for N with cool moist climatic conditions in spring contributes to decreased NH_3 emissions from slurry. In spring, in Ireland, under experimental conditions, a rate of 28,000 litres ha^{-1} of slurry is applied on grasslands. This amount of slurry supplies the equivalent of around 35 kg mineral N ha^{-1} (Humphreys *et al.*, 2004). This is close to the recommended N fertilisation rates (29 kg N ha^{-1}) for this time of the year (Humphreys, 2009).

A number of low emission slurry application methods was acknowledged as contributors to decreases in NH_3 losses (Lalor, 2008). One such example is injecting the slurry into the soil. The potential for NH_3 loss from injected slurry depends on the contact surface area between the injected slurry and the atmosphere, which is the lowest when all the slurry can be held in the slots. Compared to previous practices of injecting the slurry at 15-20 cm depth, the shallow injection, up to 7 cm depth, does not damage the grasslands (Hansen *et al.*, 2003) through plant dying-back along the injection slots (Huijsmans, 2003). Depending on the depth of the injection, the NH_3 emissions can be reduced by up to 90 % compared to surface application. However, this method is impractical on slopy and stone-containing soils (Webb *et al.*, 2005) as it is often the case in Ireland (Stevens *et al.*, 1998). Therefore, in Ireland, the trailing shoe method is considered to be the most suitable for grassland. It minimises the grass contamination observed with band spreading and splash plate application (broadcast) and contributes to reductions in NH_3 emissions (0.91 kg NH_3 m^{-3} of applied slurry) compared to the splash plate (1.26 kg NH_3 $m^{-3}$of applied slurry) (Lalor, 2008). Current agronomic advice in Ireland assumes that larger savings on fertiliser N can be made by applying slurry to grassland in the spring (February to April) period, with an increase of 25 % in N fertiliser replacement value (NFRV) of slurry (Coulter, 2004). Lalor (2008) valued the N from slurry (€ m^{-3}) at €1.15 for spring (February-April) application compared with €0.63 for summer (June-July) application, based on the potential for NH_3 emissions and N availability to plants. This difference is an additional incentive for practising spring slurry application.

Application of dilute cattle slurry is associated with more efficient use of N compared to thick slurry (Hennessy *et al.*, 2009), the former infiltrating faster than the latter. This way the N in the dilute slurry becomes more rapidly available to plants. Thicker, more viscous, slurry tends to lodge on the sward and soil surface and is therefore longer in contact with air, resulting in greater NH_3 emissions (Humphreys *et al*., 2004).

Acidification of slurry with nitric acid is a technique that has been investigated for lowering the NH_3 content (Stevens *et al.*, 1998). This is possible because the acid lowers the pH of the slurry below 5.5 which coincides with the equilibrium between NH_4 and NH_3 components. Thus, there is prevalence of NH_4, which is not volatile (Watson, 2001).

2.5.4.2. Strategic feeding

On dairy farms, concentrates are imported to supplement the feed available to animals on-farm (Van Keulen *et al*., 2000). Some of the N imported in concentrates is transformed into milk and meat protein, but much of it is excreted as urine and dung, which are susceptible to be lost through NO_3 leaching and NH_3 volatilisation as detailed in sections 2.5.2.1. and 2.5.2.2. It was found that ruminants excrete between 75 and 80 % of ingested N (Smith *et al.*, 1995; Hilhorst *et al.*, 2001; Watson and Foy, 2001). The high amounts of excreted N are due to rumen losses (Valk *et al.*, 2000) caused by insufficient synthesis of the dietary protein in the rumen (Tamminga, 1992).

An important step towards increasing the utilisation of N fed to animals would be matching the animal feed requirements with the dietary protein by, for example, combining grass with low protein forages (e.g. maize, fodder beet, spring barley) (Kuipers and Mandersloot, 1999; Jarvis and Aarts, 2000; Aarts *et al.*, 2000). This would be needed when feeding very young grass, which has high N content (3.26 % of herbage

DM), only 13 % of which being recovered into animal product (Jarvis *et al.*, 1989). As an effect of balancing low and high protein feeds, the N excreted per unit milk produced may be reduced and this is likely to contribute to decreases in N losses to the environment (Ledgard *et al.*, 1999). However, low protein supplementation with either cereal-based concentrates or maize silage may enable increases in SR and therefore in N surplus per ha (Peyraud and Delaby, 2006). Reducing the protein content in grass through reduced N fertilisation has also been suggested to lower the protein content in animals' diet (Valk and Van Vuuren, 1996).

Other authors (Eckard *et al.*, 2007; Humphreys *et al.*, 2009a) recommend lower SRs to decrease imported N amounts not used by animals and lost from dairy farms. Thus, less concentrates fed to animals will result in less N excreta and therefore decreased N losses to the environment.

2.5.4.3. Use of white clover in grassland

White clover (Trifolium repens L.) is the prevailing forage legume in temperate grasslands (Peyraud and Delaby, 2006). This legume can replace mineral fertiliser N, and therefore reduce N imports onto farms. This is because legumes have the ability to fix atmospheric N_2 due to a symbiotic relationship with *Rhizobium* bacteria found in its root nodules (Humphreys *et al.*, 2004). This is called biological N_2 fixation (BNF) and is a biotic process that makes atmospheric N available to plants (Van Dommelen and Vanderleyden, 2007). In a grass/clover sward, the N fixed by white clover becomes accessible to the grass through mineralisation of rotted clover stubble or ingestion of clover plants by livestock and deposition of N in urine and dung on swards (Watson, 2001).

In The Netherlands, it was found that each tonne of clover DM per hectare is equivalent to a N fixation rate of 54 kg ha^{-1} $year^{-1}$ (Van der Meer and Baan Hofman, 1989). In Ireland, white clover can supply between 87 and 150 kg N ha^{-1} $year^{-1}$ (Humpreys *et al.*, 2008; Hennessy *et al.*, 2009). The N fertilisation rates strongly influence the amount of atmospheric N fixed by clover, the amount of N_2 fixed by clover decreasing with increasing N fertilisation rates (Ledgard *et al.*, 1997). This is because when applying fertiliser N to a grass-clover sward, the grass plants will be more competitive for N than the clover plants (Davies, 2001). This results into deterioration of clover plants and therefore a decrease of clover content in the sward and N_2 amount fixed by clover. At

Solohead research farm in Ireland, white clover swards are however fertilised with 90 kg N ha^{-1} applied between January and May, when the growth rates of clover plants and BNF are naturally low. Starting with May, the clover plants develop and the N fertilisation is replaced by BNF (Humphreys and Lawless, 2006). This indicates the potential of clover to partially replace mineral fertiliser N on grasslands (Peyraud *et al.*, 2010).

From an environmental point of view, in comparisons between N-fertilised grass and white clover-based production systems, the latter has generally been associated with higher N use efficiency, lower N surplus per hectare, lower NO_3 and NH_3 losses, and lower N_2O emissions (Jarvis *et al.*, 1996; Schils *et al.*, 2005; Andrews *et al.*, 2007; Humphreys *et al.*, 2008; Ledgard *et al.*, 2009). However, Peyraud and Delaby (2006) found increased N excretion associated with white clover relative to ryegrass (from 20.1 to 29.8 g kg^{-1} DM intake), due to higher N content of white clover compared to ryegrass (38.7 versus 26.1 g kg^{-1} DM). Increased N excreta may lead to higher N losses through leaching and volatilisation.

The problems associated with making use of white clover mostly relate to maintenance of white clover content of swards. For example, under the Irish climatic conditions, white clover lasts for, at most, five years in the pastures managed for production of white clover. After this period, the pastures need to be over-sown, which requires special skills and supplementary costs (Humphreys *et al*., 2004). In Ireland, white clover swards are over-sown with 5 kg ha^{-1} of white clover seed mixed with mineral fertiliser with a P concentration of 0.07 g g^{-1} (Humphreys *et al*., 2008). The clover seed is broadcast onto silage stubble, after first cut silage, with a fertiliser spreader. The application is made in two runs, to ensure even broadcasting. During one run, 4 ha can be covered. The total cost of this approach is €28 per 0.4 ha (Humphreys and Lawless, 2008).

Other problems related to using white clover in grasslands are lower herbage production, in comparison with mineral N fertilised grasslands (10.81 t DM ha^{-1} versus 12.61 t DM ha^{-1}; Humpreys *et al*., 2008; Humphreys *et al*., 2009b). Therefore, some farmers can be reluctant to replace mineral N fertilisers with N supplied by clover plants (Eckard *et al*., 2007).

2.6. Phosphorus in grassland soils

Phosphorus (P) is a macronutrient highly important for both terrestrial as well as aquatic ecosystems' productivity (Antikainen *et al.*, 2005). In agricultural systems, P contributes to root and seed formation, and microbial decomposition of plant residue (Lynch and Caffrey, 1997; Haygarth and Jarvis, 1999). Also, P is an important contributor to production on grass-based systems. The role of P in grass plants is emphasised by responses of DM yields, of 7.7 kg DM kg P^{-1}, for example, achieved without mineral P fertilisation (Ryan and Finn, 1976). In the animal body, P has essential physiological functions that include energy transfer, structure of bone, teeth, and membranes, and buffering pH changes in the rumen (salivary phosphate). In most of the grains used for animals' diets, P is found as phytate. This form of P is used in larger proportion by ruminants than non-ruminants because rumen microbes produce phytase, the enzyme that hydrolyses P from phytate (Satter *et al.*, 2005). For the above reasons, it can be stated that P has no substitute in agricultural production (Cordell *et al.*, 2011).

2.6.1. Phosphorus cycle

Parfitt (1980) explained the P cycle as it typically occurs on grazed grass-based dairy farms (Fig. 2.3). The P needed for growth of grass plants is taken up from the "Available P" pool. The P in the plant biomass is consumed by grazing dairy livestock. The ingested P is exported in animal products (milk, livestock) or excreted in dung. The dung deposited on the pasture and plant residues, containing both inorganic and organic P, re-enters the soil pool, either in the "Labile organic P" (the organic P) pool or in the "Available P" (the inorganic P) pool. Alternatively, the P in the dung can be lost through surface and erosion runoff, if all the conditions are met. However, there are exchanges between the two soil P pools, because every year 70 % of the organic P decomposes to inorganic P or phosphate (PO_4). In this form, P is readily available to plant roots and micro-organisms as well. The P that is not used by plants or micro-organisms either builds up in the soil in non-accessible forms (inositol polyphosphate), or leaches down the soil profile, where it reacts with the aluminium (Al), iron (Fe) and calcium (Ca) contained by soil particles, a process known as “fixation”. The fixed P may be released under certain conditions. The mineral P applied on soil is either lost through surface and erosion runoff, if all the conditions are met, or taken up by the grass plants until feed requirements are met. All these processes are detailed in the next two sections (2.6.2. and 2.6.3.).

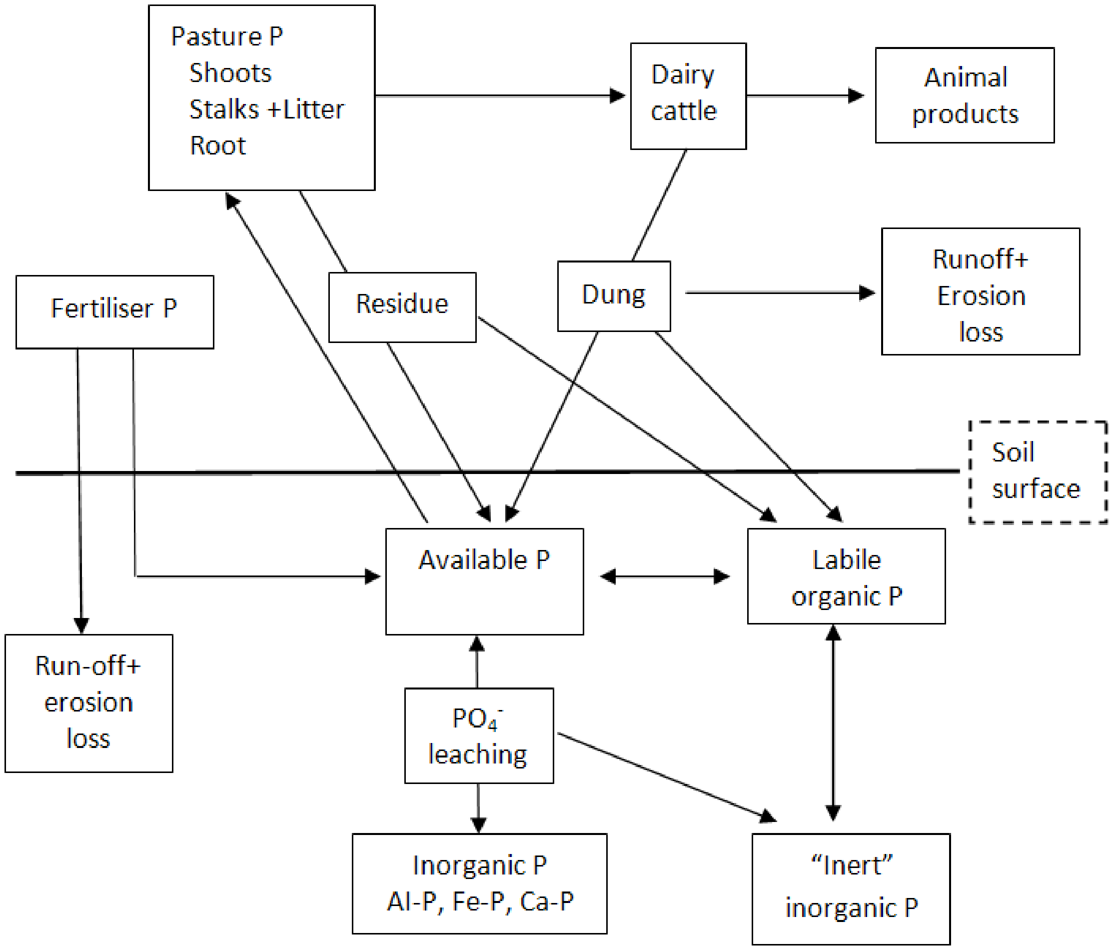


Fig. 2.3. Simplified P cycle on grasslands (adapted from Parfitt, (1980))

2.6.2. Soil Phosphorus

In the soil, P exists in inorganic and organic forms. In most agricultural soils, between 50 and 75 % of the P is inorganic. The inorganic P forms are mostly Al and Fe phosphates in acidic, non-calcareous soils and Ca phosphates in alkaline, calcareous soils. The organic P fraction includes unstable forms such as inositol phosphates, of which phytic acid is the most significant component, phospholipids, nucleic acids, and fulvic acids, more stabile forms being represented by humic acids (Sharpley and Rekolainen, 1997).

Four distinct classes of organic P compounds were detected in soil solution of grasslands in south-east Ireland through ^{31}P nuclear magnetic resonance (NMR) spectroscopy: inorganic orthophosphate (δ = 6.79 ppm, on average), orthophosphate monoesters such as inositol hexakisphosphate (δ = 6.11 ppm, on average), orthophosphate diesters such as phospholipids, ribonucleic acid (RNA) and deoxyribonucleic acid (DNA) (δ = -0.32 to 0.54 ppm), and pyrophosphates (δ = -3.26 ppm, on average) (Bourke *et al*., 2008). For plant growth and development, P is needed as orthophosphate (PO_4^{3-}) ion in the soil solution (Morgan, 1997; Sharpley and Rekolainen, 1997). The organic P can be classified in distinct fractions whether they occur in soil solution, runoff, leachate, streams or lakes. These fractions are important for understanding the fate and transport of P. The P fractions can be: (i) molybdate or dissolved or soluble reactive P (MRP), which is the form filtered through a 0.45 µm membrane; this is dominantly inorganic form of P and is mostly mobile and therefore easily released to soil solution and then transferred to waters; (ii) unreactive P (SUP), which is represented by organic forms coming from plant residues and soil organisms. These two fractions together are known as total phosphorus (TP) (Haygarth and Jarvis, 1999).

In agricultural systems, P is applied to soils in inorganic or organic forms to ensure soil P supply for plant uptake. Once applied, P is either taken up by the crop (1 to 2 % of soil total phosphorus content; Antikainen *et al*., 2005) through diffusion (i.e. movement of $H_2PO_4^-$ from higher to lower concentration of soil solution; Syers *et al*., 2008) or is adsorbed on to Al, Fe and Ca charged soil particles (Syers and Curtin, 1988). Adsorption is the conversion of inorganic P dissolved in the soil water to less soluble states and immobilisation in the soil organic fraction. This occurs as a result of exchange between the $H_2PO_4^-$ (dihydrogen phosphate) anion and hydroxyl (OH^-) ions associated with the Fe or Al hydroxyl compounds occurring as separate particles or as coatings on other soil particles, especially clay. Also, $H_2PO_4^-$ anions in soil solution may go through precipitation reactions, the nature of which varies with the pH of the soil. When soil pH is between 6 and 6.5, the P immobilisation is minimal (Heathwaite, 1997). The P immobilisation capacity directly influences the P losses. Thus, the P from sandy soils, with low P immobilisation capacity, is susceptible to be lost through subsurface runoff. The soils with high P immobilisation capacity, such as clayey soils and loams (Toor *et al.,* 2005), will still loose P in surface runoff if all the conditions are met (poor land cover, slope, weak soil structure, excessive application of P fertilisers associated with rainfall events) (Heathwaite, 1997). On these soils, P fertilisation may turn inefficient (Simpson *et al*., 2011) because most of the supplied P will not be used by plants but will accumulate in the soil at rates as high as 23.4 kg P ha^{-1} $year^{-1}$ (Smith *et al*., 2003).

However, the adsorption process is reversible, meaning that soil P can be desorbed or released from Al and Fe hydroxyl compounds when soil pH is neutral (equal to 7) (Kiely *et al*., 2007), when soil P immobilisation capacity is exceeded (Lawrie *et al*., 2004) or when the P concentration in the soil solution decreases through plant uptake (Syers *et al*., 2008). The P desorption is controlled by the P-buffer capacity of soils or the rate at which P in the soil solution is replenished. The higher the P-buffer capacity is, the faster is the P desorption (Syers *et al*., 2008).

The P fertiliser management on soils with high P immobilisation capacity, which are deficient in available plant P, should include a phase in which soil P fertility is increased, followed by a soil fertility maintenance phase in which soil P levels are held within a target range (Simpson *et al*., 2011). This would keep the soil P levels within the limits corresponding to plant P requirement. For grassland, under Irish conditions, the maximum recommended amount of P is 19 kg ha^{-1} $year^{-1}$ when the soil P level is between 5.1 and 8 mg P $litre^{-1}$ (European Communities, 2010a). However, in areas with intensive livestock farms, manure and mineral fertilisers are abundantly applied to grassland resulting into soil P levels higher than 8 mg P $litre^{-1}$. Under experimental conditions, it was found that from 331 mg P per kg soil coming from mineral and organic fertilisers, ryegrass plants take up 4 mg P kg^{-1} soil (Oberson *et al*., 2010). In this situation, the soil P levels are of environmental concern due to the high potential of enriching runoff with P potentially contributing to the eutrophication of water bodies (Sharpley and Rekolainen, 1997). Hence, there is continuous need to estimate the soil P content.

Soil P status can be assessed by making use of soil extraction methods. Through these methods the amount of soil P available for crop uptake can be evaluated up to 20 cm depth (Ekholm *et al.,* 2005) and the fertiliser P demand estimated. The standard method in Ireland, Morgan STP, uses a sodium acetate extractant buffered to pH 4.8 (0.72 *M* CH_3COONa + 0.52 *M* CH_3COOH, pH 4.8) (Styles *et al.,* 2006; Schulte and Herlihy, 2007). The extractant is designed to take from the soil an amount of P considered sufficient for the plant growth during one season (Byrne, 1979). It was however found that chemical extractants such as Bray-P, Olsen-P, Anionic exchange membrane-P did not provide accurate information on how much of the applied P to soils is available to plants and how much is fixed by soil in non-available forms. Therefore, sound fertiliser recommendation cannot fully rely on soil extraction indices. The actual P requirement in a P deficient soil is closely related to the soil content of amorphous Fe, which affects both the amount and the strength of P retention by the soil (Quintero *et al.*, 2003).

Also, for the interpretation of soil P status data, increased attention should be paid to soil sampling. Due to the fact that soil P content decreases from top to bottom of the soil profile in case of permanent pastures (Haygarth and Jarvis, 1999), the depth of soil sampling is of major importance.

The sites with high soil P status and therefore high potential for P losses from agricultural soils can be identified through an approach called “phosphorus index” (Haygarth and Jarvis, 1999). Since 2006, the soil P index system in the Republic of Ireland refers to separate P indices for soils under grassland and other crops (Table 2.2). Before 2006, the same index system was used for the soil lying under grassland and other crops. The new index system involved the lowering of the upper limit for index 2 from 6 to 5 mg P litre^{-1}, and the upper limit for index 3 from 10 to 8 mg P litre^{-1}. The aim was to reduce P losses from grassland while maintaining current levels of agricultural output (Treacy, 2008). The index system for other crops has remained unchanged. The change in soil P status is assessed every five years in Ireland (European Communities, 2010a).

Table 2.2. Soil P indices (Morgan’s Extractant) (European Communities, 2010a)

	Soil phosphorus concentrations (mg P litre^{-1})	
Index	Grassland	Other crops
1	0.0 – 3.0	0.0 – 3.0
2	3.1 – 5.0	3.1 – 6.0
3	5.1 – 8.0	6.1 – 10.0
4	> 8.0	> 10.0

Index 1 and 2 are associated with soils deficient in P and demanding a build-up of soil P. The target index is index 3 at which the soil can supply P in amounts meeting crop demand without having negative impacts on the environment. Soils within index 4, with high P status, are associated with no response to applied organic or mineral P. As high soil P levels result into high P losses (Kurz *et al.*, 2005; Styles *et al.*, 2006), soils in index 4 show the greatest risk for water quality. That is why it is recommended to let the P level of these soils decrease over time. According to Culleton *et al.* (1999), a deficit of between 20 and 40 kg P ha^{-1} between inputs and outputs of P is required for 1 mg $litre^{-1}$ decrease in Morgan STP to occur. Tunney *et al.* (2010) found, however, that a decrease of 30 kg applied P per ha decreased the STP level by 8 mg $litre^{-1}$.

Soil test P increases with P application rates (Morton *et al.*, 1999). Therefore, if efficient use of P fertilisers is the target, they should not be applied to soils where there is sufficient readily-plant-available P because there will not be any increase in crop yield and the expenditure on P fertilisers becomes unnecessary; P is most efficiently used when the amount applied replaces that removed in the harvested crop (Syers *et al.*, 2008). The readily-plant-available P comes from soil P reserves or residual P sourcing from previous P applications (Syers *et al.*, 2008) and it should be carefully considered when deciding about rates of P fertilisers.

Historically, Irish soils were low in P (1 mg $litre^{-1}$ in early 1950s), much of the P present in the topsoils on intensive farms being added directly in mineral fertiliser and indirectly in purchased feedstuffs starting in the 1960s (Tunney, 1990). However, the annual use of mineral fertiliser P in Ireland has been decreasing over 30 % from 1995 to 2001 due to research advice and consequent changes in farming practice (Power *et al.*, 2005; Kiely *et al.*, 2007). Also, Morgan STP decreased from 13.02 mg P $litre^{-1}$ between 1966 and 1970 (Ryan and Finn, 1976) to 6.7 mg P $litre^{-1}$ in 2003 (Bourke *et al.*, 2008) and from 7.3 to 4.0 mg P $litre^{-1}$ between 2007 and 2011 (Wall *et al.*, 2012) on Irish grassland soils.

2.6.3. Phosphorus loss pathways

Over time, the soil P build-up in agricultural land was encouraged, due to general thinking that mineral P is assimilated in organic forms in the soil and therefore is associated with negligible P losses. However, it was found that where inorganic soil P exceeds critical P levels, ensuring 90 % of maximum achievable yield (Quintero *et al.*, 2003), it increasingly accumulates in the soil with increased P application (Simpson *et* *al.*, 2011). Further, the P accumulated in the soil is susceptible to losses to water where it might cause eutrophication (Clenaghan *et al.*, 2005) or increased primary aquatic production (i.e. proliferation of algae), and therefore a shortage of oxygen and a shift of species within the aquatic ecosystems (Pieterse *et al.*, 2003; Ekholm *et al.*, 2005). Algal excessive development and low water transparency indicate eutrophication occurrence and the cyanobacterial algae prevalence among the algal species poses problems of toxins, odour and taste for drinking water (Watson and Foy, 2001). In Ireland, P is the major limiting nutrient in surface waters and increased additions may result in algal blooming (McGarrigle, 2009). Hence, following the Phosphorus Regulations implemented in 2001 by Environmental Protection Agency (EPA), target values were set for the eutrophication status of lakes and rivers, measured through the TP concentration (Table 2.3). The highest trophic status listed is hypertrophic, representing an extreme state of P enrichment of water bodies (Clenaghan *et al.*, 2005). The target of the European Union’s Water Framework Directive is to restore all surface water to good ecologic and chemical status by 2015 (Watson *et al.*, 2007).

Table 2.3. Phosphorus Regulations target values of trophic status for Irish rivers and lakes

Trophic status	Total phosphorus concentration (µg P litre^{-1})
Ultra-oligotrophic	<5
Oligotrophic	>5<10
Mesotrophic	>10<20
Eutrophic	>20<50

(Source: Clenaghan *et al.*, 2005)

Most eutrophication caused by P loss from enriched soils was reported in lakes draining catchments where agriculture is prevailing, concentrations as low as 0.02 to 0.035 mg P litre^{-1} being enough to prompt eutrophication (Brookes *et al.*, 1997). Loss rates of 2 to 3 kg P ha^{-1} year^{-1} have been recorded for Irish grasslands (Kurz *et al.*, 2005).

The main drivers of eutrophication are leaching and run-off of P from agricultural land (Nousiainen *et al.*, 2011). Losses of P from land to water can occur as: (i) water-soluble

(<45 µ) and particulate P (>45 µ) in surface runoff or overland flow, referring to P carried by rainwater which flows over land surfaces to streams or rivers; (ii) water-soluble and particulate P in subsurface runoff or leachate, referring to P transported by water which enters the soil profile and moves through the soil to streams or rivers without reaching the main water-table; (iii) water-soluble and particulate P in the groundwater, referring to P picked up by water that passes to the water-table and which is subsequently discharged to streams, rivers or lakes as seepage (Morgan, 1997). For clarification, particulate P consists of P sorbed by soil particles and organic matter eroded during surface flows. This represents the major proportion of transported P from cultivated land (Kurz *et al.*, 2005). When comparing grassland with arable land, the losses of water-soluble P in surface runoff are higher from grassland due to less P used by the grass plants and therefore accumulated in the soil. Conversely, the loss of particulate P in surface runoff is higher from arable land because of increased level of erosion (Ekholm *et al.*, 2005). The occurrence of P losses due to erosion can be translated as increased need for P input, in order to maintain the soil fertility (Schröder *et al.*, 2011).

2.6.3.1. Surface runoff

Surface runoff or overland flow is the transport of eroded material containing particulate P or the P adsorbed on to organic-rich clay soil fractions. The latter is inorganic DRP and it can reach values between 111 and 1,162 g P ha^{-1} on Irish grasslands depending on the soil P content, soil compaction by grazing animals, manure deposition and intensity of rainfall events (Kurz *et al.*, 2005).

The potential of a field for P transport in surface runoff can be assessed by taking into consideration infiltration capacity, vegetation roughness coefficient, slope, initial concentration of P in the soil, together with storm duration (Heathwaite, 1997). In Ireland, 25 % of agricultural land is lying on gleys, which are known as water saturated soils and hence prone to overland flow due to their low rate of infiltration (Finch and Gardiner, 1993). Water-saturated soils expand during heavy rainfall and contract when rain stops. These drying and wetting cycles contribute to transport of potentially mobile P (PMP) from soil when overland flow occurs (Kiely *et al.*, 2007). Loss of P from Irish agricultural land through surface runoff occurs in two phases. The first phase takes place annually with the overland flow beginning in September-October after little or no overland flow during the summer. The second phase is erratic, being associated with applications of manures or fertilisers to the land followed by rainfall events (Kiely *et al.*, 2007).

What could be called high-risk land is usually located next to a stream and has low infiltration capacity as a result of, for example, trampling by cattle or compaction by farm machines (Heathwaite, 1997). Other authors (Schulte *et al*., 2009) consider as high risk areas those where pathway factors coincide with pressure factors. The main pathway factors include poor soil drainage and short distance from fields to water courses, whereas the main pressure factors are low P immobilisation capacity, elevated STP levels (i.e. Morgan index 4) and excessive P inputs.

On pastures there is increased risk of P loss through surface runoff, because poaching (i.e. soil compaction and break-up due to animal trampling) decreases infiltration and thus increases overland flows and associated P transfer in case of rain fall events. This happens because soil becomes bare and weakly structured and therefore prone to erosion (Haygarth and Jarvis, 1999). In the situation of overland flow occurrence combined with high P application rates (80 kg P ha^{-1}), the amount of DRP per litre can reach values up to 21.5 mg on grazed pastures (Watson *et al.*, 2007). However, P concentrations in overland flow are normally not equal to P levels found in rivers and streams. This is because overland flow only occurs erratically and hence it does not represent a continuous P input to the water bodies (Kurz *et al*., 2005).

2.6.3.2. Phosphorus leaching

Subsurface runoff or leaching is influenced mostly by soil physical properties such as texture, structure, porosity. The soil physical properties help defining the rate of water movement through soil pores. Thus, the macropores, existing in structured soils or developing through cracking during dry periods, facilitate rapid bypass flow through the soil. This reduces the contact time between soil and percolating water. The bypass or macropore flow is the same as surface runoff in terms of P transport.

In the case of grasslands, the P transport takes place mostly as subsurface runoff due to reduced surface erosion. This is because of good land cover with a high vegetation roughness coefficient. The P losses through leaching take place mostly from sandy-textured, organic, and heavy-textured clayey soils. This is because the sandy and organic soils usually show little depth of soil profile to the water-table which is hence easily enriched with P. In clayey soils, even if having subsurface horizon with a large capacity to absorb P, these top layers become easily P enriched. This is due the bypass flow down cracks leading to rapid movement of P to the drains (Heathwaite, 1997). As opposed to these types of soils, the ones with high P fixing capacity such as silt loams are less prone to the P losses in soil solution and runoff (Toor *et al.*, 2005).

Manure deposition is seen as important source of P leachate because mineral P from water soluble fraction of manure (Huhtanen *et al.,* 2011) represents a direct source of PMP, i.e. P vulnerable to transfer to water (Haygarth and Jarvis, 1999). Further, the P from water soluble fraction of manure deposited on soil surface, being highly soluble (Huhtanen *et al.*, 2011), can be leached and dissolved in soil solution contributing to soil P accumulation and eventually saturation (Heathwaite, 1997).

2.6.4. Farm-gate phosphorus balance and use efficiency on dairy farms

Similar to dairy farm-gate N balance, farm-gate P balances typically account for P inputs in mineral P fertilisers, concentrate feeds, forages, bedding materials, livestock and manure, and P outputs in sold milk, livestock, crops, and manure passing the farm-gate (Aarts, 2003; Spears *et al.,* 2003b; Nielsen and Kristensen, 2005). Some authors consider these as partial P balances because they do not account for P losses (Weaver and Wong, 2011). In whole-farm P balances, there are included P inputs such as precipitation (Weaver and Wong, 2011), atmospheric deposition (Van Keulen *et al.*, 2000; Gourley *et al.*, 2012) and irrigation water (Weaver and Wong, 2011; Gourley *et al.*, 2012) and outputs such as P losses from manure storage (Van Keulen *et al.*, 2000).

Phosphorus imports, in the form of concentrate feeds and fertilisers, are key drivers of increased herbage yields and saleable milk export on most dairy farms (Aarts, 2003; Spears *et al.,* 2003; Gourley *et al.*, 2012). However, P imports typically exceed P exports in milk and livestock exported off the farms (Van Keulen *et al.,* 2000). This imbalance results in surplus P that is either accumulated in soil or lost from the dairy farms (Arriaga *et al*., 2009; Gourley *et al*., 2010).

Farm-gate P surplus is commonly used as an environmental indicator for the risk of phosphorus losses to the environment (Swensson, 2003; Huhtanen *et al.,* 2011; Weaver and Wong, 2011). Even if surplus P does not predict the actual losses and loss pathways, it is a long-term risk indicator of P losses (Jarvis and Aarts, 2000). However, unlike N surpluses which are seen, necessarily, as an economic waste and potential environmental problem, P surpluses may be necessary, for a period of time, on farms where an increase in soil P content is required to achieve agronomic optimal soil P (Culleton *et al.,* 1999) without posing a risk to the environment, if managed correctly. Surplus P potentially accumulates in the soil (Gourley *et al.,* 2010), building soil fertility, or is lost in eroded material containing particulate P or P adsorbed on to organic-rich clay soil fractions (Kurz *et al*., 2005) or in soluble forms through leaching (Heathwaite, 1997) or runoff.

Grass-based farms can be sources of diffuse P losses (Kiely *et al.,* 2007), because, by fertilising grassland with mineral and organic fertilisers, high concentrations of PMP are placed at or near the soil surface, where it may be susceptible to mobilisation and transport to water bodies (Herlihy *et al*., 2004). These P losses can have negative environmental impacts such as eutrophication of surface waters (Clenaghan *et al.,* 2005), and pollution of groundwater aquifers (Heathwaite, 1997).

Human intervention in the global phosphorus cycle has mobilised nearly half a billion tonnes of the element from phosphate rock into the hydrosphere over the past half century. The resultant water pollution concerns have been the main driver for sustainable phosphorus use (Cordell *et al.,* 2011). Also, the on-going debate over P supply and demand together with the concern for water quality affected by P lost from agricultural land support the need to ensure that P is used efficiently on farms (Pieterse *et al*., 2003; Syers *et al.,* 2008; Weaver and Wong, 2011; Simpson *et al.,* 2011). Therefore, in the EU, the Water Framework Directive (WFD) (2000/60/EC) was introduced with the objective of protecting and improving groundwater and surface water bodies' quality. In Ireland, the WFD was first implemented as the Water Policy Regulations (European Communities, 2003), in 2003. To ensure water quality, these regulations established a concentration limit of 0.03 mg Molybdate Reactive Phosphorus (MRP) litre^{-1} or 35 µg PO_4 litre^{-1} (European Communities, 2009). Additionally, the Nitrates Directive (91/676/EEC) (European Council, 1991) has established the quantity of available P that can be applied to grass and other crops (depending on factors such as SR, soil test P (STP) and crop type) (Table 2.4.).

Table 2.4. Annual maximum fertilisation rates of phosphorus on grassland (European Communities, 2010a)

Grassland stocking rate (kg organic N ha^{-1})	Phosphorus index			
	1	2	3	4
	Available phosphorus (organic and mineral fertilisers) (kg P ha^{-1})			
≤ 130	35	25	15	0
131-170	39	29	19	0
171-210	44	34	24	0
211-250	49	39	29	0
>250	49	39	29	0

However, the P use efficiency (PUE; proportion of P imports recovered in agricultural exports (Aarts, 2003) in dairy production systems is highly variable. For example, in Europe, PUE values of between 0.37 and 0.85 have been recorded (Mounsey *et al.*, 1998; Van Keulen *et al*., 2000; Steinshamn *et al*., 2004; Nielsen and Kristensen, 2005; Raison *et al.*, 2006; Huhtanen *et al*., 2011).

This is because in grass-based dairy production systems, there is a number of factors affecting PUE, such as soil P-sorption capacity in relation to soil P inputs, uneven dispersal of excreta leading to uneven soil P content (in grazing enterprises), the ability of grass plants to convert P from applied mineral fertiliser and manure into biomass in herbage, utilisation by animals of grass herbage grown and the biological potential of cows to convert P from concentrate feeds and herbage into milk (Gourley *et al.*, 2010). More effective use of P imports in concentrate feeds and fertiliser P, and soil P resources, can potentially contribute to decreased imports and increased PUE (Nielsen and Kristensen, 2005; Huhtanen *et al.*, 2011).

The GAP measures are intended to increase PUE and retention of P within the production systems and minimise losses from farms to water. However, most of the existing data on dairy farm P balances in Ireland date from the period before the implementation of the Regulations in 2006 (Mounsey *et al.*, 1998; Treacy, 2008). There is only one study on farm-gate P balance on Irish dairy production systems after the implementation of GAP regulations (Buckley *et al*., 2013). In the European context also, there are few farm-gate P balances on grassland-based dairy farms (e. g. Van

Keulen *et al*., 2000; Aarts, 2003; Swensson, 2003; Nielsen and Kristensen, 2005; Raison *et al.,* 2006). Steinshamn *et al*. (2004) and Huhtanen *et al.* (2011) examined P balances and use efficiencies in dairy production systems but these were based on modelling and experimental studies.

2.6.5. Lowering phosphorus surpluses and losses

Agriculture was found to be the major contributor (70 %) to the total phosphorus in the Irish surface waters (Toner *et al.*, 2005). Therefore, there is a need to reduce the P losses from farms to surface waters. This can be achieved by finding equilibrium between P imports in purchased feeds and fertilisers with P exports in agricultural products at the same time with maintaining soil P content at satisfactory levels for crop requirements (Simpson *et al*., 2011). At farm level, different groups of practices were identified aiming at different sources of P losses, such as source management, manure management, and transport management. (Sharpley and Rekolainen, 1997).

2.6.5.1. Source management

It is accepted that eutrophication is caused by high soil P levels, which are partially the result of P imports in concentrate feeds and fertilisers on farms exceeding P exports in agricultural products. Therefore, reductions in these two sources of P imports on farms can contribute to decreases in soil P levels (Lynch and Caffrey, 1997; Van Keulen *et al*., 2000). Among the strategies aiming at decreasing these P imports, are the reduced use of P fertilisers, use of low P content feeds and use of phytates instead of purchased concentrate feeds.

Fertilising grassland with mineral fertilisers and organic manures can increase P losses by increasing the concentrations of PMP on soil surface, from where it can be mobilised by water (Kiely *et al*., 2007). Mineral P fertilisers have higher effect on increasing soil P content than manure in the short term (Toor *et al*., 2005), and this reads as an increased need to reduce the use of the former compared to the latter on farms. Jouany *et al*. (2004) and Huhtanen *et al*. (2011) found that reducing fertiliser P imports on farms contributed to decreases in P surplus. This most likely contributes to reduced soil P content. However, farmers are reluctant in reducing fertiliser P use, which is commonly associated with lower soil fertility. That is why there are promoted practices such as "equilibrium fertilisation", meaning that no more P is applied than taken up by the crop (Van Keulen *et al*., 2000). However, there are two main factors that should be considered when making decisions about mineral fertiliser P imports and application. Pastures can take up to 40 kg P ha^{-1} during one season (Ryan and Finn, 1976; Power *et al.*, 2005). This means that it should be ensured that soil P content, coming from organic or inorganic sources, meets this requirement to obtain economic herbage yields. Also, the herbage fed to animals should have a P concentration satisfying the dietary requirements of 3.5 mg P kg DM^{-1} (Haygarth *et al.*, 1998). Therefore, the P application rates should ensure high enough P content of herbage to meet the above feeding requirement to maintain animal performance.

Also, the intensification of dairy production has resulted in large numbers of animals producing more manure than can be used by pastures. Much of the P in such manures is the inorganic P from animal feeds (Syers *et al*., 2008). Knowing that ruminants excrete more than 70 per cent of the ingested P as manure (Watson and Foy, 2001), any additional inorganic dietary P in excess to animals' need only results in excessive excretion of P and water soluble P in faeces (Powell and Satter, 2001) which is susceptible to P losses. Therefore, reduction of inorganic dietary P as to match animals' requirements (between 3.2 and 4.2 g P kg^{-1} concentrate; Steinshamn *et al*., 2004) can reduce P excretion in manures (Satter *et al*., 2005). Inorganic dietary P can be reduced also by adjusting it to the actual levels of production of the individual animals, with the older ones receiving less P (Schröder *et al.*, 2011). Arriaga *et al*. (2009) found that a decrease in the amount of dietary P by 17 % leads to a decrease in P excreta by 35 %. The decreased P excreta may contribute to decreases in P surplus (Powell and Satter, 2001) and therefore decreases in the P accumulated in soils (Toor *et al*., 2005). It was also found that the P content of diets greatly influences the P concentration of runoff flow from manure-amended fields. For example, when manure derived from dairy cows fed a high (49 % of total DM intake) and low (31 % of total DM intake) P diet were applied at equal amounts, difference in the amount of P lost through runoff between plots amended with manure from cows fed high P diet was 8 to 10 times greater than from plots amended with manure from cows fed low P diet (Satter *et al*., 2005). However, reducing P manure through low P rations depends very much on the availability and, especially, prices of low P dietary ingredients. For example, by-products from ethanol production can be available at a low price (Huhtanen *et al.,*

2011) but they have often a high P concentration (corn distillers grains, 71 % P of total starch concentration; Eghball, 2005).

Use of phytates originating from the seeds of home-grown concentrates is recommended compared with purchased P feeds because in ruminants they are completely hydrolysed due to occurrence of phytase enzyme in the rumen (Watson and Foy, 2001). This reads as complete utilisation of P from phytates by grazing livestock and minimal P excreta susceptible to losses.

2.6.5.2. Manure management

Considering that approximately 75 % of ingested P in concentrated feeds and forages is excreted, P management on agricultural land via manure application is one important step towards mitigating P losses to surface waters (Satter *et al.*, 2005).

Timing and rate of manure application on land should be carefully considered in order to prevent P losses through leaching or runoff. Farmers usually find time for manure application during autumn and winter. As plant growth is minimal during that time of the year, the plant uptake is small, and hence the potential for P loss with rainwater is high (Sharpley and Rekolainen, 1997). In Ireland, the GAP Regulations established closed periods during which spreading of organic fertilisers is restricted (between October and January, depending on the location in the country) (European Communities, 2010a) and the application of manure on grasslands is linked to SR (Table 2.4).

On permanent grasslands, the continuous and uneven deposition of manure often exceeds pasture requirements for P. This can lead to P accumulation in soil (Satter *et al*., 2005; Kiely *et al.*, 2007; Simpson *et al*., 2011), which is further susceptible to leaching. One way of reducing manure deposition on grasslands is restricted grazing time. For example, restricted grazing time to about one third has led to a decrease down to 17 % of the total P excreta at De Marke experimental farm in Netherlands (Van Keulen *et al.*, 2000). Uneven distribution of dung spots is associated with higher STP levels. The STP levels under dung-pats can be three-to four-fold compared to areas with no dung-pats (Kiely *et al.,* 2007). The rotational grazing and night time confinement to allow collection and redistribution of manure on different paddocks could be adopted as strategies to relief P load on grazed grassland (Lawrie *et al.,* 2004). However, manure transportation for spreading it on different paddocks incurs supplementary costs (Satter *et al.*, 2005) often having an impact on decisions relating to manure spreading practices.

2.6.5.3. Phosphorus transport management

Where livestock grazes riparian land, there is no buffer between the land and the stream. This means that little transformation or trapping of P exported from the land is possible before it enters the stream. The direct deposition of dung and urine into watercourses has been shown to be a major source of P in suspended sediment (Richards *et al.*, 2009). Therefore, riparian land needs to be carefully managed to control P losses. This can be done through introduction of buffer zones of various widths, as necessary, to intercept P rich runoff, the role of roughness coefficient of the vegetation being critical (Heathwaite, 1997). In Ireland, the width of the buffer zones varies between 5 and 200 m, depending on the type and the use of the water source (European Communities, 2010a). As grasslands are also associated with considerable amounts of P losses in overland flow, strategies to improve soil structure and hence water infiltration should also be taken into consideration (Watson *et al.*, 2007). However, improved water infiltration may lead to increased P leaching.

In Ireland, the buffer zones are not widely adopted due to various reasons. Within Agricultural Catchments Programme, the cost of implementation of buffer zones was estimated at €1.51 per linear metre. Fifty-three per cent of 247 interviewed farmers with regards to their willingness to supply area for buffer zones were not willing to adopt this measure citing, among other reasons, potential loss of production area and income, and risk of proliferation of weeds (Teagasc, 2012). However, the recent reduction of the buffer zones to 2 m (DAFM, 2012) might have contributed to increased adoption of this measure for decreasing P losses by Irish farmers.

2.7. Economics of dairy farms

2.7.1. Economic performance of dairy production systems in Ireland

Some authors (Shadbolt *et al.*, 2013) proposed farm productivity and profitability as measures of grazed-grass based dairy farms' economic performance. Other authors (Ridler, 2008) doubted the relevance of using productivity parameters, such as production per cow or per hectare, SR and kg feed per kg milk, as measures of economic performance on such farms. This is because these parameters are strongly influenced by the size of animals with direct implications on the feed requirement, quantity and quality of feeds, and therefore differences in associated expenditures and farm profitability. A number of Irish studies (Shalloo *et al*., 2004c; McCarthy *et al.*, 2007) indicated the genetic potential of dairy cows as an important factor impacting on farm expenditures and profitability. The strains with low milk potential, more suitable for grazed grass-based dairy production systems, showed the highest profitability, of €586 ha^{-1} (average value for both studies) or 4.41 cents (c) $litre^{-1}$ milk (average value for both studies).

Ridler (2008) also argued that the above-mentioned productivity parameters realistically reflect the economic performance of grass-based dairy farms when accounting for the different compositions of animal diets in relation to the milk yields per cow. In Ireland, this was well illustrated by Patton *et al.* (2012), who found milk yields of 5,606 kg cow^{-1} associated with animal diets consisting of 578 kg $cow^{-1}year^{-1}$ of concentrate feeds, the remainder being grazed grass. Comparatively, milk yields of 6,049 kg cow^{-1} were achieved with a concentrate input of 1,365 kg $cow^{-1}year^{-1}$, with the remainder accounted for by grazed grass.

Besides physical parameters, the importance of financial parameters, e.g. variable and fixed expenditures, as contributors to farm profitability, was also stressed in a number of Irish studies (Shalloo *et al.,* 2004b; Donnellan *et al.*, 2011; Humphreys *et al.,* 2012). In the study of Shalloo *et al.* (2004b), the main variable expenditures were concentrates (€190 ha^{-1}) and mineral fertilisers, lime and reseeding (€291 ha^{-1}), whereas the main fixed expenditure was hired labour (€972 ha^{-1}), with a farm profitability of €1,414 ha^{-1}. Donnellan *et al.* (2011) used profitability as a measure of competitiveness of Irish dairy farms, therefore considering both expenditures and returns. They calculated the proportion of expenditures out of total dairy output (milk and livestock sales). The highest calculated values were for feedstuffs (16.5 %), stock-related expenditures (9.4 %) and mineral fertilisers (7.5 %). Humphreys *et al.* (2012) stressed the importance of expenditure on mineral N fertiliser in relation to farm profitability by comparing mineral fertilised grass systems with white clover-based dairy production systems. This expenditure was €156 ha^{-1} higher on the fertilised grass systems. However, the profitability was similar for the two types of systems (€1,274 ha^{-1}), mostly due to similar returns (€2,094 ha^{-1}).

It can be concluded that different management strategies can generate different levels of profitability on grass-based Irish dairy production systems. Generally, on grazed grass-based dairy production systems, the profitability is increased by controlling the expenditures on buildings, plant, machinery and power used, and making maximum use of cheap grazed and conserved forages (Chamberlain, 2012).

2.7.2. Economic factors affecting Irish dairy systems

There are a number of identified factors influencing Irish dairy production systems, such as targeted milk yields, SR, input (concentrates, mineral fertilisers) and output (milk) market prices, milk quotas, milk quota system, and agricultural and environmental policies.

2.7.2.1. Targeted milk yields

Typically, on dairy farms, the levels of purchased supplements and ration balancing are dictated by targeted milk output per cow (Nousiainen *et al.*, 2011; Chamberlain, 2012). From an economic point of view, this is relevant in terms of feed use efficiency, reflected by the milk production response to supplementation (Shalloo *et al.*, 2004c) and expenditures. For example, Shalloo *et al.* (2004c) found an overall milk production response to increased concentrate supplementation (from 376 kg cow^{-1} to 1,540 kg cow^{-1}) of 1.06 litres milk per cow per additional kg of concentrate, which is at the high end of the normal response range. However, in the above study there were differences between the different strains in terms of response of milk production to imported concentrates. The increase in supplementation level was also associated with an increase of 1.7 c litre^{-1} milk in the expenditure on purchased feeds. Similarly, Patton *et al.* (2012) found a milk production response to increased concentrate supplementation (from 578 to 1,365 kg cow^{-1} year^{-1}) of 1.53 kg of milk additional kg of concentrate. However, increased feed supplementation was associated with an increase of €16,355 year^{-1} in the expenditure on purchased feed. These two studies underline the influence of on-farm management on expenditures, which ultimately impact on farm profitability.

2.7.2.2. *Stocking rate*

The SR may have economic implications on milk returns, on one hand, and expenditures on purchased feeds, herd maintenance, animal housing and labour on the other hand. McCarthy *et al*. (2007) found that an increase of 0.27 LU ha^{-1} in SR, without increasing the level of supplementation (336 kg cow^{-1}), was associated with an increase of €248 ha^{-1} in milk returns but also €360 ha^{-1} in total expenditure. However, the profitability was similar (€630 ha^{-1}) at lower and higher SR, suggesting that when land is a limiting factor to dairy production, as it often happens on Irish dairy farms (Donnellan *et al.,* 2011; Patton *et al.,* 2012; Kelly *et al.,* 2013), higher stocked systems will be the most profitable, due to their capability for low-cost high milk productivity per hectare (McCarthy *et al*., 2007).

Patton *et al.* (2012) reported that an increase of 0.47 LU ha^{-1} in SR, coupled with increased level of supplementation (by 787 kg cow^{-1}), was associated with an increase of €348 ha^{-1} in milk returns but also €335 ha^{-1} in total expenditure. However, similar to McCarthy *et al*. (2007), the profitability was similar (€1,218 ha^{-1}) at lower and higher SR. This was because increased feed, stock turnover, labour, and animal housing-related expenditures were compensated by the increased milk returns achieved at higher SR.

These two studies partially support the argument of Brennan and Patton (2010) that in grazed grass-based dairy production systems, increases in SR can determine increases in farm profitability when there is a good match between SR and the grass growing potential of the farm, to allow increased grass utilisation, if at the same time there are no major additional imports of either concentrate or mineral fertilisers, associated with increased expenditures.

2.7.2.3. *Input costs and output prices*

Animal feeds and mineral fertilisers are the main inputs which affect the cost of milk production (Donnellan *et al.*, 2011). Relatively high milk prices (€0.29 litre^{-1}, on average; CSO, 2012) between 2001 and 2011 and low fertiliser N prices (€0.80 kg N^{-1}, on average; CSO, 2012) between 2000 and 2010 in Ireland have encouraged high use of inputs in the form of mineral N fertilisers (238 kg N ha^{-1}, Treacy *et al.*, 2008; 178 kg N ha^{-1}, Buckley *et al.*, 2013), and concentrate feeds (between 699 and 971 kg DM cow^{-1}, Shalloo *et al.*, 2004c; McCarthy *et al.*, 2007; Patton *et al.*, 2012) on grazed grass-based dairy production systems. These high fertiliser applications may often be attributed to risk aversion to lower yields or incentive emerging from fertiliser pricing. The lower the relative price of fertiliser, the greater the incentive to apply it to offset potential risk and yield uncertainty (Buckley and Carney, 2013). Due to the fact that herbage yields are not stable during the season and that there is large inter-annual grass growth variability (Peyraud *et al.*, 2010), intensive grass-based milk production systems generally rely on strategic concentrate supplementation during times of herbage deficit, to sustain milk output per hectare at economically viable levels (Ryan *et al.,* 2011). However, the volume of bought-in feeds is often driven more by the desire to produce specific volumes of product rather than by the most efficient use of inputs. Concurrently, there has been a general tendency to overlook the importance of the 'free' resource (pasture) (Ridler, 2010).

Labour is another important and costly input on dairy farms. Farm labour requirement on dairy farms includes milking, maintenance, grassland management, calf care, cleaning, and veterinary as main tasks. The labour expenditure, calculated assuming 1,848 hours year^{-1} for one labour unit, may be equated with an expenditure of €22,855 year^{-1} (Shalloo *et al.,* 2004b). Finneran *et al.* (2012) emphasised the importance of contractor expenditure on grass-based Irish livestock production systems by comparing the expenditures on a grazed grass-based with a cut silage-based livestock production system. The fertiliser expenditure was €236 ha^{-1} higher on the cut silage-based system due to the contractor spreading expenditure. The contractor charge alone was €528 ha^{-1} higher on the cut silage-based system. These charges resulted in a total feed expenditure which was €1,156 ha^{-1} higher on the cut silage-based system, with potential negative impacts on the profitability. Therefore, dairy farmers can only keep labour expenditure under control by questioning and justifying any extra routines undertaken, as well as relying more on family rather than hired labour (Chamberlain, 2012), because they have no control on the size of agricultural wages for the different tasks on their farms.

For any business, a primary economic key driver is the demand for its products and the resulting price from sales (Von Keyserlingk *et al.,* 2013). For the dairy farmer, this driver is mainly the milk and processed dairy products ultimately purchased by consumers. However, the existence of many producers competing for the sale of milk means that in today's market, there is limited possibility for dairy farmers to influence the price they receive (Von Keyserlingk *et al.,* 2013). Moreover, due to increasing instability of milk prices and increasing expenditures on inputs (Soder and Rotz, 2001), as well as rising labour expenditures (MacDonald *et al.,* 2008), dairy farmers are searching for ways to decrease expenditures on milk production, and grazed grass-based dairy systems offer opportunities to reduce these expenditures (Soder and Rotz, 2001; MacDonald *et al.,* 2008).

Among the strategies that can be considered to reduce expenditures on grazed grass-dairy production systems are the increases in resource (grassland, labour, supplements) use efficiency (through size of grassland and herd, targeted herbage and milk yields, quality of supplements and animal intake) (Ridler, 2008; Finneran *et al.*, 2011; Patton *et al.,* 2012; Kelly *et al.,* 2013), increases in N-eco-efficiency (the amount of milk produced per kg of N surplus, Nevens *et al.*, 2006; Beukes *et al.,* 2012), and accounting for NFRV of organic N contained in cattle slurry (Lalor, 2008) or N_2 fixed by white clover in pastures (Humphreys *et al.*, 2012). Therefore, there has been a rejuvenated interest in grazed grass-based dairy production systems internationally (MacDonald *et al.,* 2008)**.**

2.7.2.4. Milk quota system

The introduction of milk quotas took place in 1984 in the EU, as an instrument for regulating milk production, which had increased dramatically along with associated expenses for subsidised exports and storage. Since its introduction, the milk quota has become a scarce production factor, and, as a consequence, allowed profitable milk producer prices and maintained dairy activities in less competitive regions (Kempen *et al.*, 2011). In Ireland, milk quota is allocated in amounts ranging between <50,000 litres and >450,000 litres per farm, depending on the area used for milk production. Supplementarily, dairy farmers lease in milk quotas between 151,109 and 8,487,765 litres (DAFM, 2013a).

In 2008, the "Health Check" decisions of CAP included the expiry of the milk quota system after 2014 and an increase of quotas by 1 % annually from 2009 to 2013 to allow for a "soft landing" of the milk sector with expiring quotas (Kempen *et al.,* 2011; Geary *et al.*, 2010, McDonald *et al*., 2013). In Ireland, the on-going EU dairy sector policy reform will result in quota abolition by 1 April 2015, while milk quotas are being expanded between 2008 and 2014 to facilitate this market transition (O'Donnell *et al.,* 2011). Therefore, the Irish dairy industry is targeting a 50 % increase in dairy output by

2020, as set out in the Food Harvest 2020 report, the national strategy for sustainable growth of the agricultural sector (DAFM, 2013b).

It is anticipated that the abolition of milk quotas will create an imbalance between milk supply and milk demand and therefore high milk price volatility (Kelly *et al.,* 2012). In fact, in the EU 27 countries, milk price has been highly volatile since 2007, ranging between 27 and 35 c litre^{-1} (CSO, 2013). Moreover, the adoption of market prices without any subsidy would increase the pressure to intensify, making the intensive dairy production systems unviable (Goulding *et al.,* 2008). The economic viability of farms refers to the ability to generate sufficient funds to sustain their production potential in the long run (European Comission, 2001).

Where quota is not limiting, output from the farm is maximised through increasing milk sales until marginal revenue from additional milk sales is equal to the marginal cost of the additional milk produced (McCarthy *et al.*, 2007). For example, Shalloo *et al.* (2004c) reported a €642 ha^{-1} net profit in a non-quota situation. This profit was associated with €254 ha^{-1} lower milk returns caused by a decline in milk price by 4.7 c litre^{-1}, which can occur in a market not being constrained by milk supply (Geary *et al.,* 2012).

2.7.2.5. Agri-environmental legislation

In many developed countries, much of commercial farming operates under the influence of society's increasingly multifunctional expectations. Such farming must thus be sustainable within a range of economic and environmental criteria (Crosson *et al.,* 2007). Therefore, a comprehensive evaluation of management effects on dairy farms must consider farm performance, environmental impacts, and potential profit (Rotz *et al.,* 2005).

As grasslands are recognised to have various roles in providing regulating and supporting services (Peyraud *et al.,* 2010), in the EU, grass-based milk production is regulated by a number of agricultural and environmental policies. One of them was the Luxembourg Agreement, introduced in 2003, which included Single Farm Payments (SFP) to directly support dairy farm incomes. The aim was the reduction in the intervention support prices for butter and skimmed milk powder, which was anticipated to determine a reduction in the farm-gate milk prices in future years (Donnellan *et al.,* 2011). Due to the Mid Term Review of CAP, SFP have been decoupled from production. The decoupling of SFP was implemented on January 1st, 2005 in Ireland. From this date, SPF is applicable to farmers who actively farmed during the reference years 2000, 2001 and 2002, who were paid Livestock Premia or Arable Aid in one or more of those years and who will continue to farm in the current year of claiming the payment (Hennessy *et al.*, 2005). After decoupling, SFP are based on the average number of animals or the average number of hectares (in the case of Arable Aid) on which payments were made in the three reference years (Connolly *et al.*, 2009). With decoupling, the farmers still receive these payments, regardless of production levels, as long as land is maintained in accordance with good farming practice (Hennessy *et al.*, 2005).

The GAP Regulations limit the SR and N and P use on Irish farms (European Communities, 2010a). Additionally, some of the Irish dairy farmers chose to participate in the Rural Environment Protection Scheme (REPS). This was a programme co-funded by the EU and the Irish government whereby farmers were rewarded financially for operating to a set of guidelines consistent with an agri-environmental plan drawn up by an approved planner. Important conditions for receiving REPS financial support were to limit SR to 2 LU ha^{-1} and to apply fertilisers to the farming area according to fertiliser plans drawn up for their farms (DAFM, 2013c).

All these regulations have economic implications on Irish dairy farms. For example, the limits on N and P use imposed through GAP Regulations and REPS, together with any good farming practice allowing for SFP aids, are most likely to lower the herbage yields and the profitability of Irish dairy farms. This is because the application of N fertilisers on grass-based dairy production systems is required to support DM yields of grass (Hennessy *et al.*, 2008), and, therefore, assuming maximum grass utilisation by the herd and all other factors being equal, to increase grass DM intake by the herd, and in turn, to increase milk production (Stakelum and Dillon 2007; Coleman *et al.*, 2010). The P fertilisation is needed in amounts ensuring that the herbage fed to animals has high enough P concentration to satisfy the dietary requirements of grazing livestock (Haygarth *et al.*, 1998). Also, concentrated feeds, which might be limited under good farming practices, were found to contribute to increases in milk yields and profitability on Irish dairy production systems (Shalloo *et al.*, 2004c; McCarthy *et al.*, 2007; Patton *et al.*, 2012).

Moreover, the limit on SR may be associated with a cost of compliance, representing the number of animals that need to be removed from the farm and associated loss of profit. This cost can reach €2,000 ha^{-1} (Hennessy *et al*., 2005).

On the other hand, the limit on N use may contribute to increased N-eco-efficiency on dairy farms. Nevens *et al.* (2006) found that the threshold of maximum 150 kg N ha^{-1} for N surplus, considered safe for complying with the limit of <50 mg NO_3 $litre^{-1}$ in the groundwater, can be attained at production levels of up to 10,000 litres milk ha^{-1}, which were reached by 18 progressive Flemish farms. A target value of 85 (range: 60-110) litres milk kg N^{-1} surplus was also established. In Ireland, Dillon and Delaby (2009) calculated a mean N-eco-efficiency of 48 litres milk kg N^{-1} surplus for a range of Irish dairy production systems recording a mean milk yield of 7,736 litres ha^{-1} and a mean N surplus of 162 kg N ha^{-1}.

Another positive aspect of N limit is the possibility to save on expenditures on N fertilisers. This can be done through the introduction of white clover in swards, which can help lowering expenditures on mineral fertilisers by €148 ha^{-1} due to the replacement of mineral fertiliser N by biologically fixed N in white clover (Humphreys *et al.*, 2012). Also, in Ireland, there is a price premium paid when 50 % of annual milk produced on organic farms is supplied between September and March (Keogh *et al*., 2010). If this opportunity for niche market (Von Keyserlingk *et al.*, 2013) is added to reduction in expenditures on mineral N fertilisers, the white clover-based dairy production systems represent a profitable alternative to mineral N fertilised grass-based dairy production systems in Ireland (Keogh *et al*., 2010).

2.7.3. Economic comparison of Irish dairy systems to those in other countries

Donnellan *et al.* (2011) conducted a study on the competitiveness of the Irish dairy sector at farm level under the circumstances of less prevalent EU policy supports and expected abolition of milk quota in 2015. The authors found that among EU dairy specialist producers (Belgium, Denmark, France, Germany, Italy, Netherlands, UK), Ireland has expenditure disadvantage in terms of mineral fertilisers (6.3 % of total output, compared with 2.32 %, on average, for the EU producers), and expenditure advantage in terms of purchased feedstuffs (3.8 % of total output, compared with 4.05 %, on average, for the EU producers).

As part of GREENDAIRY project, concerned with improvements of environmental performance and competitiveness of dairy farms, Raison *et al.* (2006) studied nutrient management and economic performance on 139 farms located in 9 regions on the Atlantic seaboard of Europe for three years. Among these farms, 47 farms in the northern regions (Scotland, South of Ireland, and South-West England) and 28 French farms were grazed grass, maize silage-based dairy production systems, 9 French farms were <10 % grazed grass, maize silage-based dairy production systems, while 55 farms in the southern regions (Basque Country, Galicia, and North Portugal) were completely in-door zero-grazing dairy systems. Mean milk receipts (€1,804 ha^{-1}) on 24 Irish dairy farms were lower than mean milk receipts (€2,391 ha^{-1}) on the Scottish and English farms and similar to mean milk receipts (€1,812 ha^{-1}) on the French farms. The average milk price (27 c litre^{-1}) received on the Scottish and English farms was lower compared with the Irish farms (30 c litre^{-1}), but the average milk yields (8,501 litres ha^{-1}) were higher compared with the Irish farms (7,757 litres ha^{-1}). Conversely, on the French farms, the average milk price of the sold milk (32 c litre^{-1}) was higher but the average milk yields (5,401 litres ha^{-1}) were lower compared with the Irish farms.

In the same study, mean concentrate expenditure was €295 ha^{-1} on the Irish farms, with concentrate inputs of 580 kg cow^{-1} compared with €449 ha^{-1}, on average, on Scottish and English farms, with 1,890 kg concentrate cow^{-1}, on average, and €1,593 ha^{-1}, on average, on the Spanish and Portuguese farms, with 3,615 kg concentrate cow^{-1}, on average. This reflects the lower input Irish dairy system, with low use of concentrates and high reliance on grazed grass.

Also, mean net profit on the Irish farms (€643 ha^{-1}) was higher than that on Scottish and English farms (€444 ha^{-1}), French farms (€311 ha^{-1}) and Spanish and Portuguese farms (€215 ha^{-1}). It is noticeable that the net margin was the lowest on the Spanish and Portuguese farms, which were confined dairy farms, with zero-grazing, as opposed to the Irish, Scottish and English farms, which included grazed grass as an input for milk production. This is very likely because of the cumulative effect of much higher feed, housing, herd maintenance, and labour expenditures on the confined farms.

On a unit product basis, mean milk receipts in two Irish studies were lower (25 cents kg milk^{-1}, McCarthy *et al.*, 2007; 28 cents kg milk^{-1}, Patton *et al.*, 2012) than mean milk receipts (35 cents litre^{-1}) on English dairy farms (DairyCo, 2013). This was mostly because of higher price (34 cents litre^{-1}) received for sold milk on the English farms (DairyCo, 2013) compared with the Irish farms (22 cents kg milk^{-1}, McCarthy *et al.*, 2007; 26 cents kg milk^{-1}, Patton *et al.*, 2012).

Mean feed expenditure on Irish dairy farms (6.14 cents kg milk^{-1}, Shalloo *et al.*, 2004c; 5.17 cents kg milk^{-1}, McCarthy *et al.*, 2007) was found to be similar to English dairy farms (6.23 pence litre^{-1}, Chamberlain, 2012). This was most likely because of similar mean milk yields (6,625 litres cow^{-1}, Shalloo *et al.*, 2004a; 6,576 litres cow^{-1}, McCarthy *et al.*, 2007; 6,996 litres cow^{-1}, Chamberlain, 2012) associated with similar feed rations.

Wilson (2011) showed that net profit per litre milk is linked to milk price and total expenditure. The author found that the average highest net profit (4.00 pence litre^{-1}) on 50 dairy farms in England was associated with a milk price of 23.6 pence litre^{-1} and a total expenditure of 18.5 pence litre^{-1}. DairyCo (2013) emphasised that there is little impact of milk price on profit, considerable variation in profit between farms being present at any given milk price. Therefore, there is scope to increase profit at any milk price. This report indicated that total expenditure had the strongest impact on net profit, with an increase of 11 pence litre^{-1} in total expenditure determining a decrease of 10.5 pence litre^{-1} in net profit. The highest net profit in the study of Wilson (2011) was similar to Irish studies (3.37 cents litre^{-1}, Shalloo *et al.*, 2004a; 4.15 cents litre^{-1}, McCarthy *et al.*, 2007). This illustrates the similarities in production systems and operational management.

It can be concluded that the lower milk yields but also lower expenditures on concentrates reflect low input system that is more typical in Ireland, with seasonal milk production (compact spring calving), low use of imported concentrates and forages, and high use of grazed grass. Comparatively, on continental Europe (except Britain), it is common to operate a high input system of dairy production, characterised by year-round milk production, high use of imported concentrates and forages, and lower use of grazed grass. The profitability of dairy farms largely depends on inputs and outputs prices, but ultimately on the ability of farmers to identify and control the highest expenditures on their farms.

Comparatively, the present study, as part of DAIRYMAN project (Plant Research International, 2013) focussed on identifying pathways for efficient use of increasingly expensive resources (mineral fertilisers and feeds) and for increasing competitiveness of 128 intensive dairy farms in ten regions from north-western Europe, of which 21 farms were Irish dairy farms. This research was conducted in the context of high potential for nutrient losses due to low efficiency of the use of mineral fertilisers and feeds on dairy farms (Steinshamn *et al.*, 2004; Nousiainen *et al.*, 2011) posing threats on their economic viability (Buckley and Carney, 2013).

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3. **Nitrogen balance and use efficiency on twenty-one intensive grass-based dairy farms in the South of Ireland**

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Summary

There is an increasing concern about balancing agronomic and environmental gains from nitrogen (N) usage on dairy farms. Data from a 3 year (2009-2011) survey were used to assess farm-gate N balances and N use efficiency (NUE) on 21 intensive grass-based dairy farms operating under the Good Agricultural Practice (GAP) regulations in Ireland. Mean stocking rate (SR) was 2.06 LU ha^{-1}, mean N surplus was 175 kg ha^{-1}, or 0.28 kg N kg MS^{-1} (milk solids), and mean NUE was 0.23. Nitrogen inputs were dominated by inorganic fertiliser (186 kg N ha^{-1}) and concentrates (26.6 kg N ha^{-1}), while outputs were dominated by milk (40.2 kg N ha^{-1}) and livestock (12.8 kg N ha^{-1}). Comparison to similar studies carried out before the introduction of the GAP regulations in 2006 would suggest that N surplus, both per ha and per kg MS, have significantly decreased (by 114 kg N ha^{-1} and 0.013 kg N kg MS^{-1}, respectively) and NUE increased (by 0.06), mostly due to decreased inorganic fertiliser N input and improvements in N management, with a notable shift towards spring application of organic manures, indicating improved awareness of the fertiliser value of organic manures and good compliance with the GAP regulations regarding fertiliser application timing. These results would suggest a positive impact of the GAP regulations on dairy farm N surplus and NUE, indicating an improvement in both environmental and economic sustainability of dairy production through improved resource use efficiencies. Such improvements will be necessary to achieve national targets of improved water quality and increased dairy production. The weak impact of SR on N surplus found in this study would suggest that, with good management, increased SR and milk output per ha may be achievable, while decreasing N surplus per ha. Mean N surplus was lower than the overall mean surplus (224 kg N ha^{-1}) from six studies of northern and continental European dairy farms, while mean NUE was similar, largely due to the low input system that is more typical in Ireland, with seasonal milk production (compact spring calving), low use of concentrates, imported feed and forages, high use of grazed grass and lower milk yields per ha.

3.1. Introduction

Irish dairy production systems tend to be relatively intensively managed compared to other Irish grassland agricultural production systems, and are pasture-based, with the objective of producing milk in a low cost system through maximising the proportion of grazed grass in the cows' diet. Increasing the proportion of grazed grass reduces milk production costs and can increase the profitability of grass based milk production systems in Ireland and other temperate regions (Dillon *et al.*, 2005; Dillon, 2011). Nitrogen (N) inputs, in the form of fertiliser and concentrate feeds, are key drivers of increased herbage yields and milk saleable output on most dairy farms (Treacy *et al.*, 2008; Ryan *et al.*, 2011; Gourley *et al.*, 2012). However, N inputs typically exceed N outputs in milk and livestock exported off the farms (Jarvis, 1993; Van Keulen *et al.*, 2000; Goodlass *et al.*, 2003; Aarts, 2003; Humphreys *et al.*, 2008). This imbalance results in surplus N that is either accumulated on, or lost from, the dairy farm (Gourley *et al.*, 2010; Cherry *et al.*, 2012).

As N surplus is commonly associated with excessive, inefficient N use on farms, as well as harmful environmental impacts (Leach and Roberts, 2002; Eckard *et al.*, 2004; Powell *et al.*, 2010), it is considered to be an indicator of potential N losses and environmental performance (Schröder *et al.*, 2003; Carpani *et al.*, 2008). Nitrogen surplus potentially accumulates in soil organic matter (SOM) (Jarvis 1993) or is lost through denitrification, nitrate (NO_3) leaching, ammonia (NH_3) volatilisation (Pain, 2000; Jarvis and Aarts, 2000; Del Prado *et al.*, 2006) or through runoff to surface waters (De Vries *et al.*, 2001). Denitrification is naturally facilitated in Ireland, due to common anaerobic soil conditions and the generally high content of organic carbon (C) in soils (between 2 and 7 %; Dillon and Delaby 2009) enabling development of denitrifying bacteria. These N losses can have negative environmental impacts such as eutrophication of surface waters, pollution of groundwater aquifers, ozone depletion, and anthropogenic climate change (in the case of N_2O emissions) (Leach and Roberts, 2002; Eckard *et al.*, 2004; O'Connell *et al.*, 2004). It has been emphasised that dairy production should ideally be achieved in a sustainable manner, without impairing natural capital (soils, water, and biodiversity) (Goodland, 1997). Improved nutrient use efficiency has a significant role to play in the development of more sustainable dairy production systems (Goulding *et al.*, 2008). Among the nutrient imports in dairy production systems, N is particularly important as it is used in large quantities, between

172 and 301 kg N ha^{-1} (Groot *et al.*, 2006; Nevens *et al.*, 2006; Roberts *et al.*, 2007; Ryan *et al.*, 2011; Cherry *et al.*, 2012) but with generally low efficiency (Goulding *et al.*, 2008). In Europe, N use efficiency (NUE; proportion of N imports recovered in agricultural products (Ryan *et al.*, 2012)) values of between 0.17 and 0.38 have been recorded (Mounsey *et al*., 1998; Groot *et al.*, 2006; Nevens *et al.*, 2006; Raison *et al.*, 2006; Roberts *et al.*, 2007; Treacy *et al.*, 2008; Cherry *et al.*, 2012; Oenema *et al.*, 2012).

In grass-based dairy production systems, there are a number of factors limiting NUE, such as N losses from manure, slurry and mineral fertiliser management and application to land (Webb *et al.*, 2005), losses from dung and urine deposited by grazing animals, the ability of grass plants to convert N from applied mineral fertiliser and manure into biomass in herbage, utilisation by animals of grass herbage grown and the biological potential of cows to convert N from concentrate feeds and herbage into milk (Powell *et al.*, 2010). More effective use of N imports in fertiliser N and concentrate feeds can potentially contribute to decreased imports and increased rates of NUE (Groot *et al*., 2006). Irish dairy production systems benefit from mild winters (5.1 ^{0}C in January) and annual rainfall between 800 and 1200 mm, allowing grass growth all year around and an extended grazing season that can be as long as February to November (Humphreys *et al.*, 2009a), varying with location and soil type. Irish dairy farms are unique in Europe in that the majority operate a seasonal milk production system with compact spring calving (from January to April) so that milk production matches grass growth. The proportion of grazed grass in the diet of dairy stock is hence maximised (Humphreys *et al.*, 2009a), allowing for the maximum amount of milk to be produced from grazed grass and reducing requirements for feeding concentrate feeds post-calving (Dillon *et al.*, 1995). For these reasons, the potential for more effective use of N on-farm and management strategies to achieve improved NUE may be expected to differ from those of the year-round feed-based dairy production systems more typical of continental Europe and Britain.

In this context, farm-gate N balances, as the difference between total N input and total N output passing the farm-gate (Aarts, 2003), are a useful tool for farmers, scientists and policy-makers to: (i) understand N flows and identify potential N losses (Watson and Atkinson, 1999); (ii) understand factors affecting, and develop strategies to control, potential N losses (Gourley *et al*., 2007; Beukes *et al*., 2012); and (iii) increase farmers' awareness of environmental regulations on farms and implementation of these regulations to control N losses to the environment (Oenema *et al.,* 2003; Carpani *et al.,* 2008).

In the European Union (EU), the Nitrates Directive (91/676/EEC) (European Council, 1991) has established guidelines in relation to farming practices to reduce NO_3 leaching that are implemented in each member state through a National Action Programme (NAP). In Ireland, these are legislated as the Good Agricultural Practice (GAP) Regulations (European Communities 2010), first passed in 2006. Under the Regulations, farms are limited to a stocking rate (SR) of 170 kg organic N ha^{-1}, equivalent to 2 livestock units (LU) ha^{-1}, or 2 dairy cows ha^{-1}. The Regulations also establish the quantity of available N that can be applied to grass and other crops (depending on factors such as SR or crop type), the volume of slurry and slurry storage required (depending on factors such as rainfall and stock type and number) and closed periods in winter months during which spreading of organic and inorganic fertilisers is restricted (depending on location in the country), as well as other measures on farm yard and field management aimed at minimising N losses to water. Farmers can apply for derogation to stock at up to 250 kg organic N ha^{-1} (2.9 LU ha^{-1}), subject to more stringent requirements, and this derogation is principally taken up by the more intensive dairy farms.

Although explicitly aimed at decreasing N losses to water, these Regulations might be expected to have improved NUE on farms, as most of the measures aim to decrease losses by increasing retention of N within the production systems. However, most of the existing data on dairy farm N balances in Ireland date from the period before the implementation of the Regulations in 2006 (Mounsey *et al.,* 1998; Treacy *et al.,* 2008). Ryan *et al.* (2011) and Ryan *et al.* (2012) examined N balances and use efficiencies in Irish dairy production systems but these were based on modelling and experimental studies. In the European context also, there are few farm-gate N balances on grassland-based dairy farms post the implementation of the Nitrates Directive (e. g. Groot *et al.,* 2006; Nevens *et al*., 2006; Raison *et al.,* 2006; Roberts *et al*., 2007; Cherry *et al.,* 2012; Oenema *et al*., 2012).

Therefore, the objectives of the current study were: (i) to assess farm-gate N balances and use efficiencies on 21 commercial intensive dairy farms operating under the Nitrate Regulations in Ireland and compare these to pre-Regulations studies to investigate the impact of the Regulations; (ii) to identify the factors influencing NUE on these farms; (iii) to explore potential approaches to increase NUE and decrease N surpluses on these farms. For this purpose, data on N imports and exports were recorded on 21 dairy farms participating in the INTERREG-funded DAIRYMAN project over 3 years, from 2009 to 2011.

3.2. **Materials and Methods**

3.2.1. Farm selection and data collection

Twenty-one commercial intensive dairy farms were selected, located in the South of Ireland, in counties Cork, Limerick, Waterford, Tipperary, Kilkenny, and Wicklow. These farms were pilot farms involved in the INTERREG-funded DAIRYMAN project (www.interregdairyman.eu) focusing on improving resource use efficiency on dairy farms in Northwest Europe. Farm selection was based on the likely accuracy of data recording, 8 of the farms in the current study having been involved in a previous similar study (GREENDAIRY; Treacy *et al.,* 2008), and all the farmers being willing to provide data. The selected farms were known as being progressive in their approach to farm management and, therefore, may not be fully representative for the Irish dairy industry as a whole. However, comparing farm area, stocking rate and milk yield per cow showed that the farms were close to, but slightly above, the national average for dairy farms. Grass-based milk production from spring calving cows was the main enterprise on all the selected farms.

Key farm characteristics are given in Table 3.1. Mean total utilised agricultural area (TUAA) was 71 (S.D. = 24.8) ha, mean SR was 2.06 (S.D. = 0.32) LU ha^{-1}, and mean milk yield was 5,308 (S.D. = 464) litres (l) cow^{-1} between 2009 and 2011, whereas national mean values for dairy farms were 52 ha for TUAA, 1.90 LU ha^{-1} for SR, and 4,956 litres cow^{-1} for milk yield, between 2009 and 2011 (Connolly *et al.*, 2009; Hennessy *et al.,* 2010; Hennessy *et al.,* 2011). Seventeen of the farms in the current study participated in the Rural Environment Protection Scheme (REPS). This was a program co-funded by the EU and the Irish government whereby farmers were rewarded financially for operating to a set of guidelines consistent with an agri-environmental plan drawn up by an approved planner. Important conditions for receiving REPS financial support were to limit SR to 2 LU ha^{-1} and to apply N fertilisers to the farming area according to fertiliser plans drawn up for each farm (DAFM, 2013c). Eight of the 21 farms had a SR higher than 170 kg organic N ha^{-1} or 2 LU ha^{-1}. According to GAP regulations and REPS conditions (for the participating farms), these farms had to apply for a derogation allowing a maximum SR of 250 kg organic N ha^{-1} or 2.9 LU ha^{-1}. However, the 17 farms closely adhering to GAP regulations were not fully representative of the Irish dairy farms and this may bias the interpretation of the results of the current study.

Data were collected on a monthly basis between 2010 and 2011 on the selected farms. The information collected included grassland area, area under crops, type of crops and percentage of crops fed to livestock, livestock numbers and type of livestock, number of days spent grazing, and imports of manure, concentrate feeds, bedding material, silage, mineral N fertilisers and other agro-chemicals, as well as exports of milk, crops, manure, and silage. For mineral N fertilisers, amounts imported onto farms as well as amounts applied to land were recorded on a monthly basis. For year 2009, similar data were obtained from farm records and farm advisors. Data collected for the 3 years were cross-checked with secondary data sources such as Single Farm Payment forms and Nitrates' Declaration forms (data forms required from farmers for participation in state schemes) (DAFM, 2013a, b). Data on livestock imports and exports were extracted from the Dairy Management Information System (DAIRYMIS) (Crosse, 1991). Values for amounts of milk sold off the farms were extracted from the reports on milk deliveries coming from the cooperatives supplied by the farmers. Data on soil types were extracted from REPS forms for the participating farms and from the national soil survey (Finch and Gardiner, 1993) for the remainder. Data on mean annual rainfall and temperature were extracted from an Irish Meteorological Service database for different weather stations located in, or close to, the area of study, at Cork airport, Roche's point, Gurteen, Johnstown Castle and Oak Park (Irish Meteorological Service, 2013).

Table 3.1. Total utilised agricultural area (and crop area), annual air temperature, annual rainfall, stocking rate, milk yields, milk solids exports, concentrate feeds, and estimated harvested grass through grazing and silage; soil type for 21 Irish dairy farms between 2009 and 2011

Farm	TUAA (crops) (ha)	Temp. (^{0}C)	Rainfall (mm year^{-1})	Soil type	SR (LU ha^{-1})	Milk yield (l cow^{-1})	MS exports (kg ha^{-1})	Conc. (kg DM LU^{-1})	Grass (kg DM LU^{-1})
1	85	9.6	1,077	CL	2.15	5,319	618	268	4,139
2	67	9.8	1,124	C	2.41	6,010	782	499	4,169
3	73	9.8	1,124	C	2.07	5,688	664	221	4,304
4	50	10.1	1,373	L	2.68	5,309	709	571	3,691
5	74 (1.2)	10.1	1,373	L	1.82	5,149	510	611	3,891
6	63 (3.9)	10.1	1,373	L	1.92	5,672	612	568	3,632
7	47	9.6	1,077	L	2.41	5,080	781	471	3,922
8	58	10.1	1,373	C	2.50	5,671	749	580	4,033
9	51	9.6	1,077	C	2.01	5,431	620	466	4,089
10	130 (5.5)	10.1	1,373	L	1.97	5,207	544	394	3,898
11	40	10.1	1,373	L	2.39	4,229	563	615	3,508
12	52	10.1	1,373	L	1.77	5,613	527	604	3,886
13	81	9.6	1,077	C	1.84	5,290	531	710	3,730
14	96 (6.7)	9.8	1,124	SL	1.80	4,415	437	302	3,472
15	128	9.8	1,124	L	1.88	4,671	446	484	3,858
16	78 (13.4)	10.2	1,453	C	1.58	6,038	474	801	3,746
17	72	9.6	1,077	C	2.47	4,928	707	463	4,002
18	48	9.8	1,124	CL	1.92	5,549	532	732	3,567
19	71 (2.3)	9.8	1,124	C	2.22	5,500	362	251	2,919
20	76 (6.2)	10.1	1,373	SL	1.97	5,174	584	265	4,011
21	48 (1.6)	10.1	1,373	L	1.40	5,522	443	386	4,108
Mean	71 (5.6)	9.9	1,235	-	2.06	5,308	581	488	3,837
S.D.	24.8 (3.91)	0.22	145	-	0.32	464	119	166	309

TUAA, total utilised agricultural area; temp., temperature; CL, clay-loam; L, loam; C, clay; SL, sandy-loam; SR, stocking rate; LU, livestock units; l, litres; MS, milk solids; conc., concentrate feeds; DM, dry matter; S.D., standard deviation.

The annual amount of pasture harvested through grazing and silage on each farm was modelled using the Grass Calculator (Teagasc, 2011) based on the difference between the net energy (NE) provided by imported feeds (concentrates and forages) and the net energy requirements of animals for maintenance, milk production, and body weight change (Jarrige, 1989). It was assumed that 1 kg dry matter (DM) of grass equals 1 feed unit for lactation (UFL).

Stocking rate was expressed as LU per ha for TUAA. One dairy cow was considered equivalent to 1 LU and 1 bovine less than 1 year old equivalent to 0.3 LU (Connolly *et al*., 2009).

3.2.2. Farm-gate nitrogen inputs, outputs, balances and use efficiencies

Nitrogen inputs and outputs were calculated both on a monthly and an annual basis. Nitrogen in fertiliser N was calculated by taking into account the N content of fertilisers applied to land. Monthly imported amounts of concentrate feeds and forages were assumed to be exhausted in the end of each month. Nitrogen imports in concentrate feeds, forages and bedding material onto farms were calculated by multiplying the total quantity by its crude protein (CP) concentration divided by 6.25 (McDonald *et al*,. 1995). Nitrogen fixed by clover was not included as an input due to the low prevalence of clover on the farms and resultant small contribution to the N budget (Gourley *et al.*, 2007). Nitrogen in livestock imported onto, or leaving, the farms was calculated by using standard values for live weight (Treacy, 2008) and multiplying it by 0.029 for calves and by 0.024 for older animals (ARC, 1994). Nitrogen in exported milk was calculated by dividing the milk protein concentration by 6.38 (ARC, 1994).

The farm-gate N balance was calculated as the difference between total N input and total N output and was expressed on both an areal basis (kg N ha^{-1}) and a unit product basis (kg N kg^{-1} milk solids (MS)) (Ryan *et al.*, 2012). Nitrogen use efficiency was calculated as the ratio between total N output and total N input, expressed as a proportion (Swensson, 2003).

3.2.3. Statistical analysis

Descriptive statistics were applied using SPSS to calculate means and standard errors (Darren and Mallery, 2008). Normal distribution of residuals was tested using Shapiro-Wilk, with values lower than 0.05 indicating a non-normal distribution. The log transformation was required to ensure homogeneity of variance (Tunney *et al.*, 2010) for some of the variables. Therefore, TUAA, milk fat and protein concentration, N inputs per ha from fertiliser N, concentrate feeds, forages, bedding material and livestock, NUE, N inputs per kg MS from fertiliser N and concentrate feeds, MS exports per cow, comparative N inputs from concentrate, N exports in sold milk, and NUE between the current study and two previous similar studies were transformed using a log10 base (y=log10(x)).

Differences in mean TUAA, SR, milk yields, milk protein and fat concentration, concentrate feed imports, N inputs, N outputs, N surplus, NUE and surplus N per kg MS between years and farms were analysed using repeated measures NOVA. A significance level of 0.05 or less (0.01 and 0.001) indicated statistically significant differences among the means. A significance level of 0.05 or higher indicated a 95 or higher percent of certainty that the differences among the means were not the result of random chance (Darren and Mallery, 2008). Such results were presented as not significant (NS).

The statistical models included farm and year effects on each of the tested variables. The 21 farms were considered as replicates. The models used were:

1. $Y_i = \mu + a_i + e_{i,}$ where Y_i = tested variable, a_i = the effect of *i*th farm ($i = 1,\ldots,21$), and e_i = the residual error term;
2. $Y_i = \mu + b_j + e_{i,}$ where Y_i = tested variable, b_j = the effect of *j*th year (j = 2009, 2010, 2011), and e_i = the residual error term.

Multiple stepwise linear regression was undertaken to investigate relationships between key dependent and independent variables presented in Table 3.2. The choice of the statistical models was dependent on the potential significance of independent variables and their potential impact on the dependent variables. Non-significant ($P > 0.05$) independent variables were automatically removed from the models (Table 3.2). The probability for acceptance of new terms (F) was 0.10 (Groot *et al.*, 2006) and the confidence interval was 0.95. All relationships between variables were assessed for outliers, normality and colinearity. Any identified outliers were diminished through log transformation.

Table 3.2. Investigated and significant multiple stepwise linear regression models

Investigated	Significant
LgFN = μ + βLgTUAA + βSR + βMSE + βGD + σ_{est}	LgFN = μ + SR + σ_{est}
LgCN = μ + βSR + βMSE+ βGD + σ_{est}	NS
MN = μ + βSR + βMSE + βGD + βLgFN + βLgCN + σ_{est}	MN = μ + SR+ σ_{est}
LN = μ + βSR + βGD + βLgFN + βLgCN + σ_{est}	NS
NSR = μ + βLgTUAA + βSR + βMSE + βGD+ βLgFN + βLgCN + σ_{est}	NSR = μ + βSR + βLgFN + βLgCN + σ_{est}
LgNUE = μ + βSR + βMSE + βGD + βLgFN + βLgCN + σ_{est}	LgNUE = μ - LgFN + σ_{est}
NMS = μ + βLgMS + βGD + βLgFNMS + βLgCNMS + σ_{est}	NMS = μ + βLgFNMS + βLgCNMS - βLgMS σ_{est}

LgFN, mean log transformed fertiliser N input; LgCN, log transformed concentrate N input; MN, milk N output; LN, livestock N output; NSR, N surplus per ha; LgNUE, log transformed N use efficiency; NMS, surplus N per kg milk solids; LgTUAA, mean log transformed total utilised agricultural area; SR, stocking rate; MSE, milk solids exports per ha; GD, number of days spent grazing; LgMS, log transformed milk solids exports per cow; LgFNMS, log transformed fertiliser N input per kg milk solids; LgCNMS, log transformed concentrate N input per kg milk solids; β, standardized coefficient of regression; σ_{est}, standard error of the estimate; NS, not significant.

Uncertainty analysis was carried out by calculating the coefficient of variation as the ratio between standard deviation and mean value (Gourley *et al.*, 2010) for each N input, N output, N balance and NUE on the 21 farms between 2009 and 2011, expressed as a proportion.

3.3. **Results**

3.3.1. Nitrogen inputs

There was a high degree of variation in mean N inputs, between years and farms (Table 3.3). Mean total N input was 228 kg N ha^{-1} (Table 3.3.). There were significant differences in mean total N input between farms, ranging from 118 to 301 kg N ha^{-1} over the 3 years (Table 3.3.). The coefficient of variation (mean value divided by standard deviation) for mean total N input between farms was 0.25 over the 3 years. There were also significant differences in mean total N input between years, ranging from 191 kg N ha^{-1} to 265 kg N ha^{-1} (Table 3.3.). The main sources of N input onto farms were mineral N fertilisers and concentrate feeds, accounting for 0.81 and 0.11, respectively, of total N input. Mean fertiliser N input was 186 kg N ha^{-1} (Table 3.3.). There were significant differences in mean fertiliser N input between farms, ranging from 101 to 261 kg N ha^{-1} over the 3 years (Table 3.3.). The coefficient of variation for mean fertiliser N input between farms was 0.27 over the 3 years. There were also significant differences in mean fertiliser N input between years, ranging from 160 kg N ha^{-1} to 209 kg N ha^{-1} (Table 3.3.). On a monthly basis, mean fertiliser N input was highest between March and June, at 40 (S.D. = 4.84) kg N ha^{-1} (Fig. 3.1). Mean concentrate N input was 26.6 kg N ha^{-1} (Table 3.3.). There were significant differences in mean concentrate N input between farms, ranging from 7.7 to 40.3 kg N ha^{-1} over the 3 years (Table 3.3.). The coefficient of variation for mean concentrate N input between farms was 0.39 over the 3 years. There were also significant differences in mean concentrate N input between years, varying between 25.3 kg N ha^{-1} and 34.4 kg N ha^{-1} (Table 3.3.).

Table 3.3. Mean values (and standard errors), grand means between years and ranges between farms for N inputs in mineral fertilisers, concentrate feeds, forages, bedding material and livestock, N outputs in sold milk and livestock, farm-gate N balances, N use efficiencies and surplus N per kg milk solids for 21 Irish dairy farms between 2009 and 2011; standard error of the means for transformed data in brackets; P-values from ANOVA are included

	Year			Grand mean	S.E.M.	Range farms	*P*-value	
	2009	2010	2011				Y	F
N inputs (kg N ha^{-1})								
Mineral fertilisers	160	209	191	186	7.50(0.01)	101 - 261	<0.05	<0.001
Concentrate feeds	25.3	34.4	20.1	26.6	1.70(0.03)	7.7 - 40.3	<0.05	<0.001
Forage	0.0	14.2	10.6	12.4	2.94(0.06)	0.6 - 41.9	<0.05	NS
Bedding material	0.0	4.7	3.4	4.0	0.63(0.04)	0.9 - 12.8	<0.001	NS
Livestock	5.3	2.2	3.9	3.8	1.41(0.04)	0.1- 11.1	NS	NS
Total	191	265	229	228	8.43	118 - 301	<0.01	<0.001
N outputs (kg N ha^{-1})								
Milk	37.4	43.3	39.9	40.2	1.12	26.8 - 55.3	NS	<0.001
Livestock	11.3	13.9	13.4	12.8	0.68	6.7 - 23.3	NS	<0.01
Total	48.7	57.2	53.2	53.0	1.62	37.1 - 75.3	<0.05	<0.001
N balance (kg N ha^{-1})	142	207	176	175	7.40	69 - 239	<0.01	<0.001
N use efficiency	0.25	0.21	0.23	0.23	0.009(0.013)	0.18 - 0.42	NS	<0.01
Surplus N kg kg MS^{-1}	0.25	0.32	0.28	0.28	0.001	0.16 - 0.44	NS	<0.05

N, nitrogen; MS, milk solids; S.E.M., standard error of the means; Y, year; F, farm; NS, not significant.

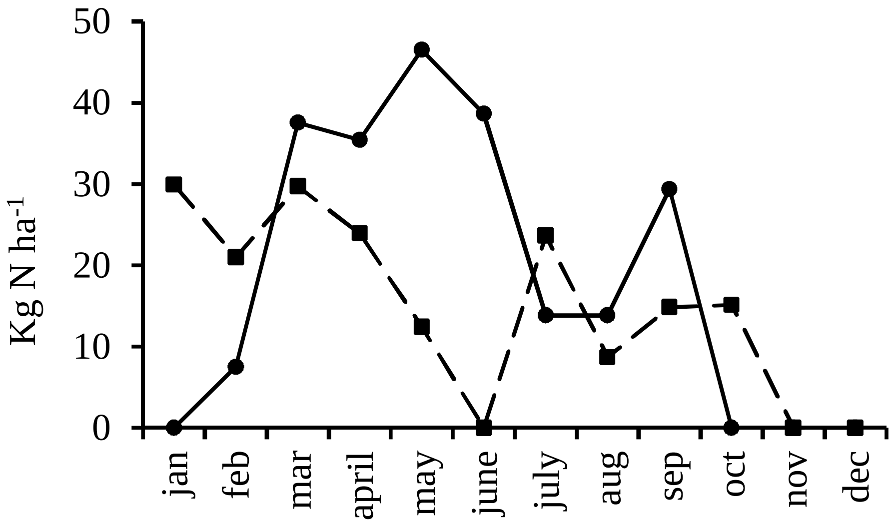

Fig. 3.1. Monthly application rates of mineral (——) and organic (- -- -) N fertilisers (kg N ha^{-1}) on 21 Irish dairy farms between 2009 and 2011

There was a significant positive relationship (R^2 = 0.49; P < 0.001) between mean log transformed fertiliser N input and mean SR. An increase of 0.07 LU ha^{-1} in mean SR was associated with an increase of 0.01 (9, not transformed) kg N ha^{-1} in mean log transformed fertiliser N input. There was no significant relationship between mean log transformed concentrate N input and mean SR, MS export, and number of days spent grazing (Table 3.2).

3.3.2. Nitrogen outputs

Mean total N output was 54.3 kg N ha^{-1} (Table 3.3.). There were significant differences in mean total N output between farms, ranging from 37.1 to 75.3 kg N ha^{-1} over the 3 years (Table 3.3.). The coefficient of variation for mean total N output between farms was 0.19 over the 3 years. There were also significant differences in mean N output between years, ranging from 48.7 kg N ha^{-1} to 57.2 kg N ha^{-1} (Table 3.3.). The main sources of N output were sold milk and livestock, accounting for 0.76 and 0.24, respectively, of total N output. Mean milk N output was 40.2 kg N ha^{-1}, ranging from 37.4 kg N ha^{-1} to 43.3 kg N ha^{-1} (Table 3.3.). There were significant differences in mean milk N output between farms, ranging from 26.8 to 55.3 kg N ha^{-1} over the 3 years

(Table 3.3.). The coefficient of variation for mean milk N output between farms was 0.19 over the 3 years. Mean livestock N output was 12.8 kg N ha^{-1}, ranging from 11.3 kg N ha^{-1} to 13.9 kg N ha^{-1} (Table 3.3.). There were significant differences in mean livestock N output between farms, ranging from 6.7 to 23.3 kg N ha^{-1} over the 3 years (Table 3.3.). The coefficient of variation for mean livestock N output between farms was 0.31 over the 3 years.

There was a significant positive relationship (R^2 = 0.49; P < 0.001) between mean milk N output and mean SR. An increase of 0.07 LU ha^{-1} in mean SR was associated with an increase of 1.43 kg N ha^{-1} in mean milk N output. There was no significant relationship between mean livestock N output and mean SR, number of days spent grazing, log transformed fertiliser N input and log transformed concentrate N input (Table 3.2.).

3.3.3. *Nitrogen balance and nitrogen use efficiency*

The N balance on all farms was in surplus. Mean N surplus (N inputs less N outputs) was 175 kg N ha^{-1} (Table 3.3.). There were significant differences in mean N surplus between farms, ranging from 69 to 239 kg N ha^{-1} over the 3 years (Table 3.3.). The coefficient of variation for mean N surplus between farms was 0.29 over the 3 years. There were also significant differences in mean N surplus between years, ranging from 142 kg N ha^{-1} to 207 kg N ha^{-1} (Table 3.3.). Mean NUE (N outputs divided by N inputs) was 0.23, varying from 0.21 to 0.25 (Table 3.3.). There were significant differences in mean NUE between farms, ranging from 0.18 to 0.42 over the 3 years (Table 3.3.). The coefficient of variation for mean NUE between farms was 0.20 over the 3 years. Mean surplus N per kg MS was 0.28 kg N kg MS^{-1}, ranging from 0.25 kg N kg MS^{-1} to 0.32 kg N kg MS^{-1} (Table 3.3.). There were significant differences in mean annual surplus N per kg MS between farms, ranging from 0.16 to 0.44 kg N kg MS^{-1} over the 3 years (Table 3.3.). The coefficient of variation for mean surplus N kg MS^{-1} between farms was 0.24 over the 3 years.

There was a significant positive relationship (R^2 = 0.91; P < 0.001) between mean N surplus and mean log transformed fertiliser N input (β = 0.91), mean log transformed concentrate N input (β = 0.14), and mean SR (β = 0.02). An increase of 0.01 (9, not transformed) kg N ha^{-1} in mean log transformed fertiliser N input, 0.02 (1.63, not transformed) kg N ha^{-1} in mean log transformed concentrate N input and 0.07 LU ha^{-1} in mean SR was associated with an increase of 8 kg N ha^{-1} in N surplus.

There was a significant negative relationship (R^2 = 0.42; P < 0.001) between mean log transformed NUE and mean log transformed fertiliser N input (β = -0.42). An increase of 0.01 (9, not transformed) kg N ha^{-1} in mean log transformed fertiliser N input was associated with a decrease of 0.019 (0.012, not transformed) in NUE.

There was a significant relationship (R^2 = 0.88; P < 0.001) between mean surplus N per kg MS and mean log transformed fertiliser N input per kg MS (β = 0.90), mean log transformed concentrate N input per kg MS (β = 0.17) and mean log transformed MS export per cow (β = -0.15). An increase of 0.018 (0.012, not transformed) kg N kg MS^{-1} in mean log transformed fertiliser N input and 0.02 (0.003, not transformed) kg N kg MS^{-1} in mean log transformed concentrate N input was associated with an increase of 0.01 in surplus N per kg MS. An increase of 0.01 (13, not transformed) kg MS cow^{-1} in log transformed MS exports per cow was associated with a decrease of 0.01 in surplus N per kg MS.

3.4. **Discussion**

Total N input, output and surplus in the current study were close to, but slightly above, the national average for dairy systems and NUE was close to the national average found by Buckley *et al*. (2013) (mean total N input of 178 kg N ha^{-1}, mean total N output of 41 kg N ha^{-1}, mean N surplus of 139 kg N ha^{-1} and mean NUE of 0.24) for a nationally representative sample of 195 specialist dairy farms for 2009-2010. This would suggest that results from this study can be taken as indicative of the national situation.

The overall coefficient of variation for N inputs, outputs, balances and NUE, of 0.27, was above the generally accepted limit of 0.10 (Mulier *et al.*, 2003) but within the limit of 0.30 reported in other studies on farm-gate nutrient balances (Swensson, 2003; Nevens *et al*., 2006; Fangueiro *et al*., 2008).

3.4.1. Factors affecting N balances and use efficiencies across farms

Differences in fertiliser N input between farms were principally associated with differences in SR, with a significant positive relationship between fertiliser N and SR. In a grazed grass-based dairy production system, increased SR requires increased grass DM intake by the herd (Stakelum and Dillon, 2007; Coleman *et al.*, 2010) and therefore, assuming maximum grass utilisation by the herd and all other factors being equal, increased DM yields of grass and, in turn, increased requirement for fertiliser N input (Hennessy *et al.*, 2008). However, overall available N input can potentially exceed pasture N requirement and factors such as application rates, forms and timings can lead to inefficient use of N. Stocking rate explained only 0.49 of the variation in mean fertiliser N input. The remaining variation may be explained by factors such as advisory impact and understanding and planning on the part of the farmer, economic considerations and weather and grass growth conditions.

Concentrate N input was closely associated with imported concentrate feeds, ranging between 221 and 801 kg DM LU^{-1} between farms. Feed imports were likely determined by harvested grass, ranging between an estimated 2,919 and 4,304 kg DM LU^{-1} and targeted milk yields per cow, ranging between 4,229 and 6,038 litres cow^{-1}. Targeted milk yields per cow were included in development plans introduced in 2009 for each farm by farm advisors. One of the goals in the development plans was increased milk yield per cow by amounts ranging between 100 and 400 litres cow^{-1} between 2009 and 2011.

Differences in milk N output were associated with differences in SR between farms. The significant positive relationship between milk N output and SR implies that increasing SR is an effective strategy to increase milk N output. Further, this could positively affect N surplus and NUE, because N in sold milk was the main form of exporting N inputs off the farms. However, from 228 kg N ha^{-1} of mean total N input, only 40.2 kg N ha^{-1} or 0.17, on average, was exported in sold milk, meaning that the impact of milk N output on N surplus and NUE was rather low. The N content of sold milk is very unlikely to increase and, therefore, there is a need to optimise the use of N inputs relative to N outputs in milk, especially fertiliser N, to decrease N surplus and increase NUE.

The fact that N surplus increased principally with fertiliser N input, but also with concentrate N input and, to a much lesser extent, with SR, suggests that decreasing fertiliser N and concentrate N inputs may be the most effective strategy to decrease N surplus. The weak impact of SR on N surplus would suggest that SR can be increased without considerably affecting N surplus. This has important implications in the context of achieving increased dairy production as is envisaged in the Food Harvest 2020 targets for Ireland (DAFM, 2013d), in that it suggests that, with good management, the SR increases that may be necessary on some farms to achieve these targets, may be achieved without increasing N surplus. While NUE decreased with increasing fertiliser N input, fertiliser N input explained only 0.42 of variation in NUE. The remainder could be attributed to farm-specific efficiency of N recycling and N losses between soil, pasture, animals, and milk and livestock for export (Nielsen and Kristensen, 2005) and other factors such as improved animal breeds (Ryan *et al.*, 2011), farmers' level of education, improved grass cultivars. A decrease in fertiliser N input combined with improved on-farm N recycling can increase NUE. Improved nutrient recycling on farms is one of the targets in the Food Harvest 2020 national strategy for sustainable growth of the agricultural sector (DAFM, 2013d).

Results suggest that a combination of decreased fertiliser N and concentrate N inputs and increased MS exports per cow can contribute to reduced surplus N per kg MS. However, this situation is difficult to achieve in a grazed grass-based production system because, all other factors being equal, increased feed intake is required to increase MS production per cow (Horan, 2009) and this is typically achieved through increased fertiliser N (to increase grass yields) and concentrate N inputs (Coleman *et al.*, 2010). However, increased MS production per cow may be achievable while minimising fertiliser and concentrate N use by optimising other management aspects such as grazing management, grass utilisation (O'Donovan *et al.*, 2002; Kennedy *et al.*, 2005), management of all on-farm nutrient sources (Peyraud and Delaby, 2006), and management of herd genetic potential (Berry *et al.*, 2007). On the other hand, an increase in MS production per cow can lead to increased N surplus per ha and potentially higher N losses.

3.4.2. Factors affecting N balances and use efficiencies across years

Nitrogen inputs and N surplus were greater and NUE was lower in 2010 compared to 2009 and 2011. The increased inputs were probably to support a SR that was 0.18 LU ha^{-1} greater than 2009 and 0.19 LU ha^{-1} greater than 2011 and were mainly in fertiliser N (mean of 0.81 of N input), being 49 kg N ha^{-1} greater than 2009 and 18 kg N ha^{-1} greater than 2011. The higher fertiliser N input in 2010 might also be partially due to lower mean temperatures between March and May in 2010 (8.5 ^{0}C) compared with 2009 (9.1 ^{0}C) and 2011 (9.6 ^{0}C) (Irish Meteorological Service, 2013), associated with poorer grass growth rates between March and May in 2010 (52.1 kg DM $ha^{-1}day^{-1}$) compared with 2009 (57.5 kg DM $ha^{-1}day^{-1}$) and 2011 (63.3 kg DM $ha^{-1}day^{-1}$) (Teagasc, 2013) so that additional N fertiliser may have been applied later in the year to compensate. These results highlight the necessity of assessing balances and use efficiencies in aggregate over a number of years, as results from a single year can reflect variability in weather and other factors.

The higher SR in 2010 was also associated with higher feed imports, both in kg per ha and in kg per LU, and with higher milk yields per cow, of 5,411 litres cow^{-1} in 2010 compared with 5,120 litres cow^{-1} in 2009 and 5,291 litres cow^{-1} in 2011. This equates to a response of 2.40 litres milk kg DM^{-1} of additional concentrate feed compared with 2009 and 0.69 litres milk kg DM^{-1} compared with 2011. A similar response in milk production, of 1.06 kg cow^{-1} per additional kg of imported concentrate feeds, was reported by Shalloo *et al.* (2004).

Despite increased output in milk and livestock in 2010, the increase in fertiliser N and concentrate N inputs resulted in an increase in N surplus (207 kg N ha^{-1}) of 32 % compared with 2009, and 15 % compared with 2011, a decrease in NUE, and also an increase in surplus N per kg MS. Others have found similar results (Treacy *et al.*, 2008; Humphreys *et al.*, 2008). The principal reason would appear to be reductions in the efficiency of N use associated with the increase in fertiliser N input.

3.4.3. Nitrogen balance and use efficiency before and after the GAP regulations

The results of the current study were compared with similar studies, completed between 2003 and 2006 (Treacy *et al.*, 2008) and in 1997 (Mounsey *et al.*, 1998), before the introduction of the GAP regulations, to investigate possible impacts of these Regulations on N balances and NUE on Irish dairy farms. The study of Treacy *et al.* (2008) was carried out on 21 intensive dairy farms, of which 8 were also involved in the current study, whereas the study of Mounsey *et al.* (1998) was on 12 intensive dairy farms. These intensive farms had SRs of 2.37 LU ha^{-1} (Treacy *et al.*, 2008) and 2.58 LU ha^{-1} (Mounsey *et al.*, 1998), respectively, compared with the national average SR of 1.85 LU ha^{-1} in 2005-2006 (Connolly *et al.*, 2006; 2007) and 1.47 LU ha^{-1} in 1997 (Fingleton, 1997) (Table 3.4.). Mean N surplus was significantly lower ($P < 0.001$) in the current study, at 175 kg N ha^{-1}, than Treacy *et al.* (2008) (227 kg N ha^{-1}) and Mounsey *et al.* (1998) (289 kg N ha^{-1}), while NUE was significantly higher ($P < 0.001$), at 0.23, compared to Treacy *et al.* (2008) (0.19) and Mounsey *et al.* (1998) (0.17) (Table 3.4.). Similarly, mean surplus N per kg MS was significantly lower ($P < 0.001$), at 0.28 kg N kg MS^{-1}, compared to Treacy *et al.* (2008) (0.37 kg N kg MS^{-1}) and Mounsey *et al.* (1998) (0.41 kg N kg MS^{-1}) (Table 3.4.). Results suggest a trend for decreased N surplus per ha and per kg MS and improved NUE on Irish dairy farms over the period covered by these studies (1997 to 2011) and following the introduction of the GAP regulations in 2006. This trend would have both agronomic and environmental benefits, indicating a move towards improved sustainability of dairy production, at least with regard to N. This demonstrates that is possible to improve both environmental and economic sustainability of dairy production through improved resource use efficiencies.

Table 3.4. Comparative mean values (and standard errors) for total utilised agricultural area, stocking rate, national average stocking rate, milk yield, milk protein and fat concentration, concentrate feed, imports of N in mineral fertilisers, concentrate feeds, forages, bedding material, and livestock, exports of N in milk and livestock, farm-gate N balances, N use efficiencies, and surplus N per kg milk solids on dairy farms before and after the implementation of Good Agricultural Practice regulations in Ireland; standard error of the means for transformed data in brackets; P-values from ANOVA are included

	Current study	Treacy *et al.* 2008	Mounsey *et al.* 1998	S.E.M.	*P* value
TUAA (ha)	71	59	65	3.27(0.02)	NS
Stocking rate (LU ha^{-1})	2.06	2.37	2.58	0.049	<0.001
National stocking rate (LU ha^{-1})	1.90	1.85	1.47	-	-
Milk yield (l cow^{-1})	5,308	5,167	5,588	65.4	NS
Milk protein (%)	3.4	3.4	3.3	0.01(0.001)	<0.001
Milk fat (%)	4.0	3.8	3.7	0.02(0.002)	<0.001
Concentrate fed (kg DM LU^{-1})	488	549	480	29.4	<0.05
N inputs (kg N ha^{-1})					
Mineral fertiliser	186	239	317	9.5	<0.001
Concentrate feed	26.6	43.6	32.8	2.30(0.02)	<0.01
Forage	12.4	0.0	0.0	-	-
Bedding material	4.0	0.0	0.0	-	-
Livestock	3.8	0.0	0.0	-	-
Total	228	283	350	10.8	<0.001
N outputs (kg N ha^{-1})					
Milk	40.2	43.6	52.2	1.60(0.01)	<0.05
Livestock	12.8	12.3	8.3	0.54	<0.01
Total	53.0	55.9	60.5	1.55	NS
N balance (kg N ha^{-1})	175	227	289	10.1	<0.001
N use efficiency	0.23	0.19	0.17	0.007(0.014)	<0.001
Surplus N kg kg MS^{-1}	0.28	0.37	0.41	0.001	<0.001

TUAA, total utilised agricultural area; LU, livestock units; l, litres; DM, dry matter; N, nitrogen; MS, milk solids; S.E.M., standard error of the means; NS, not significant.

There are a number of factors determining these differences between the three studies. The first factor was a significantly lower ($P < 0.001$) mean SR in the current study, of 2.06 LU ha^{-1}, in comparison with 2.37 LU ha^{-1} in Treacy *et al.* (2008) and 2.58 LU ha^{-1} in Mounsey *et al.* (1998). The lower SR in the current study had further impacts on fertiliser N, concentrate N inputs and milk N output.

The second factor was a significantly lower ($P < 0.001$) mean fertiliser N input, of 186 kg N ha^{-1}, in the current study, compared with 239 kg N ha^{-1} in Treacy *et al.* (2008) and 317 kg N ha^{-1} in Mounsey *et al.* (1998). While some of this decrease in fertiliser N input was doubtless associated with lower SRs, SR was 21 % lower in this study than in Mounsey *et al.* (1998) while fertiliser N input was 42 % lower, indicating that the decrease in fertiliser N input was not only associated with changes in SR. It would also seem likely that fertiliser N input decreased due to improved N management such as more appropriate rates and timing of application and better use of on-farm organic N fertilisers.

The third factor differing between the studies suggests that this was indeed the case, as 57 % of annual mineral N fertiliser was applied from February to May in the current study, compared with 59 % in Treacy *et al.* (2008) and 45 % applied mid-January in Mounsey *et al.* (1998). There was no application of mineral N fertiliser after September in the current study and in Treacy *et al.* (2008) while in Mounsey *et al*. (1998) mineral N fertilisers were applied up until the end of October. Also, 58 % of annual organic fertiliser N (farm yard manure and slurry) was applied between mid-January and April in the current study, compared with 55 % in Treacy *et al*. (2008) and 14 % in Mounsey *et al.* (1998). There was no application of organic fertilisers after October in the current study and in Treacy *et al*. (2008), whereas in Mounsey *et al.* (1998), 31 % was applied between November and January. This significant shift in the timing of organic N fertiliser application is consistent with advice on best practice indicating better fertiliser replacement value for spring application (Alexander *et al.*, 2008) and with the GAP regulations (European Communities, 2010), introduced in 2006, that prohibit application of organic fertilisers during the 'closed period', from mid-October to mid/end January. The concurrent decrease in mineral fertiliser N use and shift towards later application of this mineral fertiliser N both indicate an improved awareness of the fertiliser value of organic manures and accounting for them in nutrient management planning.

The fourth factor was the significantly lower ($P < 0.01$) concentrate N input per ha in the current study (26.6 kg N ha^{-1}) compared to Treacy *et al*. (2008) (43.6 kg N ha^{-1}) and Mounsey *et al.* (1998) (32.8 kg N ha^{-1}). While some of this decrease in concentrate N input was doubtless associated with lower SRs, SR was only 14 % lower in this study than in Treacy *et al.* (2008), while concentrate N input was 39 % lower. It would seem likely that concentrate N input also decreased due to improved feed management with increased grass and decreased concentrate feed per LU. Best practice in the seasonal grazed-grass-based production model, as would be advised by Teagasc (Irish state Agriculture and Food Development Authority), would be to minimise such feed inputs and maximise the proportion of grass in the diet (Dillon *et al.,* 1995; Horan, 2009).

Despite the decreases in fertiliser N and concentrate N inputs per ha, milk N output in the current study was only 3.4 kg N ha^{-1} lower than in Treacy *et al.* (2008) and 12 kg N ha^{-1} lower than in Mounsey *et al.* (1998). The 21 % lower SR compared to Mounsey *et al*. (1998) was matched by a 23 % lower milk N output per ha.

3.4.4. Nitrogen balance and use efficiency of Irish dairy farms in an international context

The results of the current study were compared with similar European studies completed after the implementation of the Nitrates Directive and with a study from New Zealand, as outlined in Table 3.5. In this comparison, the term 'continental European farms' refers to the Dutch farms in Groot *et al.* (2006) and Oenema *et al.* (2012), the Flemish farms in Nevens *et al.* (2006), and the French farms in Raison *et al.* (2006), while 'northern European farms' refers to the English and Irish farms in Raison *et al.* (2006), the Scottish farms in Roberts *et al.* (2007) and the English farms in Cherry *et al.* (2012).

Fertiliser N input in the current study (186 kg N ha^{-1}) was similar to the Dutch farms in Groot *et al.* (2006) (186 kg N ha^{-1}), lower than the English and Irish farms in Raison *et al.* (2006) (205 kg N ha^{-1}), the Flemish farms in Nevens *et al.* (2006) (257 kg N ha^{-1}) and the Scottish farms in Roberts *et al.* (2007) (301 kg N ha^{-1}), but higher than the French farms in Raison *et al.* (2006) (90 kg N ha^{-1}), the Dutch farms in Oenema *et al.* (2012) (142 kg N ha^{-1}), the English farms in Cherry *et al.* (2012) (172 kg N ha^{-1}) and the New Zealand farms in Beukes *et al.* (2012) (121 kg N ha^{-1}). .

Table 3.5. Comparative number of farms, type of system, grassland area, crop area and type of crop, stocking rate, milk yield, N input from mineral fertilisers, N balances, and N use efficiency in different regions

Reference	Region	No. Farms	Type of system	Grassland (proportion of TUAA)	Crop (proportion of TUAA)	SR (LU ha^{-1})	Milk yield (l ha^{-1})	Fertiliser N input (kg N ha^{-1})	N balance (kg N ha^{-1})	NUE
Current study	South of Ireland	21	G/C	0.93	0.07 (MS/W/T/K)	2.06	7,569	186	175	0.23
Groot *et al.* (2006)	The Netherlands	45	G/C	0.95	0.05(MS)	1.91	11,321	186	218	0.25
Nevens *et al.* (2006)	Flanders	120	G/C	0.64	0.36 (W/B/O)	3.00	9,906	257	295	0.19
Raison *et al.* (2006)	Scotland	10	G/C	0.94	0.06(MS)	1.60	7,155	114	134	0.26
	South of Ireland	24	G/C	1.00	0.00	2.10	7,757	269	240	0.20
	SW England	13	G/C	0.84	0.16(MS)	2.20	9,847	234	266	0.19
	Brittany	15	G/MS	0.70	0.30(MS)	1.40	5,315	57	117	0.39
	Pays de la Loire	13	G/MS	0.65	0.35(MS)	1.30	4,837	66	93	0.40
	Aquitaine	9	C/MS	0.39	0.61 (MS/MG)	1.20	6,053	147	155	0.35
	Basque country	16	0G	0.88	0.12(MS)	2.70	15,304	28	257	0.27
	Galicia	18	0G	0.58	0.42(MS)	3.00	19,723	136	349	0.24
	North Portugal	21	0G	0.00	1.00(MS)	6.10	34,760	212	502	0.33
Roberts *et al.* (2007)	Scotland	9	G/C	0.88	0.12(MS)	2.09	14,147	301	357	0.18
Cherry *et al.* (2012)	SW England	5	G/C	0.90	0.10(MS)	N/A	N/A	172	255	0.18
Oenema *et al.* (2012)	The Netherlands	16	G/C	0.76	0.24(MS)	1.89	15,860	142	191	0.34
Beukes *et al.* (2012)	New Zealand	247	G/C	0.94	0.06 (MS/B/O)	2.80	11,904	121	155	N/A

No., number; G/C, grazing-cutting; G/MS, grazing-maize for silage; C/MS, cutting-maize for silage; 0G, zero grazing; TUAA, total utilised agricultural area; MS, maize silage; W, wheat; B, barley; O, oat; K, kale; T, typhoon; MG, maize for grain; SR, stocking rate; LU, livestock units; l, litres; N, nitrogen; NUE, N use efficiency.

Concentrate N input in the current study (26.6 kg N ha^{-1}) was much lower compared with Nevens *et al.* (2006) (90 kg N ha^{-1}), Groot *et al.* (2006) (100 kg N ha^{-1}) and Raison *et al.* (the French farms) (2006) (59 kg N ha^{-1}). The main reason for higher concentrate N inputs in these studies was the high input system of dairy production that is more typical of dairy production in continental Europe, characterised by year-round milk production, high use of concentrates, imported feeds and forages, lower use of grazed grass and high milk yields per ha. In contrast, a low input system is more typical in Ireland, with seasonal milk production (compact spring calving), low use of concentrates, imported feeds and forages, high use of grazed grass and lower milk yields per ha. The continental European studies had much higher milk yields per ha (11,321 litres ha^{-1}, Groot *et al.*, 2006; 9,906 litres ha^{-1}, Nevens *et al.*, 2006), compared with the current Irish study (7,569 litres ha^{-1}). The French farms in Raison *et al.* (2006) had lower mean milk yields per ha (5,401 litres ha^{-1}) due to mixed agricultural production (milk, maize for export) on some of the farms. The higher milk yields per ha were also associated with higher mean milk N outputs per ha (73.6 kg N ha^{-1}, Groot *et al.*, 2006; 48.0 kg N ha^{-1}, Nevens *et al.*, 2006) compared with the current study (40 kg N ha^{-1}). On the French farms in Raison *et al.* (2006), the mean milk N output, of 29.0 kg N ha^{-1}, was lower than in the current study, likely due to their lower milk yields, SR and fertiliser N input.

In the study of Beukes *et al.* (2012), in New Zealand, the farms were considered to rely on home-grown low protein supplements (maize, barley and oat), with low imports of concentrate feeds. These farms had a mean concentrate feed import of 474 kg DM cow^{-1} and higher milk yields, of 11,904 litres ha^{-1}. These values were considered representative for the Waikato region in New Zealand. This indicates that dairy farmers in New Zealand operate milk production systems similar to the Irish, albeit with higher output per ha due to much higher SRs.

Despite the relatively low milk N output per ha, mean N surplus (175 kg N ha^{-1}) in the current study was lower than the mean N surplus reported by Groot *et al.* (2006) (218 kg N ha^{-1}), Raison *et al.* (the English and Irish farms) (2006) (213 kg N ha^{-1}), Nevens *et al.* (2006) (295 kg N ha^{-1}), Roberts *et al.* (2007) (357 kg N ha^{-1}), Cherry *et al.* (2012) (255 kg N ha^{-1}) and Oenema *et al.* (2012) (191 kg N ha^{-1}). This reflects the low input/ouput model of dairy production in Ireland. Mean N surplus in the current study was higher than Raison *et al.* (the French farms) (2006) (122 kg N ha^{-1}) and the New Zealand farms in Beukes *et al.* (2012) (155 kg N ha^{-1}). Mean NUE in the current study

(0.23) was higher than that reported by Nevens *et al.* (2006) (0.19), Raison *et al.* (the English and Irish farms) (2006) (0.21), Roberts *et al.* (2007) (0.18), and Cherry *et al.* (2012) (0.18), but lower than the mean NUE showed by Groot *et al.* (2006) (0.25), Raison *et al.* (the French farms) (2006) (0.38) and Oenema *et al.* (2012) (0.34). However, the overall mean NUE (0.24) for the continental and northern European farms was similar to mean NUE in the current Irish study (0.23).

The above values for N surplus and NUE in the continental and northern European studies represent the means for the period of study. However, in these studies deliberate efforts were made to improve N surplus and NUE and, as a result, N surplus decreased and NUE increased over time. It is notable that the Irish dairy farms in this study had an average fertiliser N input, N surplus and NUE, without intensive additional advisory and practice change efforts (beyond the usual advisory services and GAP regulations), that was within the range of the improved figures from the European studies following such advisory intervention. It is also worth noting that the dominance of fertiliser N on the input side of the Irish low input system means that efficient use of fertiliser N, and on-farm organic N sources, will play an even more important role in improving N balances and NUE.

It can be concluded that Irish dairy farms tend to operate with lower concentrate N inputs, relatively low fertiliser N inputs and lower N surpluses per ha than most other European dairy farms at lower output (litres milk ha^{-1}) and that this is largely due to the low input system that is more typical in Ireland with seasonal milk production (compact spring calving) (Buckley *et al.,* 2000), low use of concentrates, imported feeds and forages (Dillon *et al.*, 1995), high use of grazed grass (Horan, 2009), and relatively low milk yields per cow (Humphreys *et al.,* 2009a). All other factors being equal, one might expect less N losses to the environment under conditions of lower N surplus.

The dairy farms in New Zealand, that operate a grazed grass-based production system similar to Ireland, tend to operate with lower fertiliser N and concentrate N inputs and lower N surpluses than continental and northern European and Irish farms. On commercial dairy farms from 8 different locations in New Zealand, the mean N fertilisation rate was 137 kg N ha^{-1}, at a much higher mean SR (2.71 cows ha^{-1}) (Dalley and Geddes 2012; Dally and Gardner, 2012) than the continental and northern European studies, and the Irish farms in the current study. This may be due to the typically high white clover content in New Zealand pastures. Fixation by white clover is the main source of N input on New Zealand dairy farms (Ledgard *et al.,* 2001), fixing up to 300 kg N ha^{-1} (Ledgard *et al.*, 2009) and resulting in relatively low recommended N fertilisation rates of between 50 and 150 kg N ha^{-1} (Roberts and Morton, 2009). For comparison, the recommended N fertilisation rates for grazed pasture in Ireland range from 75 to 306 kg N ha^{-1}, with increasing SR from 1 to 2.4 LU ha^{-1} (Alexander *et al.*, 2008).

However, under experimental conditions, N fertilisation rates as low as 90 kg N ha^{-1} have been maintained with grass/clover grazed pastures stocked at 2 LU ha^{-1} (Humphreys *et al.*, 2008; 2009b; Keogh *et al.*, 2010). This compares very favourably with the 252 kg N ha^{-1} on fertilised grazed pastures stocked at 2.13 LU ha^{-1} in the same studies and indicates the potential for Irish dairy farms to reduce fertiliser N use and improve NUE through incorporation of clover in swards, while also increasing farm profitability through reduced fertiliser costs (Humphreys *et al.*, 2012). Moreover, the high protein content of grass-clover pastures can allow the greater use of low-protein home-grown supplements to dilute N intake without impairing milk production (Beukes *et al.*, 2012).

3.5. **Conclusions**

A survey of 21 Irish dairy farms from 2009 to 2011 found a mean N surplus of 175 kg ha^{-1}, or 0.28 kg N kg MS^{-1}, and a mean NUE of 0.23. Farm-gate N inputs were dominated by inorganic fertiliser (186 kg N ha^{-1}) and concentrates (26.6 kg N ha^{-1}), while outputs were dominated by milk (40.2 kg N ha^{-1}) and livestock (12.8 kg N ha^{-1}). Comparison to similar studies carried out before the introduction of the GAP regulations in 2006 would suggest that N surplus, both per ha and per kg MS, have significantly decreased (by 114 kg N ha^{-1} and 0.013 kg N kg MS^{-1}, respectively) and NUE increased (by 0.06) following the introduction of the GAP regulations. These improvements have mostly been achieved through decreased inorganic fertiliser N input and improvements in N management, with a notable shift towards spring application of organic manures, consistent with advice on best practice that indicate better fertiliser replacement value for spring application, and with the GAP regulations that prohibit application of organic fertilisers during the ‘closed period’ from mid-October to mid/end January. A concurrent decrease in mineral fertiliser N use and shift towards later application of this mineral fertiliser N both indicate an improved awareness of the fertiliser value of organic manures and accounting for them in nutrient management planning. These results would suggest a positive impact of the GAP regulations on dairy farm N surplus and NUE.

Taking surplus N per ha as an indicator of local environmental pressure, this indicates that the environmental sustainability of milk production has improved. The improvement in NUE also indicates that agronomic performance has improved concurrently. This demonstrates that it is possible to improve both environmental and economic sustainability of dairy production through improved resource use efficiencies. Such improvements will be necessary to achieve national targets of improved water quality under the EU Water Framework Directive, and increased dairy production, as set out in the Food Harvest 2020 Report. The weak impact of SR on N surplus found in this study would suggest that, with good management, the increases in SR and milk output per ha that may be necessary on some farms to achieve these production targets, may be achieved while decreasing N surplus per ha. The dominance of fertiliser N on the input side of the Irish low input dairy production system means that efficient use of fertiliser N, and other on-farm N sources, plays an even more important role in determining N balances and NUE and will, therefore, play a central role in improving N balances and NUE. These improvements may be achieved through optimising management aspects such as nutrient management planning, grazing management and grass utilisation, and use of clover in swards, for example.

Mean N surplus (175 kg N ha^{-1}) was lower than the overall mean surplus (224 kg N ha^{-1}) from six studies of northern and continental European dairy farms, while mean NUE was similar. It can be concluded that Irish dairy production systems, on average, tend to operate with lower concentrate N inputs, relatively low fertiliser N and lower N surpluses than other European dairy production systems and that this is largely due to the low input system that is more typical in Ireland, with seasonal milk production (compact spring calving), low use of concentrates, imported feed and forages, high use of grazed grass and lower milk yields per ha. All other factors being equal, one might expect less N losses to the environment under these conditions of lower N surplus.

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4. Phosphorus balance and use efficiency on twenty-one intensive grass-based dairy farms in the South of Ireland

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Summary

Given the finite nature of global phosphorus (P) resources, there is an increasing concern about balancing agronomic and environmental gains from phosphorus usage on dairy farms. Data from a 3 year (2009-2011) survey were used to assess farm-gate P balances and P use efficiency (PUE) on 21 intensive grass-based dairy farms operating under the Good Agricultural Practice (GAP) regulations in Ireland. Mean stocking rate (SR) was 2.06 LU ha^{-1}, mean P surplus was 5.09 kg ha^{-1}, or 0.004 kg P kg MS^{-1}, and mean PUE was 0.70. Phosphorus imports were dominated by inorganic fertiliser (7.61 kg P ha^{-1}) and feeds (7.62 kg P ha^{-1}), while exports were dominated by milk (6.66 kg P ha^{-1}) and livestock (5.10 kg P ha^{-1}). Comparison to similar studies carried out before the introduction of the GAP regulations in 2006 would suggest that P surplus, both per ha and per kg MS, have significantly decreased (by 14.41 kg P ha^{-1} and 0.017 kg P kg MS^{-1}, respectively) and PUE increased (by 0.33), mostly due to decreased inorganic fertiliser P import and improvements in P management, with a notable shift towards spring application of organic manures, indicating improved awareness of the fertiliser value of organic manures and good compliance with the GAP regulations regarding fertiliser application timing. These results would suggest a positive impact of the GAP regulations on dairy farm P surplus and PUE, indicating an improvement in both environmental and economic sustainability of dairy production through improved resource use efficiencies. Such improvements will be necessary to achieve national targets of improved water quality and increased dairy production. Results suggest that optimising fertiliser and feed P imports combined with improved on-farm P recycling may be the most effective way to increase PUE. Equally, continued monitoring of STP (soil test P) and P management will be necessary to ensure that adequate soil P fertility is maintained. Mean P surplus was lower and PUE was much higher than the overall mean surplus (15.92 kg P ha^{-1}) and PUE (0.47) from three studies of continental and English dairy farms, largely due to the low import system that is more typical in Ireland, with seasonal milk production (compact spring calving), low use of imported feeds and high use of grazed grass.

4. 1. Introduction

Given the finite nature of global phosphorus (P) resources and the need to reduce P losses to the environment (Simpson *et al.,* 2011; Huhtanen *et al.,* 2011; Cordell *et al.,* 2011), there is great concern for efficient P use in intensive farming systems. Irish dairy production systems tend to be relatively intensively managed compared to other Irish grassland agricultural production systems and are pasture-based, with the objective of producing milk in a low cost system through maximising the proportion of grazed grass in the cows' diet (Shalloo *et al*., 2004; McCarthy *et al.,* 2007; Ryan *et al.,* 2011). Increasing the proportion of grazed grass reduces milk production costs and can increase the profitability of grass based milk production systems in Ireland and other temperate regions (Dillon *et al.,* 2005; Dillon, 2011). Phosphorus imports, in the form of concentrate feeds and fertilisers, are key drivers of increased herbage yields and saleable milk export on most dairy farms (Aarts, 2003; Spears *et al.* 2003; Gourley *et al*., 2012). More precisely, mineral P fertilisers contribute to increases in herbage yield to the extent to which they supply P in a readily available form for plant uptake, which enhances root development (Lynch and Caffrey, 1997) and photosynthesis (Alexander *et al.*, 2008). These improved processes positively impact on overall development of grass plants and, therefore, herbage yields. However, P imports typically exceed P exports in milk and livestock exported off the farms (Van Keulen *et al.*, 2000). This imbalance results in surplus P that is either accumulated in soil or lost from the dairy farms (Arriaga *et al*., 2009; Gourley *et al*., 2010).

Farm-gate P surplus is commonly used as an environmental indicator for the risk of P losses to the environment (Swensson, 2003; Huhtanen *et al.,* 2011; Weaver and Wong, 2011). Even if surplus P does not predict the actual losses and loss pathways, it is a long-term risk indicator of P losses (Jarvis and Aarts, 2000). However, unlike N surpluses which are seen, necessarily, as an economic waste and potential environmental problem, P surpluses may be necessary, for a period of time, on farms where an increase in soil P content is required to achieve agronomic optimal soil P (Culleton *et al.,* 1999) without posing a risk to the environment, if managed correctly. Surplus P potentially accumulates in the soil (Gourley *et al.,* 2010), building soil fertility, or is lost in eroded material containing particulate P or P adsorbed on to organic-rich clay soil fractions (Kurz *et al*., 2005) or in soluble forms through leaching (Heathwaite, 1997) or runoff. Grass-based farms can be sources of diffuse P losses (Kiely *et al.,* 2007), because, by fertilising grassland with mineral and organic fertilisers, high concentrations of potentially mobile P (PMP) are placed at or near the soil surface, where it may be susceptible to mobilisation and transport to water bodies (Herlihy *et al*., 2004). These P losses can have negative environmental impacts such as eutrophication of surface waters (Clenaghan *et al.*, 2005), and pollution of groundwater aquifers (Heathwaite, 1997). In Ireland, phosphorus is the major limiting nutrient in surface fresh waters and increased additions may result in algal blooming (McGarrigle, 2009). Losses of P also incur economic costs in two ways; the cost of wasted N and P inputs, at farm level, and the cost of clean-up associated with pollution caused as a result of such losses, more typically at regional to national levels (Buckley and Carney, 2013). It has been emphasised that dairy production should ideally be achieved in a sustainable manner, without impairing natural capital (soils, water, and biodiversity) (Goodland, 1997). Therefore, in the current study, P surplus, as an indicator of potential for P losses, which can be associated with environmental and economic implications, is referred to as an indicator of environmental and economic (farms' ability to generate sufficient funds to sustain their production potential in the long run; European Commission, 2001) sustainability.

Nutrient use efficiencies indicate farms' resource use and related management decisions, therefore being considered more as an indicator of farms' agronomic performance (Halberg, 1999; Oenema *et al.*, 2003; Gourley *et al.*, 2012). However, due to the potential economic implications of P that is not used on farms (Buckley and Carney, 2013), in the current study, PUE is also considered as an indicator of economic sustainability, along with P surplus. Hence, improved nutrient use efficiency has a significant role to play in the development of more sustainable dairy production systems (Goulding *et al.*, 2008). The P use efficiency (PUE; proportion of P imports recovered in agricultural exports (Aarts, 2003)) in dairy production systems is highly variable. For example, in Europe, PUE values of between 0.37 and 0.85 have been recorded (Mounsey *et al.*, 1998; Van Keulen *et al*., 2000; Steinshamn *et al*., 2004; Nielsen and Kristensen, 2005; Raison *et al.*, 2006; Huhtanen *et al*., 2011).

Irish dairy production systems benefit from mild winters (5.1 ^{0}C in January) and annual rainfall between 800 and 1,200 mm, allowing grass growth all year around and an extended grazing season that can be as long as February to November (Humphreys *et al.,* 2009), varying with location and soil type. Irish dairy farms are unique in Europe in that the majority operate a seasonal milk production system with compact spring calving (from January to April) so that milk production matches grass growth. The proportion of grazed grass in the diet of dairy stock is hence maximised (Humphreys *et al.*, 2009), allowing for the maximum amount of milk to be produced from grazed grass and reducing requirements for feeding concentrate feeds post-calving (Dillon *et al.*, 1995). For these reasons, the potential for more effective use of P on-farm and management strategies to achieve improved PUE may be expected to differ from those of the year-round feed-based dairy production systems more typical of continental Europe and Britain. In grass-based dairy production systems, there are a number of factors affecting PUE, such as soil P-sorption capacity in relation to soil P inputs, uneven dispersal of excreta leading to uneven soil P content (in grazing enterprises), the ability of grass plants to convert P from applied mineral P fertiliser and manure into biomass in herbage, utilisation by animals of grass herbage grown and the biological potential of cows to convert P from concentrate feeds and herbage into milk (Gourley *et al.,* 2010). More effective use of P imports in concentrate feeds and fertiliser P, and soil P resources, can potentially contribute to decreased imports and increased PUE (Nielsen and Kristensen, 2005; Huhtanen *et al.,* 2011).

The on-going debate over P supply and demand together with the concern for water quality affected by P lost from agricultural land supports the need to ensure that P is used efficiently on farms (Pieterse *et al*.,, 2003; Syers *et al.* 2008; Weaver and Wong, 2011; Simpson *et al.,* 2011). In the EU, the Water Framework Directive (WFD) (2000/60/EC) was introduced with the objective of protecting and improving groundwater and surface water bodies' quality. In Ireland, the WFD was first implemented as the Water Policy Regulations (European Communities, 2003), in 2003. To ensure water quality, these regulations established a concentration limit of 0.03 mg Molybdate Reactive Phosphorus (MRP) litre^{-1} or 35 µg PO_4 litre^{-1} (European Communities, 2009). Additionally, the Nitrates Directive (91/676/EEC) (European Council, 1991) has established guidelines in relation to farming practices to reduce nitrate (NO_3) leaching that are implemented in each member state through a National Action Programme (NAP). In Ireland, these are legislated as the Good Agricultural Practice (GAP) Regulations (European Communities, 2010), first passed in 2006. The GAP Regulations establish farming practices to reduce nitrate (NO_3) leaching but also limit P use on farms and establish soil P indices. Under the Regulations, farms are limited to a stocking rate (SR) of 170 kg organic N ha^{-1}, equivalent to 2 livestock units (LU) ha^{-1}, or 2 dairy cows ha^{-1}. The Regulations also establish the quantity of available P that can be applied to grass and other crops (depending on factors such as SR, soil test P (STP) and crop type), the volume of slurry storage required (depending on factors such as location, local rainfall, and stock type and number), closed periods in winter months during which spreading of organic and inorganic fertilisers is restricted (depending on location in the country) and other restrictions on spreading based on soil conditions, topography, weather and distance to water features.

The GAP Regulations established a P index system for grassland soils based on soil test P (STP). Index 1 (0.0-3.0 mg P litre $(l)^{-1}$) and 2 (3.1-5.0 mg P litre^{-1}) soils are considered deficient in P and require a build-up of soil P to reach agronomic optimum. The target index is 3 (5.1-8.0 mg P litre^{-1}), at which the soil is considered to have optimum P to meet crop demand without having negative impacts on the environment (Ryan and Finn, 1976; Herlihy *et al.*, 2004; Power *et al.*, 2005). Soils within index 4 (>8 mg P litre^{-1}), with high P status, are considered in excess of agronomic optimum and at greater risk of P loss to water. The new index system involved the lowering of the upper limit for index 2 from 6 to 5 mg P litre^{-1}, and the upper limit for index 3 from 10 to 8 mg P litre^{-1} than was previously advised for grassland soils. The aim was to reduce P losses from grassland while maintaining agricultural production (Treacy, 2008). Soil P status is assessed every five years on Irish farms (European Communities, 2010). For SRs up to 2 LU ha^{-1}, the maximum allowed P fertiliser application ranges between 39 kg ha^{-1} for soils in index 1 to 0 kg ha^{-1} for soils in index 4 (European Communities, 2010).

The GAP measures are intended to increase PUE and retention of N and P within the production systems and minimise losses from farms to water. However, most of the existing data on dairy farm P balances in Ireland date from the period before the implementation of the Regulations in 2006 (Mounsey *et al.*, 1998; Treacy, 2008). There is no study on farm-gate P balance on Irish dairy production systems after the implementation of GAP regulations. In the European context also, there are very few farm-gate P balances on grassland-based dairy farms (e. g. Van Keulen *et al*., 2000; Aarts, 2003; Swensson, 2003; Nielsen and Kristensen, 2005; Raison *et al.*, 2006; Gamer and Zeddies, 2006). Steinshamn *et al*. (2004) and Huhtanen *et al*. (2011) examined P balances and use efficiencies in dairy production systems but these were based on modelling and experimental studies.

Therefore, the objectives of the current study were: (i) to assess farm-gate P balances and use efficiencies on 21 commercial intensive dairy farms operating under the GAP Regulations in Ireland and compare these to pre-Regulations studies to investigate the impact of the Regulations; (ii) to identify the factors influencing PUE on these farms; (iii) to explore potential approaches to increase PUE and decrease P surpluses on these farms. For this purpose, data on P imports and exports were recorded on 21 dairy farms participating in the INTERREG-funded DAIRYMAN project over three years, from 2009 to 2011.

4. 2. Materials and Methods

4.2.1. Farm selection and data collection

Twenty-one commercial intensive dairy farms were selected, located in the South of Ireland, in counties Cork, Limerick, Waterford, Tipperary, Kilkenny and Wicklow. These farms were pilot farms involved in the INTERREG-funded DAIRYMAN project (www.interregdairyman.eu) focusing on improving resource use efficiency on dairy farms in Northwest Europe. Farm selection was based on the likely accuracy of data recording, 8 of the farms in the current study having been involved in a previous similar study (GREENDAIRY; Treacy, 2008), and all the farmers being willing to provide data. The selected farms were known as being progressive in their approach to farm management and, therefore, may not be fully representative of all Irish dairy farms. However, the farm area, stocking rate and milk yield per cow showed that the participating farms were close to, but slightly above, the national average for dairy farms. Grass-based milk production from spring calving cows was the main enterprise on all the selected farms.

Key farm characteristics are given in Table 4.1. Mean total utilised agricultural area (TUAA) was 71 (S.D. = 24.8) ha, mean SR was 2.06 (S.D. = 0.32) LU ha^{-1}, and mean milk yield was 5,308 (S.D. = 464) litres cow^{-1} between 2009 and 2011. For comparison, national mean values for dairy farms were 52 ha for TUAA, 1.90 LU ha^{-1} for SR, and 4,956 litres cow^{-1} for milk yield, during the same timeframe (Connolly *et al.*, 2009; Hennessy *et al.*, 2010; 2011). Seventeen of the farms in the current study participated in the Rural Environment Protection Scheme (REPS). This is a program co-funded by the EU and the Irish government whereby farmers are rewarded financially for operating to a set of guidelines consistent with an agri-environmental plan drawn up by an approved planning agency (DAFM, 2013b). Important conditions for receiving REPS financial support were to limit SR to 2 LU ha^{-1} and to apply mineral fertilisers to the farming area according to fertiliser plans drawn up for their farms (DAFM, 2013b). However, the 17 farms closely adhering to GAP regulations were not fully representative of the Irish dairy farms and this may bias the interpretation of the results of the current study.

Table 4.1. Total utilised agricultural area (and crop area), annual air temperature, annual rainfall, soil test phosphorus, pH, stocking rate, milk yields, concentrate feeds, and estimated harvested grass through grazing and silage; soil type for 21 Irish dairy farms between 2009 and 2011

Farm	TUAA (crops) (ha)	Temp. (^{0}C)	Rainfall (mm year^{-1})	Soil type	STP (mg litre^{-1})	pH	SR (LU ha^{-1})	Milk yield (l cow^{-1})	Conc (kg DM LU^{-1})	Grass (kg DM LU^{-1})
1	85	9.6	1,077	CL	6.42	5.89	2.15	5,319	268	4,139
2	67	9.8	1,124	C	4.49	6.43	2.41	6,010	499	4,169
3	73	9.8	1,124	C	8.99	6.47	2.07	5,688	221	4,304
4	50	10.1	1,373	L	6.50	6.49	2.68	5,309	571	3,691
5	74 (1.2)	10.1	1,373	L	6.50	5.65	1.82	5,149	611	3,891
6	63 (3.9)	10.1	1,373	L	3.36	5.29	1.92	5,672	568	3,632
7	47	9.6	1,077	L	2.29	5.64	2.41	5,080	471	3,922
8	58	10.1	1,373	C	6.61	5.94	2.50	5,671	580	4,033
9	51	9.6	1,077	C	5.84	5.91	2.01	5,431	466	4,089
10	130 (5.5)	10.1	1,373	L	6.50	5.65	1.97	5,207	394	3,898
11	40	10.1	1,373	L	3.79	5.32	2.39	4,229	615	3,508
12	52	10.1	1,373	L	7.71	6.03	1.77	5,613	604	3,886
13	81	9.6	1,077	C	7.95	5.77	1.84	5,290	710	3,730
14	96 (6.7)	9.8	1,124	SL	4.99	5.97	1.80	4,415	302	3,472
15	128	9.8	1,124	L	4.54	6.17	1.88	4,671	484	3,858
16	78 (13.4)	10.2	1,453	C	6.65	6.49	1.58	6,038	801	3,746
17	72	9.6	1,077	C	5.81	6.18	2.47	4,928	463	4,002
18	48	9.8	1,124	CL	3.55	5.95	1.92	5,549	732	3,567
19	71 (2.3)	9.8	1,124	C	7.24	6.22	2.22	5,500	251	2,919
20	76 (6.2)	10.1	1,373	SL	8.51	5.78	1.97	5,174	265	4,011
21	48 (1.6)	10.1	1,373	L	2.80	5.56	1.40	5,522	386	4,108
Mean	71 (5.6)	9.9	1,235	-	5.76	5.94	2.06	5,308	488	3,837
S.D.	24.8 (3.91)	0.22	145	-	1.89	0.35	0.32	464	166	309

TUAA, total utilised agricultural area; temp., temperature; CL, clay-loam; L, loam; C, clay; SL, sandy-loam; STP, soil test phosphorus; SR, stocking rate; LU, livestock units; l, litres; conc., concentrate feeds; DM, dry matter; S.D., standard deviation.

On the selected farms, data were collected on a monthly basis between 2010 and 2011 and included grassland area, area under crops, type of crops and percentage of crops fed to livestock, livestock numbers and type of livestock, number of days spent grazing, and imports of manure, concentrate feeds, bedding material, silage, mineral P fertilisers and other agro-chemicals, as well as exports of milk, manure, crops and silage. For mineral P fertilisers, amounts imported onto farms as well as the amounts applied to land were recorded on a monthly basis. For year 2009, similar data were obtained from farm records and farm advisors. Data collected for the 3 years were cross-checked with secondary data sources such as Single Farm Payment forms (data forms required from farmers for participation in state schemes) (DAFM, 2013a). Data on livestock imports and exports were extracted from the Dairy Management Information System (DAIRYMIS) (Crosse, 1991). Values for amounts of milk sold off the farms were extracted from the reports on milk deliveries coming from the cooperatives supplied by the farmers. Data on soil types were extracted from REPS forms for the participating farms and from the national soil survey (Gardiner and Radford, 1980) for the remainder. Data on mean annual rainfall and temperature were extracted from an Irish Meteorological Service database for different weather stations located in, or close to, the area of study, at Cork airport, Roche's point, Gurteen, Johnstown Castle and Oak Park (Irish Meteorological Service, 2013).

The annual amount of pasture harvested and utilised on-farm through grazing and silage on each farm was modelled using the Grass Calculator (Teagasc, 2011) based on the difference between the net energy (NE) provided by imported feeds (concentrates and forages) and the net energy requirements of animals for maintenance, milk production, and body weight change (Jarrige, 1989). It was assumed that 1 kg dry matter (DM) of grass equals 1 feed unit for lactation (UFL).

Stocking rate was expressed as LU per ha for TUAA. One dairy cow was considered equivalent to 1 LU and 1 bovine less than 1 year old equivalent to 0.3 LU (Connolly *et al*., 2009).

4.2.2. Soil sampling and analysis

Eleven soil samples, on average, were taken per farm on one occasion during the study period, the farmers being required to sample their farms at least once every five years (European Communities, 2010). Samples were taken using a standard soil corer (50 mm diameter), sampling to a depth of 100 mm. Each sample area was not greater than 4 ha, with sample areas evenly distributed across each of the farms. The sample areas were also carefully selected to ensure areas used for grazing and silage production were both represented. At least 50 soil cores were taken from each sample area, in a zigzag pattern. Care was taken to avoid unusual spots in the sample area, such as old fences, ditches, and around gateways and feed troughs (Treacy, 2008). Each sample was carefully mixed, before smaller representative bulked samples were extracted and sent for analysis to Teagasc Johnstown Castle Research Centre. Samples were analysed for soil pH and Morgan's Soil P concentrations using the standard laboratory procedures for Ireland, as described by Byrne (1979). Soil samples were dried for 16 hours at 40 ^{0}C in a forced draught oven with moisture extraction. Soil pH was determined by mixing 10 ml of dried sieved (2 mm) soil with 20 ml of H_2O and, after being allowed to stand for ten minutes, measuring the pH of the suspension using a digital pH meter with glass and calomel electrodes. For soil P concentrations, soil samples were extracted in a one part soil to five parts solution ratio with a 10 % sodium acetate solution buffered at pH 4.8 (Morgan's solution). Six millilitres (ml) of dried soil were extracted with 30 ml of Morgan's solution using a Brunswick Gyratory shaker for 30 minutes at constant temperature (20 ^{0}C). The suspension was then filtered using No. 2 Whatman filter paper. Analysis for P content was then carried out on the clear extract by spectrophotometry (Treacy, 2008). The same sampling procedure and soil analyses were used for two similar previous studies (Treacy, 2008; Mounsey *et al.*, 1998), which the current study was compared to.

4.2.3. Farm-gate phosphorus imports, exports, balances and use efficiencies

Phosphorus imports and exports were calculated both on a monthly and an annual basis. Phosphorus in mineral fertiliser was calculated by taking into account the P content of fertilisers applied to land. Monthly imported amounts of concentrate feeds and forages were assumed to be exhausted in the end of each month. Due to the fact that P content of imported concentrates and forages onto farms was not directly measured, it was assumed to be 5 kg P per tonne of concentrate and forage (European Communities, 2010).

Phosphorus in livestock imported on, or exported off, the farms was calculated by using standard values for live weight (Treacy, 2008) and multiplying it by 0.01 (McDonald *et al.*, 1995). Phosphorus in exported milk was calculated by considering a P content of 0.0009 kg P per kg of milk (McDonald *et al.*, 1995).

The farm-gate P balance was calculated as the difference between total P import and total P export (Weaver and Wong, 2011) and was expressed on both an areal basis (kg P ha^{-1}) and a unit product basis (kg P kg milk $solids^{-1}$ (MS)) (Fangueiro *et al*., 2008) for years 2009-2011. Phosphorus use efficiency was calculated as the ratio between total P export and total P import, expressed as a proportion (Huhtanen *et al*., 2011) for years 2009-2011.

The same principles for calculating P inputs, outputs, balances and PUE were followed in two similar previous studies (Treacy, 2008; Mounsey *et al.*, 1998), which the current study was compared to.

4.2.4. Statistical analysis

Descriptive statistics were applied using SPSS to calculate means and standard errors (Darren and Mallery, 2008). Normal distribution of residuals was tested using Shapiro-Wilk, with values lower than 0.05 indicating a non-normal distribution. The log transformation was required to ensure homogeneity of variance (Tunney *et al*., 2010) for some of the variables. Therefore, TUAA, milk fat and protein concentration, P imports per ha from fertiliser P, feeds and livestock, total P import, milk P export, P balance per ha and per kg MS, PUE, P imports per kg MS from fertiliser P and feeds, MS exports per cow, comparative STP values, P imports from fertilisers and feeds, P exports in sold milk, P balance per ha and per kg MS, and PUE in the current study and the studies of Treacy (2008) and Mounsey *et al.* (1998) were transformed using a log10 base (y=log10(x)).

Differences in mean STP, TUAA, SR, milk yields, milk protein and fat concentration, concentrate feed imports, P imports, P exports, P balance per ha and per kg MS, and PUE between years and farms were analysed using repeated measures ANOVA. A significance level of 0.05 or less (0.01 and 0.001) indicated statistically significant differences among the means. A significance level of 0.05 or higher indicated a 95 or higher percent of certainty that the differences among the means are not the result of random chance (Darren and Mallery, 2008). Such results were presented as not significant (NS).

The statistical models included farm and year effects on each of the tested variables. The 21 farms were considered as replicates. The models used were:

1. $Y_i = \mu + a_i + e_{i,}$ where Y_i = tested variable, a_i = the effect of *i*th farm (i = 1,….,21), and e_i = the residual error term, and;
2. $Y_i = \mu + b_j + e_{i,}$ where Y_i = tested variable, b_j = the effect of *j*th year (j = 2009, 2010, 2011), and e_i = the residual error term.

Multiple stepwise linear regression was undertaken to investigate relationships between key dependent and independent variables presented in Table 4.2. The choice of the statistical models was dependent on the potential significance of independent variables and their potential impact on the dependent variables. Non-significant ($P > 0.05$) independent variables were automatically removed from the models (Table 4.2.). The probability for acceptance of new terms (F) was 0.10 (Groot *et al.*, 2006) and the confidence interval was 0.95. All relationships between variables were assessed for outliers, normality and colinearity. The identified outliers were diminished through log transformation.

Uncertainty analysis was carried out by calculating the coefficient of variation as the ratio between standard deviation and mean values (Gourley *et al.,* 2010) for each P import, P export, P balance and PUE on the 21 farms between 2009 and 2011, expressed as a proportion.

Table 4.2. Investigated and significant multiple stepwise linear regression models

Investigated	Significant
LgFrtP = μ + βLgTUAA + βSTP + βSR + βMSE + βGD + σ_{est}	LgFrtP = μ - STP + σ_{est}
LgFdP = μ + βSR + βMSE+ βGD + σ_{est}	LgFdP = μ + βSR - βGD + σ_{est}
LgMP = μ + βSR + βMSE + βGD + βLgFrtP + βLgFdP + σ_{est}	LgMP = μ + SR + σ_{est}
LP = μ + βSR + βGD + βLgFrtP + βLgFdP + σ_{est}	NS
LgPbal = μ + βSTP + βSR + βMSE + βGD+ βLgFrtP + βLgFdP + σ_{est}	NS
LgPUE = μ + βSR + βMSE + βGD + βLgFrtP + βLgFdP + σ_{est}	LgPUE = μ - βLgFrtP - βLgFdP + σ_{est}
LgPMS = μ + βLgMS + βGD + βLgFrtPMS + βLgFdMS + σ_{est}	LgPMS = μ - βLgMS + σ_{est}

LgFrtP, log transformed mineral fertiliser P applied to land; LgFdP, log transformed feeds phosphorus (P) import; LgMP, log transformed milk P export, LP, livestock P export, LgPbal, log transformed P balance per ha, LgPUE, log transformed P use efficiency, LgPMS, log transformed surplus P per kg milk solids; LgTUAA, log transformed total utilised agricultural area; STP, soil test P; SR, stocking rate; MSE, milk solids export per ha; GD, number of grazing days; LgMS, log transformed milk solids export per cow; LgFrtPMS, log transformed mineral fertiliser P applied to land per kg milk solids; LgFdMS, log transformed feeds P import per kg milk solids; β = standardized coefficient of regression, σ_{est}, standard error of the estimate; NS, not significant.

4.3. Results

4.3.1. Phosphorus imports

There was a high degree of variation in mean P imports between years and farms (Table 4.3.). Mean total P import was 16.85 kg P ha^{-1} (Table 4.3.). There were significant differences in mean total P import between farms, ranging from 3.64 to 26.94 kg ha^{-1} over the three years (Table 4.3.). The coefficient of variation for mean total P import between farms was 0.39 over the 3 years. There were also significant differences in mean total P import between years, ranging from 15.21 to 19.99 kg ha^{-1} (Table 4.3.). The main sources of P import onto farms were imported feeds and mineral fertilisers, accounting for around 0.50, each, of total P import. Mean P import from feeds was 7.62 kg P ha^{-1} (Table 4.3.). There were no significant differences in mean P import from feeds between farms (Table 4.3.). There were significant differences in mean P import from feeds between years, ranging from 4.69 to 11.13 kg ha^{-1} (Table 4.3.). Mean fertiliser P import was 7.61 kg P ha^{-1} (Table 4.3). There were significant differences in mean fertiliser P import between farms, ranging from 1.69 to 20.15 kg ha^{-1} over the three years (Table 4.3.). The coefficient of variation for mean fertiliser P import between farms was 0.64 over the 3 years. There were no significant differences in mean fertiliser P import between years (Table 4.3.). On a monthly basis, mean mineral fertiliser P applied to land was the highest between April and June, at 2.83 (S.D. = 3.14) kg P ha^{-1} (Fig. 4.1.).

Table 4.3. Mean values (and standard errors), grand means between years and ranges between farms for mineral P fertilisers applied to land, P imports in feed stuffs and livestock, P exports in sold milk and livestock, farm-gate P balances, P use efficiencies per ha and P balance per kg milk solids for 21 Irish dairy farms between 2009 and 2011; standard error of the means for transformed data in brackets; P-values from ANOVA are included.

	Year			Grand mean	S.E.M.	Range farms	*P*-value	
	2009	2010	2011				Y	F
P imports (kg P ha^{-1})								
Mineral fertiliser applied	8.43	7.91	6.50	7.61	0.783(0.054)	1.69-20.15	NS	<0.01
Feeds	4.69	11.13	7.04	7.62	0.602(0.033)	2.52-13.44	<0.001	NS
Livestock	2.24	0.95	1.67	1.61	0.134(0.041)	0.06-4.62	NS	NS
Total	15.36	19.99	15.21	16.85	1.040(0.032)	3.64-26.94	<0.01	<0.05
P exports (kg P ha^{-1})								
Milk	6.22	7.22	6.56	6.66	0.204(0.013)	4.27-9.52	NS	<0.001
Livestock	4.46	5.52	5.32	5.10	0.277	2.63-9.43	NS	<0.01
Total	10.68	12.74	11.88	11.76	0.412	7.44-17.45	NS	<0.001
P balance (kg P ha^{-1})	4.68	7.25	3.33	5.09	1.073(0.067)	-7.42 - +19.48	<0.05	0.01
P use efficiency	0.69	0.63	0.78	0.70	0.096(0.034)	0.30-1.58	NS	0.01
P balance (kg kg MS^{-1})	0.0004	0.011	0.003	0.004	0.001(0.062)	-0.01 - +0.03	NS	NS

P, phosphorus; MS, milk solids; S.E.M., standard error of the means; Y, year; F, farm; NS, not significant.

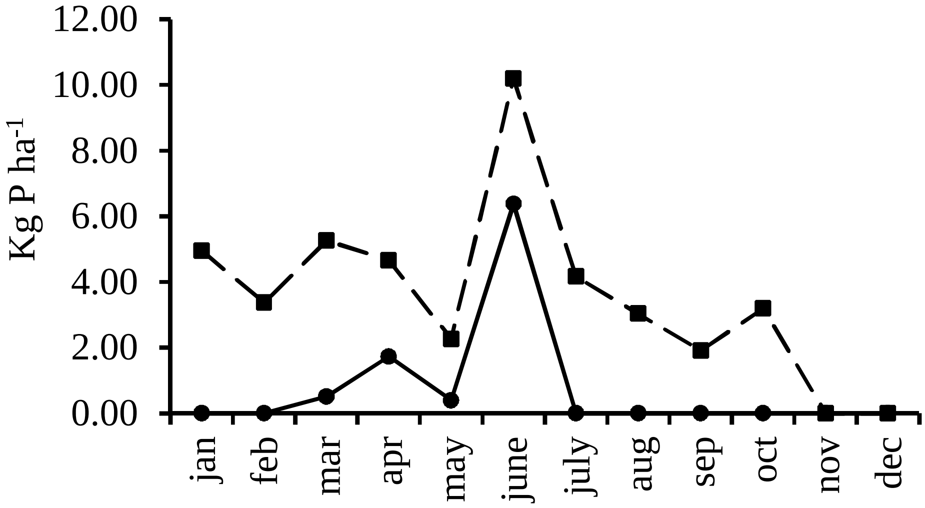

Fig. 4. 1. Monthly application rates of mineral **(____)** and organic (- -- **-)** P fertilisers (kg P ha^{-1}) on 1 Irish dairy farms between 2009 and 2011

There was a significant negative relationship (R^2 = 0.21; P < 0.05) between mean log transformed mineral fertiliser P applied to land and STP (β = -0.46). An increase of 0.34 mg litre^{-1} in mean STP was associated with a decrease of 0.03 (0.92, not transformed) kg ha^{-1} in mean log transformed mineral fertiliser P applied to land.

There was a significant relationship (R^2 = 0.20; P < 0.01) between mean log transformed feed P import and mean SR (β = 0.34) and mean number of days spent grazing (β = -0.24). An increase of 0.07 LU ha^{-1} in mean SR was associated with an increase of 0.02 (0.55, not transformed) kg ha^{-1} in mean log transformed feed P import. An increase of 2.20 days per year in mean number of days spent grazing was associated with a decrease of 0.02 (0.55, not transformed) kg ha^{-1} in mean log transformed feed P import.

4.3.2. Phosphorus exports

There was a high degree of variation in mean P exports between farms (Table 4.3.). Mean total P export was 11.76 kg P ha^{-1} (Table 4.3.). There were significant differences in mean total P export between farms, ranging from 7.44 to 17.45 kg ha^{-1} over the 3 years (Table 4.3.). The coefficient of variation for mean total P export between farms was 0.24 over the 3 years. There were no significant differences in mean total P export between years (Table 4.3.). The main sources of P export were sold milk and livestock, accounting for 0.56 and 0.44, respectively, of total P export. Mean milk P export was 6. 66 kg P ha^{-1} (Table 4. 3.). There were significant differences in mean milk P export between farms, ranging from 4.27 to 9.52 kg ha^{-1} over the three years (Table 4.3.). The coefficient of variation for mean milk P export between farms was 0.21 over the 3 years. There were no significant differences in mean milk P export between years (Table 4.3.). Mean livestock P export was 5.10 kg P ha^{-1} (Table 4. 3.). There were significant differences in mean livestock P export between farms, ranging from 2.63 to 9.43 kg ha^{-1} over the 3 years (Table 4.3.). The coefficient of variation for mean livestock P export between farms was 0.32 over the 3 years. There were no significant differences in mean livestock P export between years (Table 4.3.).

There was a significant positive relationship (R^2 = 0.45; $P < 0.001$) between mean log transformed milk P export and mean SR (β = 0.67). An increase of 0.07 LU ha^{-1} in mean SR was associated with an increase of 0.008 (0.26, not transformed) kg ha^{-1} in mean log transformed milk P export.

There was no significant relationship between livestock P export and mean SR, number of days spent grazing, log transformed mineral fertiliser P applied to land or log transformed feed P import (Table 4.2.).

4.3.3. Phosphorus balance and phosphorus use efficiency

There was a P deficit on 8 farms and a P surplus on 13 farms. Mean P balance (P imports less P exports) was 5.09 kg P ha^{-1} (Table 4.3.). There were significant differences in mean P balance between farms, ranging from -7.42 to +19.48 kg ha^{-1} over the 3 years (Table 4.3.). The coefficient of variation for mean P balance between farms was 1 over the 3 years. There were also significant differences in mean P balance between years, ranging from 3.33 to 7.25

kg ha^{-1} in 2010 (Table 4.3.). Mean PUE (P exports divided by P imports) was 0.70 (Table 4.3.). There were significant differences in mean PUE between farms, ranging from 0.30 to 1.58 over the 3 years (Table 4.3.). The coefficient of variation for mean PUE between farms was 0.40 over the 3 years. There were no significant differences in mean PUE between years (Table 4.3.). Mean P balance per kg MS was 0.004 (Table 4.3.). There were no significant differences in mean P balance per kg MS between farms and years (Table 4.3.).

There was a significant negative relationship (R^2 = 0.71; $P < 0.001$) between mean log transformed PUE and mean log transformed mineral fertiliser P applied to land (β = -0.75) and mean log transformed feed P import (β = -0.30). An increase of 0.03 (0.92, not transformed) kg ha^{-1} in mean log transformed mineral fertiliser P applied to land and of 0.02 (0.55, not transformed) kg P ha^{-1} in mean log transformed feed P import was associated with a decrease of 0.03 (0.13, not transformed) in mean log transformed PUE.

There was a significant negative relationship (R^2 = 0.20; $P<0.01$) between mean log transformed P balance per kg MS and mean log transformed MS export per cow (β = -0.45). An increase of 0.02 (13, not transformed) kg MS cow^{-1} in mean log transformed MS export per cow was associated with a decrease of 0.05 (0.003, not transformed) kg P kg MS^{-1} in mean log transformed P balance per kg MS.

There was no significant relationship between P balance per ha and mean STP, SR, MS export, number of days spent grazing, log transformed mineral fertiliser P applied to land and log transformed feed P import (Table 4.2.).

4.4. Discussion

Total P import, export and surplus in the current study were close to, but slightly above, the national average for dairy farms and PUE was slightly lower than the national average found by Buckley *et al.* (2013) (mean total P import of 13 kg P ha^{-1}, mean total P export of 8.9 kg P ha^{-1}, mean P surplus of 4.1 kg P ha^{-1}, and mean PUE of 0.83). This would suggest that results from this study may be taken as indicative of the national situation. However, caution must be taken in this regard due to the relatively low number of farms involved (21).

The overall coefficient of variation for P imports, exports and balances and PUE, of 0.54, was within the range reported in other studies on farm-gate nutrient balances (0.64, Mounsey *et al.,* 1998; 0.51, Nielsen and Kristensen, 2005; 0.48, Treacy, 2008).

4.4.1. Factors affecting Phosphorus balances and use efficiencies across farms

Differences in mean mineral P fertilisers applied to land per ha between farms were principally associated with differences in mean STP. Mean STP content varied between 2.29 and 8.99 mg P litre^{-1} between farms. For the scope of this study (assessment of farm-gate P balances on dairy farms operating under GAP regulations), we investigated the relationship between mineral fertiliser P applied to land and soil P status to illustrate the extent to which the farmers complied to the GAP regulations imposing higher P fertilisation rates for soils with low P status and lower P fertilisation rates on soils with higher soil P status; the compliance with GAP regulations in terms of P fertilisation rates is one reason explaining the mineral fertiliser P imports and the actual P application to land. The results showed differences between recommended amounts of mineral P fertilisers, in line with GAP regulations, and the actual amounts of P applied to land. More precisely, in the fertiliser plans, the recommended mineral fertiliser P application rates ranged between 0 and 37.50 kg P ha^{-1}, the higher rates corresponding to farms with a higher proportion of Index 1 and 2 soils. In practice, P fertiliser application rates, averaged across the farm area, ranged between 1.69 and 20.15 kg P ha^{-1} between farms. The actual values and the negative relationship between mean mineral fertiliser P applied to land and mean STP indicate compliance with recommended fertilisation rates and the GAP regulations. The difference between the recommended and actual P fertilisation rates indicates that farmers with high P soils are relying more on soil P reserves to support herbage yields, and are not fully replacing P being removed in herbage. The actual P fertilisation rates were lower than the rates between 14 and 40 kg P ha^{-1}, which can be taken up by pastures in one growing season, in Ireland (Ryan and Finn, 1976; Power *et al.*, 2005). Of course, there are also P inputs to pastures from imported feeds and recycling to soil of P taken up in the sward. This trend will save money on inputs in the short term and can be expected to reduce the proportion of high P (Index 4) soils, reducing the risk of P loss to water, as was intended in the GAP regulations. At the same time, it will be necessary to monitor soil P contents and P application rates to ensure adequate soil fertility is maintained in the future (Lalor *et al*., 2010). The fact that STP explained only 0.21 of the variation in mean mineral fertiliser P applied to land indicates that a number of other factors are important, such as use of organic P fertilisers, concentrate P imports (affect the overall farm mineral fertiliser P allowance under the GAP regulations), economic considerations, weather and grass growth conditions, advisory impact and understanding and planning on the part of the farmer, for example.

The significant positive relationship between feed P import and SR suggests increased requirement for feed imports to support higher stocking rates. Concentrate feed imports per animal varied significantly between farms, from 221 to 801 kg DM LU^{-1}. These imports were likely determined by harvested grass, ranging between an estimated 2,919 and 4,304 kg DM LU^{-1} and targeted milk yields per cow, ranging between 4,229 and 6,038 litres cow^{-1}. Targeted milk yields per cow were included in development plans introduced in 2009 for each farm by farm advisors. One of the goals in the development plans was increased milk yield per cow by amounts ranging between 100 and 400 litres cow^{-1} between 2009 and 2011. The decrease in feed P import with number of days grazing suggests that extending the grazing season is an effective strategy to decrease feed P import, by increasing the proportion of grazed grass in the diet. The fact that SR and days grazing explained only 0.20 of the variation in feed P import suggests that other factors are important, such as advisory impact, economic and environmental factors.

The significant positive relationship between milk P export per ha and SR implies that increasing SR is an effective strategy to increase milk P export. Further, this could decrease P surplus and increase PUE, because P in sold milk was the main form of exporting P off the farms. However, from 16.85 kg P ha^{-1} of mean total P import, only 6.66 kg P ha^{-1} or 0.39, on average, was exported in sold milk, meaning that the impact of milk P export on P surplus and PUE was rather low. The P content of sold milk is very unlikely to increase, and therefore there is a need to optimise the use of P imports, principally feed, and on-farm P resources relative to P exports in milk, to decrease P surplus and increase PUE. It is also notable that livestock exports accounted for a large proportion of P exports and there may also be scope to improve P balances and PUE here.

The fact that PUE decreased principally with mineral fertiliser P applied to land but also feed P import, explaining 0.71 of the variation in PUE, suggests that decreasing fertiliser P and feed P imports may be the most effective strategy to increase PUE. The remainder of the variation in PUE could be attributed to factors such as differences in soil P status relative to the agronomic optimum (between 5.1 and 8.0 mg P $litre^{-1}$; Ryan and Finn, 1976; Herlihy *et* *al.*, 2004; Power *et al.,* 2005) and farm-specific efficiency of P recycling and P losses between soil, pasture, animals and milk and livestock for export (Spears *et al.,* 2003). At farm level, P recycling can contribute to increases in PUE by correcting the imbalance in soil P status across fields, which indicates legacy of past management with over- and under-applications of fertiliser P causing either soil P build-up or soil P fall below agronomic optimum levels (Wall *et al*., 2012). However, farms with high soil P status have low requirement for P fertilisers (Wall *et al*., 2012) in the long term, as P can be immobilised by soil particles and can remain stored in the soil, being sparingly available to plants (Buckley *et al*., 2013). This low P requirement means decreased P imports which can further result in apparent increases in PUE. A decrease in fertiliser and feed P imports combined with improved on-farm P recycling may increase PUE. Improved nutrient recycling on farms is consistent with one of the targets in the Food Harvest 2020 national strategy for sustainable growth of the agricultural sector (DAFM, 2013c). On a global scale, increases in PUE over the long term, along with P recovery and reuse from all waste streams throughout the food production system (from animal excreta to crop wastes) are suggested to contribute to sustainable P use (Cordell *et al.,* 2011).

Results suggest that an increase in MS exports per cow can contribute to reduced P surplus per kg MS. In grazed grass-based production systems, increased MS production and exports per cow may be achievable with low fertiliser and feed P use by optimising other management aspects such as grazing management, grass utilisation (O'Donovan *et al.,* 2002; Kennedy *et al.,* 2005), and management of herd genetic potential (Berry *et al,*. 2007). On the other hand, an increase in MS production per cow can lead to increased P surplus per ha and potentially higher P losses, if it is not achieved in an efficient manner.

4.4.2. Factors affecting Phosphorus balances and use efficiencies across years

Phosphorus feed P imports and P surplus per ha were greater in 2010 compared with 2009 and 2011. The increased feed P imports were probably to support a SR that was 0.18 LU ha^{-1} greater than 2009 and 0.19 LU ha^{-1} greater than 2011. The higher SR in 2010 was associated with higher feed imports, both in kg per ha and in kg per LU, and with higher milk yields per cow, of 5,411 litres cow^{-1} in 2010 compared with 5,120 litres cow^{-1} in 2009 and 5,291 litres cow^{-1} in 2011. This equates to a response of 2.40 litres milk kg^{-1} DM of additional feeds compared with 2009 and 0.69 litres milk kg DM^{-1} compared with 2011. A similar response in milk production, of 1.06 kg cow^{-1} per additional kg of imported feeds, was reported by Shalloo *et al.* (2004).

The increase in mean feed P import in 2010 contributed to increased mean total P import, which was 4.63 kg P ha^{-1} greater compared with 2009 and 4.78 kg P ha^{-1} greater compared with 2011. The increased total P import resulted in an increase in P surplus (7.25 kg P ha^{-1}) of 36 % compared with 2009, and 55 % compared with 2011. Others have found similar results (Smith *et al.*, 2003). The principle reason would appear to be reductions in PUE associated with the increase in feed P imports. These results highlight the necessity of assessing balances and use efficiencies in aggregate over a number of years.

4.4.3. Phosphorus balance and use efficiency before and after the GAP regulations

The results of the current study were compared with similar studies, completed between 2003 and 2006 (Treacy, 2008) and in 1997 (Mounsey *et al.*, 1998), before the introduction of the GAP regulations, to investigate possible impacts of these Regulations on P balances and PUE on Irish dairy farms. The study of Treacy (2008) was carried out on 21 intensive dairy farms, of which 8 were also involved in the current study, whereas the study of Mounsey *et al.* (1998) was on 12 intensive dairy farms. However, these intensive farms had SRs of 2.37 LU/ha (Treacy, 2008) and 2.58 LU ha^{-1} (Mounsey *et al.*, 1998), respectively, compared with the national average SR of 1.85 LU ha^{-1} in 2005-2006 (Connolly *et al*., 2006; 2007) and 1.47 LU ha^{-1} in 1997 (Fingleton, 1997) (Table 4.4.). Therefore, they may not be fully representative of all Irish dairy farms. Also, the farms in those studies were stocked more intensively than the mean SR of 2.06 LU ha^{-1} in the current study.

Table 4. 4. Comparative mean values (and standard errors) for total utilised agricultural area, stocking rate, national average stocking rate, soil test P, milk yield, milk protein and fat concentration, concentrate feeds, mineral P fertilisers applied to land, imports of P in feed stuffs, and livestock, exports of P in milk and livestock, farm-gate P balances per ha, P use efficiencies, and P balances per kg milk solids on dairy farms before and after the implementation of Good Agricultural Practice regulations in Ireland; standard error of the means for transformed data in brackets; P-values from ANOVA are included

	Current study	Treacy 2008	Mounsey *et al.* 1998	S.E.M.	*P*-value
TUAA (ha)	71	59	65	3.27(0.02)	NS
Stocking rate (LU ha^{-1})	2.06	2.37	2.58	0.049	<0.001
National stocking rate (LU ha^{-1})	1.90	1.85	1.47	-	-
STP (mg $litre^{-1}$)	5.64	8.20	11.68	0.463(0.025)	<0.001
Milk yield (l cow^{-1})	5,308	5,167	5,588	65.4	NS
Milk protein (%)	3.4	3.4	3.3	0.01(0.001)	<0.001
Milk fat (%)	4.0	3.8	3.7	0.02(0.002)	<0.001
Concentrate feed (kg DM LU^{-1})	488	549	480	29.4	<0.05
P imports (kg P ha^{-1})					
Mineral fertiliser applied	7.61	10.22	23.45	1.405(0.067)	<0.01
Feeds	7.62	7.58	7.82	0.456(0.025)	NS
Livestock	1.61	0	0	-	-
Total	16.85	17.80	31.27	1.552(0.036)	<0.05
P exports (kg P ha^{-1})					
Milk	6.66	7.35	9.13	0.296(0.016)	<0.01
Livestock	5.10	4.84	2.64	0.241	<0.001
Total	11.76	12.19	11.77	0.338	NS
P balance (kg P ha^{-1})	5.09	5.61	19.50	1.282(0.084)	<0.001
P use efficiency	0.70	0.68	0.37	0.078(0.034)	<0.001
P balance kg kg MS^{-1}	0.004	0.017	0.021	0.0153(0.0629)	<0.01

TUAA, total utilised agricultural area; LU, livestock units; STP, soil test phosphorus; l, litres; DM, dry matter; MS, milk solids; S.E.M., standard error of the means; NS, not significant.

Mean P surplus was significantly lower ($P < 0.001$) in the current study, at 5.09 kg P ha^{-1}, than Treacy (2008) (5.61 kg P ha^{-1}) and Mounsey *et al.* (1998) (19.50 kg P ha^{-1}), while PUE was significantly higher ($P < 0.001$), at 0.70, than Treacy (2008) (0.68) and Mounsey *et al.* (1998) (0.37) (Table 4.4.). Similarly, mean P surplus per kg MS was significantly lower ($P < 0.01$), at 0.004 kg P kg MS^{-1}, compared to Treacy (2008) (0.017 kg P ha^{-1}) and Mounsey *et al.* (1998) (0.021 kg P ha^{-1}) (Table 4.4.). Results suggest a trend for decreased P surplus per ha and per kg MS, and improved PUE on Irish dairy farms over the period covered by these studies (1997 to 2011) and following the introduction of the GAP regulations in 2006, associated with a trend for decreasing stocking density. This trend would have both agronomic and environmental implications. From an agronomic perspective, it will be necessary to monitor soil P to ensure adequate soil fertility for sward growth (Lalor *et al.*, 2011). From an environmental perspective, this should lead to less potential for P loss from the system.

There are a number of factors determining these differences between the three studies. The first factor was a significantly lower ($P < 0.001$) mean SR in the current study, of 2.06 LU ha^{-1}, in comparison with 2.37 LU ha^{-1} in Treacy (2008) and 2.58 LU ha^{-1} in Mounsey *et al.* (1998). The lower SR in the current study had further impacts on mineral P fertiliser applied to land and milk and livestock P exports.

Second factor was a significantly lower ($P < 0.001$) mean mineral fertiliser P applied to land, of 7.61 kg P ha^{-1}, in the current study, compared with 10.22 kg P ha^{-1} in Treacy (2008) and 23.45 kg P ha^{-1} in Mounsey *et al.* (1998). It would seem likely that this decrease was due to improved awareness of management of soil P status on farms (Lalor *et al.*, 2010) and good agricultural practices in P management such as more appropriate rates of application and better use of on-farm organic P fertilisers, as introduced in the GAP regulations.

The third factor differing between the studies suggests that this was indeed the case, as 42 % of annual organic fertiliser P (farm yard manure and slurry) was applied between mid-January and April in the current study, compared with 55 % in Treacy (2008) but only 14 % in Mounsey *et al.* (1998). There was no application of organic fertilisers after October in the current study and in Treacy (2008), whereas in Mounsey *et al.* (1998), 31 % was applied between November and January. This significant shift in the timing and percentage of organic P fertiliser application is consistent with advice on best practice indicating better fertiliser replacement value for spring application (Alexander *et al.*, 2008) and with the GAP regulations (European Communities, 2010) that prohibit application of organic fertilisers during the ‘closed period’, from mid-October to mid/end January. Also, spring application of organic P, besides reducing the requirement for imports of inorganic P, coincides with the development phase of grass plants and, therefore, can improve PUE in grasslands (Alexander *et al.*, 2008) The concurrent decrease in mineral fertiliser P use indicates an improved awareness of the fertiliser value of organic manures and accounting for them in nutrient management planning. This was illustrated in Figure 1, which indicates the appreciation of on-farm organic sources of P, and also presents challenges in terms of the ability of farmers to target P, as there is more uncertainty in application rates for organic P fertilisers, and the ability to apply it can be more limited spatially and temporally in comparison with the mineral P fertilisers.

The farms in this study had a significantly lower ($P < 0.001$) mean STP content of 5.64 mg P litre^{-1} compared to Treacy (2008) (8.20 mg litre^{-1}) and Mounsey *et al.* (1998) (11.68 mg P litre^{-1}). This is in line with the historical variation in STP in agricultural Irish soils, with an increase from about 1 mg litre^{-1} in the early 1950s to 9 mg litre^{-1} in 1990s (Tunney, 1990), and a fall down to 6.7 mg litre^{-1} in 2003 (Bourke *et al.*, 2008) and from 7.3 to 4.0 mg litre^{-1} between 2007 and 2011 (Wall *et al.*, 2012). In the current study, the implementation of GAP regulations obliged the farmers to operate STP contents considered optimal for response in herbage yields, of between 5.10 and 8.00 mg litre^{-1} (European Communities, 2010). The fact that the farms in this study were operating at lower STP combined with lower surpluses and higher PUEs than the previous studies suggests much more efficient P cycling with much less potential to lose P to water.

4.4.4. Phosphorus balance and use efficiency of Irish dairy farms in an international context

The results of the current study were compared with similar European studies completed after the implementation of the Nitrates Directive and with a study from Australia, as outlined in Table 4.5. In this comparison, the term ‘continental European farms’ refers to the Dutch farms in Aarts (2003), the Danish farms in Nielsen and Kristensen (2005), and the French farms in Raison *et al.* (2006).

Mineral fertiliser P applied to land in the current study (7.61 kg P ha^{-1}) was lower than the Dutch farms in Aarts (2003) (8.50 kg P ha^{-1}), the English and Irish farms (12.46 kg P ha^{-1}) and the French farms (11.29 kg P ha^{-1}) in Raison *et al.* (2006), and the Australian farms in Gourley *et al.* (2012) (16.60 kg P ha^{-1}), but higher than the Danish farms in Nielsen and Kristensen (2005) (5.00 kg P ha^{-1}). .

Table 4. 5. Comparative number of farms, type of system, grassland area, stocking rate, milk yield, P imports from mineral fertilisers and feed stuffs, P exports in milk, P surpluses, and P use efficiencies in different regions

Reference	Region	No. Farms	Type of system	Grassland (proportion of TUAA)	SR (LU ha^{-1})	Milk yield (l ha^{-1})	Fertiliser P import (kg P ha^{-1})	Feed P import (kg P ha^{-1})	Milk P export (kg P ha^{-1})	P surplus (kg P ha^{-1})	PUE
Current study	South of Ireland	21	G/C	0.93	2.06	7,569	7.61	7.62	6.66	5.09	0.70
Aarts (2003)	The Netherlands	17	G/C	0.76	1.74	14,528	8.50	24.00	19.00	13.50	0.58
Nielsen and Kristensen (2005)	Denmark	25	D+A	0.59	1.54	12,631	5.00	22.00	7.00	16.00	0.46
Raison *et al.* (2006)	Scotland	10	G/C	0.94	1.60	7,155	13.20	12.76	7.92	17.60	0.33
	South of Ireland	24	G/C	1.00	2.10	7,757	11.00	7.04	7.48	7.92	0.62
	SW England	13	G/C	0.84	2.20	9,847	13.20	11.88	9.68	15.40	0.44
	Brittany	15	G/MS	0.70	1.40	5,315	4.40	17.16	5.28	15.84	0.48
	Pays de la Loire	13	G/MS	0.65	1.30	4,837	5.72	10.12	4.84	9.68	0.57
	Aquitaine	9	C/MS/MG	0.39	1.20	6,053	23.76	13.20	5.72	21.56	0.43
	Basque country	16	0G	0.88	2.70	15,304	10.12	45.32	16.28	39.96	0.35
	Galicia	18	0G	0.58	3.00	19,723	35.20	57.20	18.04	71.72	0.24
	North Portugal	21	0G	0.00	6.10	34,760	29.92	66.00	32.12	51.04	0.48
Gourley *et al.* (2012)	Australia	37	G/C	0.83	1.75	13,975	16.60	9.20	10.00	25.80	0.32

No., number; G/C, grazing-cutting; D + A, dairy + arable crops; G/MS, grazing-maize for silage; C/MS/MG, cutting-maize for silage-maize for grain; 0G, zero-grazing; TUAA, total utilised agricultural area; SR, stocking rate; LU, livestock units; l, litres; P, phosphorus; PUE, phosphorus use efficiency.

Feed P import in the current study (7.62 kg P ha^{-1}) was much lower compared with Aarts (2003) (24.00 kg P ha^{-1}), Nielsen and Kristensen (2005) (22.00 kg P ha^{-1}), the English and Irish farms (10.56 kg P ha^{-1}) and the French farms (13.49 kg P ha^{-1}) in Raison *et al.* (2006). The main reason for higher feed P imports in these studies was the high import system of dairy production that is more typical of dairy production in continental Europe, characterised by year-round milk production, high use of imported feeds, lower use of grazed grass and high milk yields per ha and per cow. In contrast, a low import system is more typical in Ireland, with seasonal grass-based milk production (compact spring calving), low use of imported feeds, high use of grazed grass and lower milk yields per ha and per cow. The continental European studies (14,528 litres ha^{-1}, Aarts, 2003; 12,631 litres ha^{-1}, Nielsen and Kristensen, 2005) and the English and Irish farms in Raison *et al.* (2006) (8,253 litres ha^{-1}) had much higher milk yields per ha compared with the current study (7,569 litres ha^{-1}). The French farms in Raison *et al.* (2006) had lower mean milk yield per ha (5,401 litres ha^{-1}) due to mixed agricultural production (milk, maize for export) on some of the farms. The higher milk yields per ha were also associated with higher mean milk P exports per ha on the Dutch farms in Aarts (2003) (19.00 kg P ha^{-1}) and the English and Irish farms in Raison *et al.* (2006) (8.36 kg P ha^{-1}) compared with the current study (6.66 kg P ha^{-1}). Despite the higher milk yields in Nielsen and Kristensen (2005), mean milk P export (7.00 kg P ha^{-1}) was similar to the current study, due to mixed agricultural production (milk, cereals for export). On the French farms in Raison *et al.* (2006), the mean milk P export, of 5.28 kg P ha^{-1}, was lower than in the current study, likely due to their lower milk yields and SR.

In the study of Gourley *et al.* (2012), on Australian farms, year-round grazing allowed for high use of grazed grass and therefore lower imports of feeds (9.20 kg P ha^{-1}) than the continental European farms and the English and Irish farms in Raison *et al.* (2006), but higher than the Irish farms in the current study, due to much higher milk yields per ha (13,975 litres ha^{-1}).

Despite the relatively low milk P export per ha, mean P surplus (5.09 kg P ha^{-1}) in the current study was much lower than that reported by Aarts (2003) (13.50 kg P ha^{-1}), Nielsen and Kristensen (2005) (16.00 kg P ha^{-1}), the English and Irish farms (13.64 kg P ha^{-1}) and the French farms in Raison *et al.* (2006) (15.69 kg P ha^{-1}), and the Australian farms in Gourley *et al.* (2012) (25.80 kg P ha^{-1}). This reflects the low import model of dairy production in Ireland.

Mean PUE in the current study (0.70) was much higher than that reported by Aarts (2003) (0.58), Nielsen and Kristensen (2005) (0.46), the English and Irish farms (0.46) and the

French farms (0.49) in Raison *et al.* (2006), and the Australian farms in Gourley *et al.* (2012) (0.32).

It can be concluded that Irish dairy farms tend to operate with lower feed P imports, relatively low fertiliser P imports and lower P surpluses per ha than most other European dairy farms at lower exports (litres milk ha^{-1}) and that this is largely due to the low import system that is more typical in Ireland with seasonal milk production (compact spring calving) (Buckley *et al.*, 2000), low use of imported feeds (Dillon *et al.*, 1995), high use of grazed grass (Horan, 2009), and relatively low milk yields per cow (Humphreys *et al.*, 2009). All other factors being equal, one might expect less P losses to the environment under conditions of lower P surplus.

4.5. Conclusions

A survey of 21 Irish dairy farms from 2009 to 2011 found a mean P surplus of 5.09 kg ha^{-1}, or 0.004 kg P kg MS^{-1}, and a mean PUE of 0.70. Farm-gate P imports were dominated by feeds (7.62 kg P ha^{-1}) and inorganic fertiliser (7.61 kg P ha^{-1}), while exports were dominated by milk (6.66 kg P ha^{-1}) and livestock (5.10 kg P ha^{-1}). Comparison to similar studies carried out before the introduction of the GAP regulations in 2006 would suggest that P surplus, both per ha and per kg MS, have significantly decreased (by 14.41 kg P ha^{-1} and 0.017 kg P kg MS^{-1}, respectively) and PUE increased (by 0.33) following the introduction of the GAP regulations. These improvements have mostly been achieved through decreased mineral fertiliser P applied to land and improvements in P management, with a notable shift towards spring application of organic manures, consistent with advice on best practice and with the GAP regulations that prohibit application of organic fertilisers during the 'closed period' from mid-October to mid/end January. A concurrent decrease in mineral fertiliser P use indicates an improved awareness of the fertiliser value of organic manures and accounting for them in nutrient management planning. The cumulative effect of the improvement in management of organic manures and the decrease in mineral fertilisers may have led to the lower mean STP values observed in the current study, closer to values considered optimal for pasture production. These results would suggest a positive impact of the GAP regulations on dairy farm P surplus, PUE and STP.

Taking surplus P per ha and STP as indicators of local environmental pressure, this indicates that the environmental sustainability of milk production has improved. Taking PUE as an indicator of agronomic performance, the improvement in PUE also indicates that agronomic performance has improved concurrently. This demonstrates that is possible to improve both environmental and economic sustainability of dairy production through improved resource use efficiencies. Such improvements will be necessary to achieve national targets of improved water quality under the EU Water Framework Directive and increased dairy production, as set out in the Food Harvest 2020 Report. Results suggest that optimising mineral fertiliser P applied to land and and feed P imports combined with improved on-farm P recycling may be the most effective way to increase PUE. Equally, continued monitoring of STP and P management will be necessary to ensure that adequate soil P fertility is maintained.

Mean P surplus was lower and mean PUE was higher than the overall mean surplus (15.92 kg P ha^{-1}) and mean PUE (0.47) from three studies of continental European dairy farms. It can be concluded that Irish dairy production systems, on average, tend to operate with lower mineral fertiliser P applied to land and feed P imports and lower P surpluses than other continental European dairy production systems and that this is largely due to the low import system that is more typical in Ireland, with seasonal milk production (compact spring calving), low use of imported feed stuffs, and high use of grazed grass. All other factors being equal, one might expect less P losses to the environment under conditions of lower P surplus.

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5. Economic impacts of nitrogen and phosphorus use efficiency on nineteen intensive grass-based dairy farms in the South of Ireland

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Summary

A three year (2009-2011) study found a mean N surplus of 167 kg N ha^{-1}, P balance of 1.72 kg P ha^{-1}, NUE of 0.26, PUE of 1.08, and net profit of €598 ha^{-1} on 19 Irish dairy farms. Between farms, the increase in mean net profit with mean milk receipts and the decrease with mean expenditure on mineral fertiliser N use implies that increasing milk receipts while decreasing mineral fertiliser input and expenditure is an effective strategy to increase net profit. Between years, the 86 % increase in mean net profit was not fully explained by the 37 % increase in mean milk receipts, indicating that an increase in milk receipts alone do not ensure the financial security of dairy farmers in the long term. Mean net profit was not directly related to mean N and P surplus or N and P use efficiency. However, there was an indirect link between net profit and N and P use efficiency, as indicated through significant relationships between N and P use in the form of mineral fertilisers and feeds and the associated expenditures on mineral fertilisers and feeds. The increase of mean expenditure on feeds with mean SR and feed P input and the decrease with TUAA highlights the importance of matching SR with the feed (concentrate, fodders) imports on grass-based dairy farms, when there is limited availability of grassland area, as the most effective strategy to control the feed expenditures, with potential positive impact on net profit. Results of the sensitivity analysis indicated that milk price was the main driver for changes in net profit in high and low milk price situations investigated across nine price scenarios. The increase in mean N-eco-efficiency (milk produced per kg N surplus) (51.4 litres kg N^{-1}) with mean fertiliser N input implies that efficient on-farm N management of fertiliser N inputs, aiming at supporting herbage and therefore milk production while making efficient use of N, is an effective strategy to achieve increases in milk production and therefore reduce N surplus per unit product (litre milk). Potential fertiliser N replacement values of €317 ha^{-1} for the spring and €64 ha^{-1} for summer slurry application may represent strong incentives for farmers to make increased use of organic fertilisers, as part of overall on-farm N management, with positive impacts on farm nutrient use efficiency and farm net profit. Nine farms exceeding the limit of 2 LU ha^{-1}, imposed through the Nitrates Directive, had 1.63 times higher net profit compared with the remainder, which justified the cost of compliance associated with being in derogation. The results of this study generally indicate that Irish dairy farms, as low input production systems, have the potential to achieve both economic (as indicated by net profit per ha) and

environmental (as indicated by N and P balances per ha, N and P use efficiency and N-eco-efficiency) sustainability.

5. 1. Introduction

There is an on-going debate surrounding the use of high- or low-input systems in dairy farming. The low-input systems are considered more economically and environmentally sustainable than the high-input systems (Ridler, 2008; O'Brien *et al.,* 2012) as they are less vulnerable to volatility in input and output prices (Humphreys *et al.,* 2012; Moreau *et al.,* 2012) and are associated with lower farm nutrient surpluses (Humphreys *et al.*, 2008; Ledgard *et al.*, 2009).

Relatively high milk prices between 2001 and 2011 (€0.30 litre^{-1}, on average; CSO, 2013) within the European Union (EU-27) have encouraged increased use of inputs, in the form of nitrogen (N) and phosphorus (P) mineral fertiliser (Aarts, 2003; Raison *et al.,* 2006; Groot *et al.,* 2006; Nevens *et al.,* 2006; Roberts *et al.,* 2007; Ryan *et al.*, 2011; Cherry *et al.*, 2012), and concentrate feeds in dairy production systems (Shalloo *et al.,* 2004*a*; McCarthy *et al.,* 2007; Delaby *et al.,* 2009; Patton *et al.,* 2012). These high fertiliser applications may often be attributed to risk aversion to lower crop yields or incentive emerging from fertiliser pricing. The lower the relative price of fertiliser, the greater the incentive to apply it to offset potential risk and yield uncertainty (Buckley and Carney, 2013). Also, the volume of bought-in feeds is often driven more by the desire to produce specific volumes of product rather than by the desire to make the most efficient use of inputs. Concurrently, there has been a general tendency to overlook the importance of the 'free' resource (pasture and soil nutrient supply) (Ridler, 2010).

The fertiliser and feed inputs are key drivers of increased herbage yields and milk saleable output on most dairy farms (Treacy *et al.,* 2008; Ryan *et al*., 2011; Gourley *et al.*, 2012) and therefore represent the main expenditures of milk production (Tozer *et al.*, 2003; Donnellan *et al.,* 2011). However, the N (Jarvis, 1993; Goodlass *et al.,* 2003; Aarts, 2003; Humphreys *et al*., 2008) and P inputs (Van Keulen *et al.,* 2000) from mineral fertilisers and feeds typically exceed outputs in milk and livestock exported off the farms. These imbalances result in surplus N (Gourley *et al*., 2010; Cherry *et al.,* 2012) or P (Arriaga *et al*., 2009; Gourley *et al*., 2010) that are either accumulated on, or lost from, the dairy farms. The excessive use of N and P may be associated with environmental damage. Nitrogen surplus is susceptible to be lost through denitrification, nitrate (NO_3) leaching, ammonia (NH_3) volatilisation, N_2O emissions or through runoff to surface waters (Pain, 2000; Jarvis and Aarts, 2000; De Vries *et al.,* 2001; Schils *et al.,* 2005; Del Prado *et al*., 2006). Surplus P is potentially lost through eroded material containing particulate P or P adsorbed on to organic clay soil complexes (Kurz *et al*., 2005) or through leaching (Heathwaite, 1997). Losses of N and P also incur economic costs in two ways; the cost of wasted N and P inputs, at farm level, and the cost of clean-up associated with pollution caused as a result of such losses, more typically at regional to national levels (Buckley and Carney, 2013). The same study reported average surpluses sourced from mineral fertilisers of 28.23 kg N ha^{-1} and 3.38 kg P ha^{-1}, which were found to be at least similar to losses through leaching and runoff for N and P, respectively, from intensive dairy farms, and equated to €44.8 ha^{-1}, on average. It has been proposed that these environmental costs should be factored into the sale price of milk (Von Keyserlingk *et al.,* 2013).

In addition, increasing instability of milk prices and increasing input prices (Soder and Rotz, 2001), as well as rising labour, machinery and animal housing expenditures (MacDonald *et al.,* 2008) are leading dairy farmers to search for ways to decrease milk production expenditures, and grazed grass-based dairy systems offer opportunities to reduce these expenditures during the grazing season (Soder and Rotz, 2001; MacDonald *et al.,* 2008). Strategies to reduce expenditures in grazed grass-dairy production systems include increasing resource use efficiency (Ridler, 2008; Finneran *et al.,* 2011; Patton *et al.,* 2012; Kelly *et al.,* 2013), nutrient use efficiency (Gourley *et al.,* 2010), N-eco-efficiency (the amount of milk produced per kg of N surplus) (Nevens *et al.,* 2006; Beukes *et al.,* 2012), accounting for mineral nitrogen fertiliser replacement value (NFRV) of organic N contained in slurry (Lalor, 2008) or fixed by white clover in pastures (Humphreys *et al.,* 2012). Therefore, there has been a rejuvenated interest in grass-based dairy production systems internationally (MacDonald *et al.,* 2008) as a low-input, low-cost system that may be less vulnerable to volatility in input and product prices**.**

In many developed countries, much of commercial farming operates under the influence of society's increasingly multifunctional expectations. Such farming must thus be sustainable within a range of economic and environmental criteria (Crosson *et al.,* 2007). Therefore, a comprehensive evaluation of management effects on dairy farm performance must consider environmental impacts as well as potential profit (Rotz *et* *al.*, 2005). As N (Leach and Roberts, 2002; Eckard *et al.*, 2004; Powell *et al.*, 2010) and P (Jarvis and Aarts, 2000) surpluses are commonly associated with excessive, inefficient N and P use on farms, as well as harmful environmental impacts, they are considered as indicators of potential N and P losses and environmental performance (Jarvis and Aarts, 2000; Schröder *et al.*, 2003; Carpani *et al.*, 2008). Therefore, in the current study, N and P balances were used as indicators of environmental sustainability. The economic sustainability of farms can be defined as their ability to generate sufficient funds to sustain their production potential in the long run (European Comission, 2001). In the current study, the net profit was used as an indicator of economic sustainability.

In the EU, dairy production is strongly regulated by environmental and agricultural policies. The Nitrate Directive (91/676/EEC) (European Council, 1991) and Water Framework Directive (WFD) (2000/60/EC) (European Council, 2000) have established guidelines in relation to farming practices to reduce NO_3 leaching and improve water quality. The Nitrate Directive was firstly implemented in Ireland as the Good Agricultural Practice (GAP) Regulations, in 2006 (European Communities, 2006). Under the GAP Regulations, farms are limited to a stocking rate (SR) based on organic N ha^{-1}, while fertiliser N and P application practices are governed by soil conditions, soil nutrient content and inputs from other sources of nutrients.

Also, in 2008, the “Health Check” decisions of Common Agricultural Policy (CAP) included the expiry of the milk quota system after 2014 and an increase of quotas by 1 % annually from 2009 to 2013 to allow for a “soft landing” of the milk sector with expiring quotas (Kempen *et al.*, 2011). In Ireland, the removal of milk quotas is expected in 2015. It is anticipated that this will create an imbalance between milk supply and milk demand that may lead to higher milk price volatility (Kelly *et al.*, 2012), which is inherent in a market which is not constrained by supply (Geary *et al.*, 2012). In fact, in the EU-27 countries, milk price has been highly volatile since 2007, ranging between €0.27 and €0.35 $litre^{-1}$ (CSO, 2013).

Under these conditions, work has been undertaken on grass-based dairy farms in Europe with specific focuses on N (Groot *et al.*, 2006; Nevens *et al.*, 2006; Roberts *et al.*, 2007; Treacy *et al.*, 2008; Cherry *et al.*, 2012; Oenema *et al.*, 2012) or P (Mounsey *et al.*, 1998; Van Keulen *et al.*, 2000; Steinshamn *et al.*, 2004; Nielsen and Kristensen, 2005; Huhtanen *et al.*, 2011) use efficiency and the economic impacts of implementing the Nitrate Directive (Van Calker *et al.*, 2004) and Water Framework Directive (Jacobsen,

2009). In Ireland, the economic implications of compliance with the Nitrate Directive and decoupling of single farm payments (SFP) on dairy farms were investigated by Hennessy *et al.* (2005) and those of milk quota abolition were investigated by McDonald *et al.* (2013). Crosson *et al.* (2007) investigated economic beef production systems in relation to N and P management strategies. Buckley and Carney (2013) investigated economic impacts of the management of N and P inputs from mineral fertilisers and feeds on dairy farms based on one-year data. However, none of the above studies included both economic impacts of N and P use efficiencies, economic implications of compliance with Nitrate Directive regulations, and sensitivity to volatility of milk and mineral fertiliser prices on grazed grass-based dairy farms.
Therefore, the objectives of this study were: (i) to assess the economic impacts of N and P farm-gate balances and use efficiency on 19 commercial intensive grass-based dairy farms; (ii) to assess economic implications of compliance with the Nitrate Directive regulations on these farms; (iii) to assess the sensitivity of the dairy production system on these farms to volatility in milk and fertiliser prices. For these purposes, data on N and P imports and exports and farm receipts and expenditures were recorded on 19 intensive grass-based dairy farms in the south of Ireland participating in the INTERREG-funded DAIRYMAN project over 3 years, from 2009 to 2011.

5.2. Materials and methods

5.2.1. Farm selection and data collection

Twenty-one commercial intensive dairy farms were selected, located in the South of Ireland, in counties Cork, Limerick, Waterford, Tipperary, Kilkenny, and Wicklow. These farms were pilot farms involved in the INTERREG-funded DAIRYMAN project (www.interregdairyman.eu) focusing on improving resource use efficiency and competitiveness of dairy farms in Northwest Europe. Farm selection was based on the likely accuracy of data recording, 8 of the farms in the current study having been involved in a previous similar study (GREENDAIRY; Treacy *et al.*, 2008), and all the farmers being willing to provide data. Grass-based milk production from spring calving cows was the main enterprise on all the selected farms. Key farm characteristics are given in Table 5.1.

Seventeen of the farms in the current study participated in the Rural Environment Protection Scheme (REPS) (DAFM, 2013a). This is a program co-funded by the EU and the Irish government whereby farmers are rewarded financially for operating to a set of guidelines consistent with an agri-environmental plan drawn up by an approved planning agency. Important conditions for receiving REPS financial support were to limit SR to 2 LU ha^{-1} and to apply N fertilisers to the farming area according to fertiliser plans drawn for the farm (DAFM, 2013a). Nine of the 21 farms had a stocking rate higher than 170 kg organic N ha^{-1} or 2 LU ha^{-1}. According to GAP regulations and REPS conditions (for the participating farms), these farms had to apply for a derogation allowing a maximum stocking rate of 250 kg organic N ha^{-1} or 2.9 LU ha^{-1}, mainly conditioned by prohibited application of organic fertilisers during the ‘closed period’, from mid-October to mid/end January, planning of mineral and organic fertilisers’ application relative to SR (stocking rate), and a maximum use of 279 kg N ha^{-1} and 49 kg P ha^{-1} (European Communities, 2010).

Table 5. 1. Total utilised agricultural area (and crop area), annual temperature, annual rainfall, number of days grazing, stocking rate, owned milk quotas, fat and protein corrected milk, concentrate feeds, and estimated harvested grass through grazing and silage for 19 Irish dairy farms between 2009 and 2011

Farm	TUAA (crops) (ha)	Temp. (^{0}C)	Rainfall (mm year^{-1})	GD	SR (LU ha^{-1})	Owned milk quota (litres)	FPCM output (kg ha^{-1})	Conc. (kg DM LU^{-1})	Grass (kg DM LU^{-1})
1	80	9.6	1,077	264	1.99	679,962	8,356	268	4,139
2	59	9.8	1,124	241	2.14	603,121	11,283	499	4,169
3	68	9.8	1,124	268	1.93	567,784	9,519	221	4,304
4	37	10.1	1,373	258	1.97	389,825	13,157	571	3,691
5	61 (1.0)	10.1	1,373	257	1.50	421,685	7,740	611	3,891
6	58 (3.4)	10.1	1,373	256	1.80	502,349	8,957	568	3,632
7	44	9.6	1,077	269	2.28	622,730	10,360	471	3,922
8	55	10.1	1,373	254	2.36	505,846	10,032	580	4,033
9	50	9.6	1,077	264	1.98	358,603	8,190	466	4,089
10	115 (5.3)	10.1	1,373	258	1.76	797,135	8,768	394	3,898
11	33	10.1	1,373	265	2.00	328,375	8487	615	3,508
12	47	10.1	1,373	261	1.60	330,038	7,573	604	3,886
13	70	9.6	1,077	256	1.59	532,700	8,119	710	3,730
14	82 (7.4)	9.8	1,124	269	1.54	364,993	7,636	302	3,472
15	104	9.8	1,124	267	1.53	587,768	6,854	484	3,858
16	63	9.6	1,077	252	2.11	447,326	9,651	463	4,002
17	49	9.8	1,124	250	1.98	462,000	7,279	732	3,567
18	75 (12.0)	10.1	1,373	275	1.97	537,964	8,368	265	4,011
19	39 (1.7)	10.1	1,373	255	1.16	240,942	6,829	386	4,108
Mean	63 (5.1)	9.8	1,230	260	1.85	488,481	8,798	485	3,890
S.D.	21.6 (4.10)	0.21	141	8.23	0.30	138,416	1,593	149	236

TUAA, total utilised agricultural area; temp., temperature; GD, number of days grazing; SR, stocking rate; LU, livestock units; l, litres; FPCM, fat and protein corrected milk; conc., concentrate feeds; DM, dry matter; S.D., standard deviation.

Data were collected on a monthly basis between 2010 and 2011 on the selected farms. The information collected included grassland area, area under crops, type of crops and percentage of crops fed to livestock, livestock numbers, number of days spent grazing, family and hired labour hours, imports of manure, concentrates, bedding material, silage, mineral N and phosphorus (P) fertilisers and other agro-minerals, exports of milk, crops, manure, and silage, amount of slurry applied to land and the method of application (splash plate or trailing shoe). For mineral fertilisers, amounts imported onto farms as well as amounts applied to land were recorded on a monthly basis. For year 2009, similar data were obtained from farm records and farm advisors. Data collected for the 3 years were cross-checked with secondary data sources such as Single Farm payments (SFP) forms and Nitrate' Declaration forms (data forms required from farmers for participation in state schemes) (DAFM, 2013b, c). Data on livestock imports and exports were extracted from the Dairy Management Information System (DAIRYMIS) (Crosse, 1991). Values for amounts of milk sold off the farms were extracted from the reports on milk deliveries coming from the cooperatives supplied by the farmers. Financial data on milk and livestock receipts, direct payments (SFP, REPS, and payments for disadvantaged areas), owned and leased milk quota, variable (concentrates, fodders, bedding materials, breeding, veterinary, sundry variable expenditures (removal of dead animals), livestock purchases, land rental, seed and agro-chemicals, mineral fertilisers, fences, ensiling materials, and quota and land rental) and fixed (gas, water, electricity, maintenance and insurance of buildings, hired labour, maintenance of machinery, professional fees (farm advisor, accountant, soil analyses), phone, depreciation of buildings and machinery, interest repayments-term loan) expenditures were extracted from the farmers' e-profit monitor records (Teagasc, 2012a) (a voluntary scheme for monitoring and improving farm profitability) for years 2009-2011. There were 21 farms involved in the project but two farms did not provide sufficient data for all 3 years. Therefore, data for 19 farms were used in the current economic study.

Data on mean annual rainfall and temperature were extracted from an Irish Meteorological Service database for different weather stations located in, or close to, the area of study, at Cork airport, Roche's point, Gurteen, Johnstown Castle and Oak Park (Irish Meteorological Service, 2013).

The annual amount of pasture harvested through grazing and silage on each farm was modelled using the Grass Calculator (Teagasc, 2011) based on the difference between the net energy (NE) provided by imported feeds (concentrates and fodders) and the net energy requirements of animals for maintenance, milk production, and body weight change (Jarrige, 1989).

Stocking rate was expressed as LU per ha for TUAA. One dairy cow is considered equivalent to 1 LU and 1 bovine less than 1 year old equivalent to 0.3 LU (Connolly *et al*., 2009).

The calculation of N and P inputs, outputs, balances and efficiencies are based on methods described by Watson and Atkinson (1999) and Oenema *et al*. (2003). The results on N and P inputs, outputs, balances and efficiencies for the 19 farms in the current study are presented in Table 5.6.

5.2.2. Economic model and analyses

The economic model in the current study was developed by a team of experts in the DAIRYMAN project (www.interregdairyman.eu) to facilitate comparative economic analysis of dairy production systems across the participating regions of northwest Europe. This economic model was validated on 128 dairy farms participating in the DAIRYMAN project. The inputs in the model included data on milk exports, milk protein and fat concentration, fat and protein corrected milk (FPCM), dairy livestock numbers, owned and rented TUAA, owned and rented milk quota, family and hired labour hours, milk and livestock receipts, direct payments, variable expenditures on concentrates, fodders, bedding materials, breeding and veterinary, sundry variable expenditures (removal of dead animals), livestock purchases, land rental, seed and agro-chemicals, mineral fertilisers, fences, ensiling materials, and quota rental, and fixed expenditures on gas, water, electricity, maintenance and insurance of buildings, hired labour, maintenance of machinery, professional fees (farm advisor, accountant, soil analyses), phone, depreciation of buildings and machinery and interest repayments on loans for each farm between 2009 and 2011.

The price per litre of leased quota milk was obtained from the cooperatives supplied by the farmers and it ranged between 1 and 3 cents (c) litre^{-1} in 2010, and 1 and 5 c litre^{-1} in 2011. There was no milk quota rented in 2009.

Land area was treated as an opportunity cost, with additional land rented in when required and leased out when not required for on-farm feeding of animals (McCarthy *et al.,* 2007). Own land was assigned an opportunity cost equal to the expenditure of rented land (Donnellan *et al.,* 2011). The cost of rented land represented the average regional cost of rented grassland, based on consultations with experts, with values of €355 ha^{-1} in 2009, €348 ha^{-1} in 2010 and €343 ha^{-1} in 2011. Also, family labour was assigned an opportunity cost equal to the expenditure on hired labour (Donnellan *et al.*, 2011), to ensure that this input was also accounted for (Wilson, 2011).

For the purpose of this study, profitability was expressed as the net profit, which was calculated as total receipts (milk, livestock, and subsidies) less total expenditures (variables and fixed expenditures, opportunity costs for own land and family labour) (Shalloo *et al.,* 2004b; Chamberlain, 2012). For the purpose of this study, the receipts, expenditures, and net profit were expressed on areal basis (€ ha^{-1}).

Specialist dairy farms are defined, in Ireland, as having at least two-thirds of the farm total gross profit coming from dairying activities (Donnellan *et al.,* 2011). Therefore, whole-farm TUAA, SR, livestock receipts, direct payments, variable and fixed expenditures, and net profit were allocated to the dairy enterprise according to the share of milk receipts in total farm receipts. The average allocation values were 0.88 in 2009, 0.92 in 2010 and 0.88 in 2011.

Nevens *et al.* (2006) measured the eco-efficiency of dairy farms as the amount of milk produced (as measure of production) per kg N surplus (as measure of potential environmental damage). This measure of eco-efficiency of dairy farms is in agreement with Beukes *et al*. (2012) but different than Basset-Mens *et al.* (2009), who used several measures of environmental impacts per kg of milk (Global Warming Potential (kg CO_2-eq), eutrophication (kg PO_4-eq), acidification (kg SO_2), energy use (MJ Lower Heating Values), and land use (m^2 $year^{-1}$)). In the current study, the eco-efficiency was measured similar to Nevens *et al*. (2006) and Beukes *et al*. (2012), but it is referred to as N-eco-efficiency, to differentiate from the definition of eco-efficiency (the ability of a system to fulfil a function while minimising its total impacts on the environment) used by Basset-Mens *et al.* (2009).

Accounting for NFRV of cattle slurry applied to grassland was considered as an opportunity to reduce expenditures on mineral N fertilisers throughout the year in grazed grass-dairy production systems (Lalor, 2008). For similar purpose, in the current study it was considered important to calculate the economic value of mineral N fertiliser replacement potential of cattle slurry applied to grassland throughout the ‘open’ period for slurry application (mid/end January-mid October, European Communities, 2010). For this purpose, there were considered values of €0.90 m^{-3} for the spring (January-April) application and €0.18 m^{-3} for the summer (April-late July) application (Lalor, 2008). The difference between the two values was dictated by the difference in the amount of N susceptible to be lost through NH_3 volatilisation (0.90 kg N m^{-3} of applied slurry in spring versus 1.62 kg N m^{-3} in the summer) and crop available N (0.90 kg N m^{-3} of applied slurry in spring versus 0.18 kg N m^{-3} in the summer).

A cost of compliance with the limit imposed through GAP Regulations was calculated for nine farms that had exceeded this limit (Derogation farms). The number of LUs that would need to be removed from these farms to comply with the 2 LU ha^{-1} limit was calculated, as well as the associated potential loss in net profit (Hennessy *et al.*, 2005).

5.2.3. Sensitivity analysis

Given that volatility in mineral N fertiliser prices has significantly affected expenditures on production on Irish dairy farms in recent years, it was considered important to examine the impact of changing mineral N fertiliser price (Donnellan *et al.*, 2011) on farm profitability in the current study. Therefore, a low (€0.825 kg N^{-1}), medium (€0.905 kg N^{-1}), and high (€1.029 kg N^{-1}) price were used for a sensitivity analysis of the economic effect of mineral N fertiliser prices on these farms. The low price was from 2010, the medium price was from 2009 and the high price was from 2011, indicating an increase of 12 c kg N^{-1} between 2009 and 2011 (CSO, 2012). These prices were applied to the actual amounts of mineral N fertilisers used on each farm for each year between 2009 and 2011.

Also, due to recent variation in milk price (CSO, 2012) and its expected increasing volatility after milk quota abolition in 2015 (Geary *et al.*, 2012), it seemed reasonable to examine the impact of changing milk price on farm profitability in the current study. Similar to the mineral fertiliser, a low (€0.246 $litre^{-1}$), medium (€0.309 $litre^{-1}$), and high (€0.360 $litre^{-1}$) price were used for milk. The low price was from 2009, the medium price was from 2010 and the high price was from 2011, indicating an increase of €0.11 $litre^{-1}$ between 2009 and 2011 (CSO, 2012). These prices were applied to the actual volume of milk sold off each farm for each year between 2009 and 2011. A total of nine scenarios were investigated by combining the three different fertiliser N and milk prices (Table 5.2.).

Table 5.2. Price scenarios for milk and N fertiliser used in the sensitivity analysis for 19 Irish dairy farms between 2009 and 2011

Scenario	Fertiliser N price	Milk price
S1	H	H
S2	H	M
S3	H	L
S4	M	H
S5	M	M
S6	M	L
S7	L	H
S8	L	M
S9	L	L

Fertiliser N price (H=€1.029 kg N^{-1}; M=€0.905 kg N^{-1}; L=€0.825 kg N^{-1}), milk price (H=€0.360 $litre^{-1}$; M=€0.309 $litre^{-1}$; L=€0.246 $litre^{-1}$)

Sensitivity was determined as the differences in net profit between the farms associated with the 9 scenarios of milk and fertiliser N prices (Rotz *et al.*, 2005; Humphreys *et al.*, 2012). The changes in net profit for each farm relative to the actual net profit for years 2009-2011 are presented in Table 5.5. Mean values of the changes between farms are presented in monetary units (€ ha^{-1}) and as percentages (Table 5.5.) also for each scenario, to better illustrate the impact of changing prices.

Due to the observed higher sensitivity of the net profit to milk price compared to fertiliser N price, the sensitivity of the net profit was analysed further by comparison between the six farms with the lowest milk receipts and the six farms with the highest milk receipts.

5.2.4. Statistical analysis

Descriptive statistics were applied using SPSS to calculate means and standard errors (Darren and Mallery, 2008). Normal distribution of residuals was tested using Shapiro-Wilk, with values lower than 0.05 indicating a non-normal distribution. The log transformation was required to ensure homogeneity of variance (Tunney *et al.*, 2010) for some of the variables. Therefore, own land, leased land, leased milk quota, milk output, N inputs from mineral fertiliser and concentrates, N use efficiency, milk produced per kg N surplus, P inputs from mineral fertilisers and feeds, milk P output, P balance, P use efficiency, livestock receipts, agri-environmental payments, expenditures on feeds (concentrates, fodders), veterinary, breeding, leased land and quota, grass seed and agro-chemicals (pesticides and herbicides), electricity, gas and water, hired labour machinery operation and maintenance, phone, professional fees, insurance for buildings, and interest repayments-term loan, total fixed expenditure, own land opportunity cost, total expenditure, operating profit margin, and cost of compliance were transformed using a log10 base (y=log10(x)).

Differences in mean N inputs from mineral fertilisers and concentrates, N surplus, NUE, amount of milk produced per kg N surplus, P inputs from mineral fertilisers and feeds, P balance, PUE, milk receipts, livestock receipts, SFPs, agri-environmental payments, total receipts, expenditures on mineral fertilisers and feeds, net profit, operating profit margin, and cost of compliance between years and farms were analysed using repeated measures ANOVA. A significance level of 0.05 or less (0.01 and 0.001) indicated statistically significant differences among the means. A significance level of 0.05 or higher indicated a 95 or higher percent of certainty that the differences among the means are not the result of random chance (Darren and Mallery, 2008). Such results were presented as not significant (NS).

The statistical models included farm and year effects on each of the tested variables. The 19 farms were considered as replicates. The models used were:

1. $Y_i = \mu + a_i + e_i$, where Y_i = tested variable, a_i = the effect of *i*th farm ($i = 1,\ldots,19$), and e_i = the residual error term;
2. $Y_i = \mu + b_j + e_i$, where Y_i = tested variable, b_j = the effect of *j*th year (j = 2009, 2010, 2011), and e_i = the residual error term.

Multiple stepwise linear regression was undertaken to investigate relationships between key dependent and independent variables presented in Table 5.3. The choice of the statistical models was dependent on the potential significance of independent variables and their potential impact on the dependent variables. Non-significant ($P > 0.05$) independent variables were automatically removed from the models (Table 5.3.). The probability for acceptance of new terms (F) was 0.10 (Groot *et al*., 2006) and the confidence interval was 0.95. All relationships between variables were assessed for outliers, normality and colinearity. The identified outliers were diminished through log transformation.

Uncertainty analysis was carried out by calculating the coefficient of variation as the ratio between standard deviation and mean value (Gourley *et al.,* 2010) for N inputs from mineral fertilisers and concentrates, N surplus, NUE, milk produced per kg N surplus, P inputs from mineral fertilisers, P balance, PUE, milk receipts, livestock receipts, SFPs, agri-environmental payments, total receipts and for expenditures on mineral fertilisers and feeds on the 19 farms between 2009 and 2011, expressed as a proportion.

Table 5. 3. Investigated and significant multiple stepwise linear regression models

Investigated	Significant
LgmlkNsur = μ + βLgTUAA + βSR + βGD + LgFPCMha + βLgfrtN + βLgconcN + σ_{est}	LgmlkNsur = μ + βLgfrtN + σ_{est}
Expenditurefrt = μ +βLgTUAA + βSR + βGD + βLgNfrt + βLgPfrt +σ_{est}	Expenditurefrtha = μ + βSR + βLgNfrt + βLgPfrt + σ_{est}
Lgconcexpenditure = μ + βLgTUAA + βGD + βSR + βLgconcN + σ_{est}	Lgconcexpenditureha = μ + βLgTUAA + βSR + βLgconcN + σ_{est}
Lgfeedexpenditure = μ + βLgTUAA + βGD + βSR+ βLgfeedP + σ_{est}	Lgfeedexpenditureha = μ + βLgTUAA + βSR+ βLgfeedP + σ_{est}
Lgenvpay = μ + βNsurha + βLgNUE + βLgPbal + βLgPUE + σ_{est}	NS
NP = μ +βLgTUAA + βSR + βGD + βmilkrec + βSFP + βLgenvpay + βexpenditurefrt + βLgconcexpenditure + βNsur + LgNUE + σ_{est}	NP = μ + βmilkrec + βexpenditurefrt + σ_{est}
NP = μ + βLgTUAA + βSR + βGD + βmilkrec + βSFP + βLgenvirpay + βexpenditurefrt + βLgfeedexpenditure + βLgPbal + LgPUE + σ_{est}	NS

LgmlkNsur, log transformed milk produced per kg N surplus; LgTUAA, log transformed total utilized agricultural area; SR, stocking rate; GD, number of grazing days; LgFPCMha, log transformed fat and protein corrected milk per ha; LgfrtN, log transformed mineral fertiliser nitrogen input; LgconcN, log transformed concentrate N input; LgfrtP, log transformed mineral fertiliser phosphorus input; LgfeedP, log transformed feed P input; milkrec, milk receipts; expenditurefrt, fertiliser expenditure; Lgconcexpenditure, log transformed concentrate expenditure; Lgfeedexpenditure, log transformed feed expenditure; NP, net profit; SFP, single farm payments; Lgenvirpay, log transformed agri-environmental payments; Nsur, N surplus; LgNUE, log transformed N use efficiency; LgPbalha, log transformed P balance; LgPUE, log transformed P use efficiency.

5.3. Results

5.3.1. Economic implications of nitrogen and phosphorus use efficiency and net profit

Mean net profit was €598 ha^{-1}, with no significant differences between farms, but with significant differences between years, ranging from €135 ha^{-1} to €958 ha^{-1} (Table 5.4.).

There was a significant relationship ($R^2 = 0.50$; $P < 0.001$) between mean net profit and mean milk receipts ($\beta = 0.79$) and fertiliser expenditure ($\beta = -0.10$) (Table 5.3.). An increase of €58 ha^{-1} in mean milk receipts was associated with an increase of €49 ha^{-1} in net profit. An increase of €6 ha^{-1} in mean fertiliser expenditure was associated with a decrease of €49 ha^{-1} in net profit.

There was no significant relationship found between mean net profit and mean N surplus, NUE, P balances and PUE (Table 5.3.).

Mean feed expenditure was €256 ha^{-1} (Table 5.4.), with significant differences between farms and years, ranging from €152 ha^{-1} to €387 ha^{-1} and €226 ha^{-1} to €323 ha^{-1} for farms and years, respectively (Table 5.4.).

Mean fertiliser expenditure was €201 ha^{-1}, with significant differences between farms, ranging from €111 ha^{-1} to €286 ha^{-1}, and no significant differences between years (Table 5.4.).

Table 5. 4. Mean values (and standard errors), grand means between years, ranges between farms, and coefficients of variation for milk receipts, agri-environmental payments, mineral fertilisers and feeds (concentrates, fodders) expenditures, net profit per ha and cost of compliance for 19 Irish dairy farms between 2009 and 2011; milk receipts for average national Irish dairy farms; net profit per ha for average national Irish dairy farms; standard error of the means for transformed data in brackets; P-values from ANOVA are included

	Year			Grand mean	S.E.M.	Range farms	Coeff. variation	*P*-value	
	2009	2010	2011					Y	F
Milk receipts (€ ha^{-1})	1,517	2,342	2,406	2,088	91.27	1,349-2,960	0.22	<0.05	<0.05
National average milk receipts (€ ha^{-1})	1,344	1,866	2,213	1,807	-	-	-	-	-
Agri-environmental payments (€ ha^{-1})	199	179	125	170	24.78(0.04)	23-526	0.70	NS	<0.01
Expenditure mineral fertiliser (€ ha^{-1})	172	200	212	201	18.95	111-286	0.27	NS	<0.001
Expenditure feeds (€ ha^{-1})	226	323	231	256	29.14(0.02)	152-387	0.32	<0.01	<0.01
Net profit (€ ha^{-1})	135	642	958	598	73.91	74-1261	-	<0.001	NS
National net profit (€ ha^{-1})	499	507	1251	752	-	-	-	-	-
Cost of compliance (€ ha^{-1})	1,771	1,874	2,292	1,900	361(0.13)	25-5826	-	NS	=0.07

S.E.M., standard error of the means; Coeff., coefficient; Y, year; F, farm; NS, not significant.

There was a significant positive relationship (R^2 = 0.56; P < 0.001) between mean expenditure on mineral N and P fertilisers and mean SR (β = 0.11), fertiliser N input (β = 0.62) and fertiliser P input (β = 0.23) (Table 5.3.). An increase of 0.03 LU ha^{-1} in mean SR, 4 kg N ha^{-1} in mean fertiliser N input and 0.70 kg P ha^{-1} in mean fertiliser P input was associated with an increase of €6 ha^{-1} in mean fertiliser expenditure.

There was also a significant relationship (R^2 = 0.67; P < 0.001) between mean concentrate expenditure and mean TUAA (β = -0.12), SR (β = 0.32) and concentrate N input (β = 0.59) (Table 5.3.). An increase of 1.68 ha in TUAA was associated with a decrease of €8.35 ha^{-1} in mean concentrate expenditure. An increase of 0.03 LU ha^{-1} in mean SR and 0.95 kg N ha^{-1} in concentrate N input was associated with an increase of €8.35 ha^{-1} in concentrate expenditure.

There was a significant relationship (R^2 = 0.72; P < 0.001) between mean feed (concentrates and fodders) expenditure and mean TUAA (β = -0.31), SR (β = 0.35) and feed P input (β = 0.52) (Table 5.3.). An increase of 1.68 ha in mean TUAA was associated with a decrease of €9.30 ha^{-1} in mean feed expenditure. An increase of 0.03 LU ha^{-1} in mean SR, and 0.36 kg P ha^{-1} in mean feed P input was associated with an increase of €9.30 ha^{-1} in mean feed expenditure.

5.3.2. Sensitivity analysis

Mean net profit was most sensitive to changing milk price and, to a lesser extent, changing fertiliser N price (Table 5.5.). In comparison with the actual mean net profit between farms for years 2009-2011, there was an average increase of 81 % or €549 ha^{-1} in the scenarios (S1, S4, S7) with high (€0.360 $litre^{-1}$) milk price and an average decrease of 90 % or €-212 ha^{-1} in net profit in the scenarios (S3, S6, S9) with low (€0.246 $litre^{-1}$) milk price (Table 5.5.). In the scenarios with medium milk and fertiliser N prices (S2, S5, S8), there was an average increase of 29 % or €208 ha^{-1} in net profit as compared to the actual mean net profit for years 2009-2011 (Table 5.5.).

Table 5. 5. Changes (and 3 year mean values) in net profit (€ ha^{-1}) relative to the actual 3 year mean net profit across nine price scenarios (S) combining changing milk and fertiliser prices for 19 Irish dairy farms between 2009 and 2011; mean values of changes in net profit for all farms for each scenario are included

Farm	S1	S2	S3	S4	S5	S6	S7	S8	S9
1	325	-22	-452	344	-4	-434	354	7	-423
2	178	-206	-680	202	-181	-655	224	-160	-633
3	665	240	-285	693	267	-258	717	291	-234
4	499	121	-344	525	147	-318	541	164	-302
5	410	157	-157	423	169	-144	431	177	-136
6	832	521	136	846	534	149	857	546	161
7	1,331	720	-35	1,362	751	-3	1,383	772	17
8	1,004	575	46	1,031	602	72	1,054	625	95
9	91	-274	-724	119	-246	-696	143	-222	-672
10	572	350	77	586	365	91	599	377	104
11	406	153	-160	426	173	-140	444	190	-123
12	739	381	-61	757	399	-43	768	410	-32
13	678	360	-32	690	373	-19	698	381	-12
14	498	220	-122	517	240	-102	534	257	-85
15	512	271	-26	525	284	-13	536	295	-2
16	-282	-554	-892	-256	-530	-867	-240	-514	-851
17	202	-196	-686	219	-178	-669	234	-163	-654
18	882	494	15	909	521	42	925	537	59
19	534	292	-7	544	302	3	550	308	9
Mean (€/ha)	**530**	**190**	**-231**	**551**	**210**	**-211**	**566**	**225**	**-195**
Change (%)	**79**	**24**	**89**	**82**	**30**	**90**	**83**	**34**	**92**

When comparing the six farms with the lowest mean milk receipts (€1,554 ha^{-1}) (S. D. = 187) to the six farms with the highest mean milk receipts (€2,559 ha^{-1}) (S. D. = 319), it was noticed that the net profit of the former would increase by €503 ha^{-1} in the scenarios with high milk price and would decrease by €-87 ha^{-1} in the scenarios with low milk price, while the net profit of the latter would increase by €653 ha^{-1} and decrease by €-312 ha^{-1}.

5.3.3. Economic aspects of complying with the Nitrate Directive

Economic aspects relating to compliance with the Nitrate Directive considered in this study were the N-eco-efficiency or the amount of milk produced per kg N surplus, agri-environmental payments, N fertiliser replacement value of slurry (NFRV), and cost of compliance.

The mean amount of milk produced per kg N surplus was 51.4 litres kg N^{-1} (Table 5.6.), with significant differences between farms, ranging from 30.3 to 84.8 litres kg N $surplus^{-1}$, and no significant differences between years (Table 5.6). Values close to the lower end of this range (36.4 litres kg N surplus^{-1}) were recorded on four farms, while values close to the higher end (83.0 litres kg N surplus^{-1}) were recorded on two farms. There was a significant positive relationship ($R^2 = 0.54$; $P < 0.001$) between mean milk produced per kg N surplus and mean fertiliser N input (Table 5.3.). An increase of 4 kg N ha^{-1} in mean fertiliser N input was associated with an increase of 1.90 litres kg N^{-1} in milk produced per kg N surplus.

Table 5. 6. Mean values (and standard errors), grand means between years, coefficients of variation, and ranges between farms for N and P inputs from mineral fertilisers, concentrates, feeds (concentrates and fodders), milk produced per kg N surplus, N and P balances and N and P use efficiencies for 19 Irish dairy farms between 2009 and 2011; standard error of the means for transformed data in brackets; P-values from ANOVA are included

	Year			Grand mean	S.E.M.	Range farms	Coeff. variation	*P*-value	
	2009	2010	2011					Y	F
Fertiliser N input (kg N ha^{-1})	163	211	196	190	8.82(0.022)	85- 278	0.32	NS	<0.001
Concentrate N input (kg N ha^{-1})	25.8	35.1	20.1	27.0	2.02(0.033)	7.3-60.8	0.49	<0.01	<0.05
N surplus (kg N ha^{-1})	139	195	166	167	8.75	51-286	0.35	<0.05	<0.001
NUE	0.27	0.25	0.26	0.26	0.010(0.015)	0.17 - 0.53	0.24	NS	=0.01
Milk per N surplus (litres kg N^{-1})	57.8	46.5	50.0	51.4	2.47(0.017)	30.3-84.8	0.28	NS	<0.01
Fertiliser P input (kg P ha^{-1})	8.52	8.59	6.41	7.76	0.927(0.049)	0.68-32.21	0.81	NS	<0.01
Feed P input (kg P ha^{-1})	4.77	10.44	6.81	7.33	0.689(0.041)	1.39-21.75	-	<0.01	NS
P balance (kg P ha^{-1})	-0.67	5.32	0.54	1.72	1.225(0.075)	-10.32 - +33.26	1.00	<0.05	0.01
PUE	1.38	1.02	1.16	1.08	0.102(0.037)	0.30-2.03	0.41	NS	<0.001

N, nitrogen; NUE, nitrogen use efficiency; P, phosphorus; PUE, phosphorus use efficiency; S.E.M., standard error of the means; Coeff., coefficient; Y, year; F, farm; NS, not significant.

Mean agri-environmental payments were €170 ha^{-1}, with significant differences between farms, ranging from €23 ha^{-1} to €526 ha^{-1}, and no significant differences between years (Table 5.4.).

There was no significant relationship between agri-environmental payments N surplus, NUE, P balance and PUE (Table 5.3.).

The average NFRV of slurry was €317 ha^{-1} (S.D. = 149), ranging from €123 ha^{-1} to €636 ha^{-1} between farms, for the spring (January-April) application, and €64 ha^{-1} (S.D. = 40), ranging from €12 ha^{-1} to €156 ha^{-1} between farms, for the summer (April-late July) application.

Mean cost of compliance was €1,900 ha^{-1}, with no significant differences between farms and years (Table 5.4.).

5.4. Discussion

5.4.1. Economic implications of nitrogen and phosphorus use efficiency and net profit

The increase in mean net profit with milk receipts and decrease, albeit to a much lesser extent, with mineral fertiliser expenditure imply that increasing milk receipts while maximising the use of mineral fertiliser input is an effective strategy to increase net profit. This would be an ideal situation from an environmental point of view, as decreases in inputs of mineral fertilisers and increases in milk exports, by improving management of herd genetic potential for example, may significantly contribute to decreases in N and P surpluses on farms. From an economic point of view, this situation can be difficult to achieve, because increased milk receipts could be attained through increased milk yields supported by increased imports of purchased feeds (Shalloo *et al.*, 2004*a*) and increased herbage yields supported by increased mineral fertilisers (Hennessy *et al.*, 2008), associated with increased expenditures and potentially decreased net profit. Therefore, controlling main input (fertiliser, feeds) expenditures while maintaining or increasing milk receipts would be more effective for maintaining or increasing net profit. Mean milk receipts and mineral fertiliser expenditure explained only 0.50 of the variation in mean net profit. The remaining variation could be explained by factors such as prices of inputs (mineral fertiliser, feeds, labour, seed, agro-chemicals) and outputs (sold milk and livestock), subsidies, and farmer's ability to identify and control the highest expenditures on their farms (Chamberlain, 2012).

The significant differences in mean net profit between years were partially due to a 13 % increase in the volume of milk sold off the farms and an inter-annual increase in milk price, from €0.246 litre^{-1} in 2009 to €0.360 litre^{-1} in 2011 (CSO, 2012) resulting in a 37 % increase in mean milk receipts. Similar trends were observed on average national Irish dairy farms, with milk receipts increasing from €1,344 ha^{-1} in 2009 to €2,213 ha^{-1} in 2011 and net profit increasing from €499 ha^{-1} in 2009 to €1,251 ha^{-1} in 2011 (Connolly *et al.*, 2009; Hennessy *et al.*, 2010; 2011). However, mean net profit in the current study is not very different than that of average national Irish dairy farms before the implementation of Nitrate Directive (£540 ha^{-1}; Burke and Roche, 2000) indicating that the existing farm practices, with regards to milk yields, nutrient management and associated expenditures, did not have a visible impact on farm profitability of Irish dairy farms between 1998 and 2009, at least. In the current study, the 37 % increase in mean milk receipts, 19 % decrease in mean expenditure on mineral fertilisers, and 32 % increase in milk price justified the 86 % increase in mean net profit between 2009 and 2011.

At European level, mean net profit in the current study (€598 ha^{-1}) was higher than 23 Scottish and English dairy farms (€444 ha^{-1}), 37 French dairy farms (€311 ha^{-1}) and 55 Spanish and Portuguese farms (€215 ha^{-1}) in Raison *et al.* (2006). It is noticeable that the net margin was the lowest on the Spanish and Portuguese farms, which were confined dairy farms, with zero-grazing, as opposed to the Scottish and English farms, which included grazed grass as an input for milk production. In a multi-annual project, Groot *et al.* (2006) compared the gross margin in the first year and the fourth year of 45 dairy farms grouped by their initial NUE values. Increases of €286 ha^{-1}, on average, for two groups of farms, were associated with increases in milk production and associated receipts (€310 ha^{-1}), on one group, and re-balancing or moderation of herbage and milk yields, achieved with moderate N inputs from mineral fertilisers and feeds and associated decreased expenditures (by €132 ha^{-1} for mineral N fertilisers and €283 ha^{-1} for concentrates and fodders), on the other group. It is notable that overall there was no direct link between observed increases in mean gross margin and NUE.

In the current study, mean net profit was not directly related to mean N and P surplus or N and P use efficiency. However, the relationships found between N and P use in the form of mineral fertilisers and feeds (concentrates and fodders), as components of N and

P balances, NUE and PUE, and the associated expenditures on mineral fertilisers and feeds, known as two main factors impacting on profitability of dairy farms (Tozer *et al.,* 2003; Donnellan *et al.,* 2011), indicate an indirect link between N and P use efficiency and net profit.

The increase in mean fertiliser expenditure with mean fertiliser N and P inputs and, to a lesser extent, with mean SR, suggests that decreasing SR and fertiliser N and P input may be an effective strategy to decrease fertiliser expenditure. However, on intensive dairy farms, concerned with production levels and profitability, a more achievable objective may be to optimise the use of external inputs such as mineral fertilisers (Arriaga *et al.*, 2009; Kelly *et al.*, 2013). Therefore, from an environmental point of view, SR can be maintained, or even increased, while decreasing N and P balances and improving NUE and PUE, if good management of N and P resources (fertilisers, feeds) and the overall dairy production system (grazing management, fertility management, management of herd genetic potential) are in place. From an economic point of view, controlling fertiliser expenditures while maintaining or increasing milk receipts would be more effective for maintaining or increasing net profit.

In the current study, SR and fertiliser N and P input explained only 0.56 of the variation in fertiliser expenditure. The remaining variation may be explained by factors such as levels of applied organic N and P, advisory and planning, and economic considerations.

The decrease in mean concentrate expenditure with mean TUAA and the increase with mean SR and concentrate N input implies that increasing TUAA while decreasing SR and concentrate N input are effective strategies to decrease concentrate expenditure. While an increase in area is very unlikely to happen, considering the low availability and high cost of agricultural land in Ireland (Donnellan *et al.*, 2011; Patton *et al.,* 2012; Kelly *et al.,* 2013), a decrease in SR and concentrate input can be associated with a decrease in milk production. For example, a decrease of 0.10 cows ha^{-1} in SR and 387 kg DM cow^{-1} in concentrate input were associated with a decrease of 846 kg ha^{-1} in milk yield (Shalloo *et al.*, 2004a). To avoid decreases in SR and milk yields, the efficiency of concentrate use and associated expenditures can be improved. For example, Buckley and Carney (2013) reported excessive concentrate N inputs of 7.44 kg N LU^{-1} with an associated expenditure of €84 LU^{-1} on 89 Irish specialist dairy farms. This emphasises the importance of matching animal feed requirements and concentrate imports to maximise nutrient utilisation and farm profit. In the current study, mean TUAA, SR, and concentrate N input explained only 0.67 of the variation in concentrate expenditure.

The remaining variation may be explained by factors such as targeted milk yields, grazed grass and silage intake and market price of purchased concentrates. This highlights significant potential to decrease concentrate N use and associated expenditures.

Increased concentrate expenditure associated with increased concentrate N input was recorded also on 23 Scottish and English dairy farms including grazed grass as an input for milk production and 55 confined, zero-grazing dairy farms from Spain and Portugal in Raison *et al.* (2006). The Scottish and English farms had an average concentrate N input of 59 kg N ha^{-1} and an average concentrate expenditure of €449 ha^{-1}, compared with 386 kg N ha^{-1} and €1,593 ha^{-1} on the Spanish and Portuguese farms and 26 kg N ha^{-1} and €243 ha^{-1} in the current study. The differences in concentrate N inputs and the associate expenditures might have impacted on the net profit, as the Spanish and Portuguese farms had €229 ha^{-1} lower net profit compared to the Scottish and English farms, while all the farms had €268 ha^{-1}, on average, lower net profit than the Irish farms in the current study. This reflects the lower input Irish dairy system, with low use of concentrates and high reliance on grazed grass.

The decrease in mean feed expenditure with mean TUAA, and the increase with mean SR and feed P input implies that increasing TUAA while decreasing SR and feed P input may be effective strategies to decrease feed expenditure. While an increase in area is very unlikely to happen, considering the low availability and high expenditure of agricultural land in Ireland, a decrease in SR and feed input can be similarly associated with a decrease in milk production. Potential decreases in feed P input should be made by taking into consideration economic milk yields, for which dairy cows need between 3.2 and 4.2 g P kg^{-1} of concentrate (Steinshamn *et al.*, 2004) and 3.5 mg P kg DM^{-1} silage (Haygarth *et al.,* 1998). For example, Huhtanen *et al.* (2011) reported milk yields of 7,000 kg cow^{-1} at a dietary P concentration of 4.25 g kg DM^{-1} supplement. One alternative to decreasing feed P inputs would be increasing the proportion of home grown supplements in animals' diet (Lawrie *et al.,* 2004). In the current study, mean TUAA, SR and feed P input explained only 0.72 of the variation in feed expenditure. The remaining variation may be explained by factors such as the intake of home grown supplements, prices of purchased feeds and types of purchased feeds included in animals' diet, considering also that P supplements are the third most costly diet ingredients after grain and protein supplements (Satter *et al.,* 2005). However, the significance (R^2 = 0.72) of the relationship highlights the importance of matching SR and animal feed requirements with the feed imports on grass-based dairy farms, when there is limited availability of grassland area (McCarthy *et al.,* 2007), as the most effective strategy to control the feed expenditures, with potential positive impact on net profit, on dairy farms.

5.4.2. Sensitivity analysis

The results of the sensitivity analysis indicated that milk price is the main driver for changes in net profit between 2009 and 2011, both in high and low milk price situations, in the current study. One limitation of this analysis is that it captured more the inter-annual changes for all farms but not the specific differences between farms in terms of practices relating to milk yields and exports, use of mineral fertilisers, N and P balances and N and P use efficiencies on farms.

The sensitivity of net profit to changes in milk price was illustrated by the comparable changes in net profit relative to the actual net profit of the six farms with the highest milk receipts (€2,559 ha^{-1}) and the six farms with the lowest milk receipts (€1,554 ha^{-1}). The net profit of the farms with highest milk receipts would increase by €653 ha^{-1} in high milk price situations but it would decrease by €-312 ha^{-1} in low milk price situations. In comparison, for the same scenarios, the net profit of the farms with lowest milk receipts would relatively increase by €503 ha^{-1} and decrease by only €-87 ha^{-1}, respectively. The differences in net profits between the two groups of farms were mostly associated with differences in the expenditures on mineral fertilisers. The farms with highest milk receipts had an average expenditure on mineral fertiliser of €250 ha^{-1} compared with the farms with lowest milk receipts, with an average expenditure of €159 ha^{-1}. This meant a considerable difference in fertiliser expenditure, of almost €100 ha^{-1}, between the two groups of farms. Increased financial loss (€-416 ha^{-1}) for farms with highest input expenditures compared with farms with low input expenditures (€-314 ha^{-1}) in a low milk price situation (€0.20 litre^{-1}) were also reported by Patton *et al.* (2012). This highlights an increased vulnerability of higher input and output systems to periods of low milk price and again emphasises the importance of minimising input expenditures to improve both economic and environmental sustainability. In contrast, Moreau *et al.* (2012) found no sensitivity to variation in milk and mineral fertiliser N prices for clover/grass-based French dairy systems relying on N inputs from biological N fixation (64 kg N ha^{-1}) via clover plants. This autonomy resulted in minimal N inputs in the form of mineral fertilisers (12 kg N ha^{-1}) but it was not associated with increases in net profit.

If making analogy with the baseline situation of the two groups of farms, year 2009 - associated with lowest milk price (€0.246 $litre^{-1}$; CSO, 2012) - can be considered as an example of low milk price situations, year 2010 - with lowest mineral fertiliser N price (€0.825 kg N^{-1}; CSO, 2012) - a low fertiliser N price situations, and year 2011 - with highest milk (0.360 $litre^{-1}$; CSO, 2012) and mineral fertiliser N price (€1.029 kg N^{-1}; CSO, 2012) - both high milk and fertiliser N prices situations. On farms, the increase in milk price was reflected by increases in the volume of milk produced and sold off the farms, from 5,837 litres ha^{-1} in 2009 to 6,146 litres ha^{-1} in 2011 for the farms with the lowest milk receipts and from 8,360 litres ha^{-1} in 2009 to 9,176 litres ha^{-1} in 2011 for the farms with the highest milk receipts. The associated milk receipts relatively increased from €1,086 ha^{-1} in 2009 to €1,874 ha^{-1} in 2011 for first group and from €1,683 ha^{-1} in 2009 to €3,045 ha^{-1} in 2011 for the second group.

Comparatively, the average expenditure on mineral fertilisers did not follow the same trend as the fertiliser N price, with similarly high values in 2010 and, unexpectedly, in 2011 (€166 ha^{-1} on average) and lowest average value in 2009 (€147 ha^{-1}) for the farms with the lowest milk receipts, and €276 ha^{-1} on average, for 2010 and 2011, and €198 ha^{-1} in 2009 for the farms with the highest milk receipts. These expenditures were associated with fertiliser N inputs showing similar trend, at 160 kg N ha^{-1}, on average, in 2010 and 2011 and 143 kg N ha^{-1} in 2009 for the farms with lowest milk receipts and 263 kg N ha^{-1}, on average, in 2010 and 2011, and 207 kg N ha^{-1} in 2009 for the farms with the highest milk receipts. The average fertiliser P input was the greatest in 2010 (10.33 kg P ha^{-1}) compared with 6.90 kg P ha^{-1} in 2009 and 9.17 kg P ha^{-1} in 2011 for the farms with the lowest milk receipts. Conversely, it decreased from 7.79 kg P ha^{-1} in 2009 to 5.63 kg P ha^{-1}, on average, in 2010 and 2011, for the farms with the highest milk receipts. However, the N and P fertiliser inputs on both groups of farms were mostly influenced by factors (SR, use of organic fertilisers, soil P status, farm advice and environmental legislation), aiming at reducing N and P surpluses and increase NUE and PUE on farms, rather than by market prices. The average N surplus was the highest in 2010 for both the farms with lowest milk receipts (157 kg N ha^{-1}) and highest milk receipts (242 kg N ha^{-1}), was lowest in 2009 (130 kg N ha^{-1} for the first group and 177 kg N ha^{-1} for the second group), and intermediate in 2011 (126 kg N ha^{-1} for the first group and 226 kg N ha^{-1} for the second group). Similarly, the average P balance was the highest in 2010 for both groups of farms (+7.53 kg P ha^{-1} for the first group and +0.96 kg P ha^{-1} for the second group), lowest in 2009 (+0.51 kg P ha^{-1} for the first group and -2.88 kg P ha^{-1} for the second group), and intermediate in 2011 (+4.22 kg P ha^{-1} for the first group and -2.85 kg P ha^{-1} for the second group). The highest values of N and P farm-gate balances in 2010 coincided with the lowest fertiliser price, underlining the role of mineral fertiliser prices on N and P management on the two groups of farms. However, NUE gradually increased from 0.23 to 0.31 on the farms with lowest milk receipts between 2009 and 2011 and decreased from 0.25 in 2009 to 0.22 in 2010 and 2011 for the farms with highest milk receipts. Mean PUE was the lowest in 2010 for both groups of farms (0.74 for the first group and 1.12 for the second group), highest in 2009 (1.19 for the first group and 1.41 for the second group), and intermediate in 2011 (0.92 for the first group and 1.27 for the second group). The values of PUE in 2010 partially reflected the lowest mineral fertiliser price occurring in the same year.

The average net profit relatively increased from €-3 ha^{-1} in 2009 to €765 ha^{-1} in 2011 on the farms with lowest milk receipts and from €195 ha^{-1} in 2009 to €1,209 ha^{-1} in 2011 on the farms with the highest milk receipts. The trend of the net profit was similar to the milk receipts and milk price. However, milk receipts relatively increased by 43 %, on average, on the farms with lowest milk receipts and by 45 %, on average, on the farms with the highest milk receipts, while net profit relatively increased by 100 % on the former and 84 % on the latter, between 2009 and 2011. This indicates that an increase in milk receipts, in the context of volatile milk and mineral fertiliser prices, is not sufficient to ensure the economic sustainability of dairy farms in the long term. Moreover, the existence of many producers competing for the sale of milk means that in today's market, there is limited possibility for dairy farmers to influence the price they receive. Therefore, reducing the cost of production is considered the primary management strategy available to dairy farmers for obtaining any increase in profits (Von Keyserlingk *et al.*, 2013). They can, for example, control the most impacting expenditures on their farms, such as the expenditure on mineral fertiliser.

In the current study, the results of the sensitivity analysis indicate that variation of milk price was associated with relative increases in milk receipts and net profit in the long term. Comparatively, the variation in fertiliser N price was reflected mostly by N inputs and N balances only during one year (2010). This indicates a negative impact of this variation on both economic (through increased expenditure on mineral fertiliser) and environmental (through higher N balance) sustainability in the short term. In the long term, economic sustainability may be improved by controlling expenditure on mineral fertilisers, while the environmental sustainability may be improved by reducing nutrient surpluses on farms. These goals, corresponding to the multi-functional demands now being placed on agricultural sector worldwide (Crosson *et al.*, 2007), can be satisfied, to some extent, on dairy farms through the introduction of white clover in swards. This can help both lower expenditures on mineral N fertilisers, by €148 ha^{-1}, and decrease farm-gate N balances and risks of environmental impacts attributable to N losses, due to the replacement of mineral fertiliser N by biologically fixed N via white clover (Humphreys *et al.,* 2012). This will likely have the added advantage of tackling the expected potential financial insecurity of dairy farmers in the context of milk quota abolition leading to an expected increase in milk supply and milk price volatility (Kelly *et al,*. 2012; Geary *et al.,* 2012), and a concurrent increase in the price of mineral fertilisers (Peyraud *et al.,* 2010).

5.4.3. Economic aspects of complying with the Nitrate Directive

The N-eco-efficiency is another factor that can impact on decisions about fertiliser N inputs and associated expenditure on further implications on farm-gate N balances and net profit on dairy farms. In a comparative study on farm-gate N surplus and NUE on Flemish and European specialist dairy farms, Nevens *et al.* (2006) found that the threshold on Flemish soil types and climates, of maximum 150 kg N ha^{-1} for N surplus, considered safe for complying with the limit of <50 mg NO_3 $litre^{-1}$ in the groundwater, can be attained at production levels of up to 10,000 litres milk ha^{-1}. A target value of 85 (range: 60-110) litres milk kg N $surplus^{-1}$ was also established.

In the current study, there was no obvious differentiation between more intensive or more extensive farms, due to the large differences in mean N-eco-efficiency between farms (30.3-84.8 litres kg N $surplus^{-1}$) being associated with large differences in mean fertiliser N input (85-278 kg N ha^{-1}), with which it was significantly related. The increase in mean milk produced per kg N surplus with mean fertiliser N input implies that efficient on-farm N management of fertiliser N inputs, aiming at supporting herbage and therefore milk production while making efficient use of N, is an effective strategy to achieve increases in milk production and also reduce N surplus per unit product (litre milk). This may be achievable by optimising management aspects such as grazing management, grass utilisation (O'Donovan *et al.,* 2002; Kennedy *et al.,* 2005), management of all on-farm nutrient sources (Peyraud and Delaby, 2006), improved N recycling between soil, pasture, animals, and milk and livestock for export (Nielsen and Kristensen, 2005), and management of herd genetic potential (Berry *et al.,* 2007). However, the environmental pressure per unit area under production, as indicated by N surplus per ha, may actually still increase under such a scenario.

In the current study, the adjustments in operational management towards improved on-farm management of N imported as mineral fertiliser on farms, associated with decreased N surplus, while increasing milk production and therefore milk exports off farms, may contribute to increases in net profit between farms, which was significantly related to milk receipts and expenditure on mineral fertilisers, but also between years, as milk receipts accounted for 37 % of inter-annual increase in net profit. Therefore, N-eco-efficiency can be considered as another metric to measure progress towards improved economic and environmental sustainability. However, mean fertiliser N input explained only 0.54 of the variation in mean amount of milk produced per kg N surplus. The remainder can be mainly explained by the differences in milk yields and on-farm N management resulting in variable farm-gate N surpluses, similar to other studies.

Mean N-eco-efficiency (51 litres milk kg N surplus^{-1}) in the current study was similar to a range of Irish dairy production systems (48 litres milk kg N surplus^{-1}; Dillon and Delaby, 2009) and close to the mean for progressive Flemish farms (60 litres milk kg N surplus^{-1}) in Nevens *et al.* (2006). However, mean N-eco-efficiency in the current study was much lower than the mean for New Zealand dairy farms (77 litres milk kg N surplus^{-1}) in Beukes *et al.* (2012). The different values for N-eco-efficiency reflect the differences in operational management, with lower mean milk yields on Irish farms (7,792 litres ha^{-1}, current study; 7,736 litres ha^{-1}, Dillon and Delaby, 2009), compared with the progressive Flemish farms (10,000 litres ha^{-1}, Nevens *et al.*, 2006) and New Zealand farms (12,000 litres ha^{-1}, Beukes *et al.*, 2012). Also, the mean N surplus on the Irish farms (167 kg N ha^{-1}, current study; 162 kg N ha^{-1}, Dillon and Delaby, 2009) was similar to the Flemish farms (163 kg N ha^{-1}, Nevens *et al.*, 2006) and slightly higher than the New Zealand farms (155 kg N ha^{-1}, Beukes *et al.*, 2012). This also reflects the operational management on Irish dairy farms, with lower N inputs and balance but also lower milk yields compared to dairy farms worldwide.

Accounting for NFRV of slurry applied to grassland is another opportunity for reducing the fertiliser N input and associated expenditure (Lalor, 2008), with potential positive impact on the farm-gate N balance and net profit. The associated economic values of

€317 ha^{-1} for the spring and €64 ha^{-1} for summer application should motivate the farmers to make increased use of organic fertilisers throughout the 'open' slurry application period during the year, as part of the overall on-farm N management.

Differences in agri-environmental payments between farms were associated with differences in the environmental plans for the farms participating in REPS. Important conditions to receive the financial support referred to the limit of 2 LU ha^{-1}, up to 2.9 LU ha^{-1} only if having approval for derogation, and application of mineral fertilisers according to a fertiliser plan drawn by an approved planning agency (DAFM, 2013a). Mean agri-environmental payments (€170 ha^{-1}) in the current study were much higher than similar payments received on dairy farms in the UK (£44 ha^{-1}) (Wilson *et al.,* 2013). The fact that there was no significant relationship between mean agri-environmental payments and N and P surpluses, was most likely due to the great deviation in inputs between farms in the current study.

However, compliance with environmental regulations may also be associated with increases in net profit on farms. In the current study, the nine farms exceeding the limit of 2 LU ha^{-1}, imposed through the Nitrates Directive, had 1.63 times higher net profit (€606 ha^{-1}) compared with the remainder (€371 ha^{-1}), which justified the cost of compliance associated with being in derogation. Mean cost of compliance in the current study (€1,900 ha^{-1}) was similar to the average national Irish dairy farms (€2,000 ha^{-1}, Hennessy *et al.,* 2005). Similarly, Patton *et al.* (2012) reported 1.03 times higher net profit for 0.47 LU ha^{-1} higher SR above the 2.0 LU ha^{-1} limit. In contrast, McCarthy *et al.* (2007) found that an increase of 0.27 LU ha^{-1} above this limit was not associated with an increase in farm net profit. These differences partially support the argument of Brennan and Patton (2010) that in grazed grass-based dairy production systems, increases in SR can determine increases in farm profitability as long as there is a good match between SR and the grass growing potential of the farm, to allow increased grass utilisation, with no major additional imports of either concentrate or mineral fertilisers and associated increased expenditures.

5.5. Conclusions

Between farms, the increase in mean net profit with mean milk receipts and the decrease with mean expenditure on mineral fertiliser N use implies that increasing milk receipts while maximising the use of mineral fertiliser input is an effective strategy to increase net profit. Between years, the 86 % increase in mean net profit was not fully explained by the 37 % increase in mean milk receipts, indicating that an increase in milk receipts is not enough to ensure the financial security of dairy farmers in the long term. Mean net profit was not directly related to mean N and P surplus or N and P use efficiency. However, there was an indirect link between net profit and N and P use efficiency, as indicated through significant relationships between N and P use in the form of mineral fertilisers and feeds and the associated expenditures on mineral fertilisers and feeds. The most significant relationship indicated that mean expenditure on feeds increased with mean SR and feed P input and decreased with TUAA. This highlights the importance of matching SR and animal feed requirements with the feed (concentrate, fodders) imports on grass-based dairy farms, when there is limited availability of grassland area, as the most effective strategy to control the feed expenditures, with potential positive impact on net profit, on dairy farms.

The results of the sensitivity analysis indicated that milk price was the main driver for changes in net profit between 2009 and 2011, both in high and low milk price situations investigated across nine price scenarios. The larger changes in net profit relative to the actual net profit on the six farms with highest milk receipts compared to the six farms with the lowest milk receipts in low milk price situations was mainly associated with higher expenditures on mineral N fertilisers on the former. This highlights an increased vulnerability of higher input systems to periods of low milk price and emphasises the importance of minimising input (mineral fertiliser) expenditures to improve economic sustainability.

The compliance with Nitrate Directive can be associated with both advantages and disadvantages. The increase in mean milk produced per kg N surplus with mean fertiliser N input implies that efficient on-farm N management of fertiliser N inputs, aiming at supporting herbage and therefore milk production while making efficient use of N, is an effective strategy to achieve increases in milk production and therefore reduce N surplus per unit product (litre milk). However, the environmental pressure per unit area under production, as indicated by N surplus, may actually increase with fertiliser N input. Potential fertiliser N replacement values of €317 ha^{-1} for the spring and €64 ha^{-1} for summer slurry application may represent strong incentives for farmers to make increased use of organic fertilisers, as part of overall on-farm N management, with positive impacts on farm nutrient use efficiency and farm net profit. Nine farms exceeding the limit of 2 LU ha^{-1}, imposed through the Nitrates Directive, had 1.63 times higher net profit compared with the remainder, which justified the cost of compliance associated with being in derogation.

The results of this study generally indicate that Irish dairy farms, as low input production systems, have the potential to achieve both economic (as indicated by net profit per ha) and environmental (as indicated by N and P balances per ha, N and P use efficiency and N-eco-efficiency) sustainability.

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6. An economic comparison of systems of dairy production based on N fertilised grass and grass-white clover grassland in a moist maritime environment

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J. Humphreys, E. Mihailescu, and I. A. Casey (2012) An economic comparison of systems of dairy production based on N-fertilised grass and grass-white clover grassland in a moist maritime environment. *Grass and Forage Science* 67, pp. 519-525.

Conference papers:

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Abstract

This study compared the profitability of systems of dairy production based on N fertilised grass (FN) and grass-white clover (WC) grassland and assessed sensitivity to changing fertiliser N and milk prices. Data were sourced from three system-scale studies conducted in Ireland between 2001 and 2009. Ten FN stocked between 2.0 and 2.5 LU ha^{-1} with fertiliser N input between 173 and 353 kg ha^{-1} were compared with eight WC stocked between 1.75 and 2.2 LU ha^{-1} with fertiliser N input between 79 and 105 kg ha^{-1}. Sensitivity was confined to nine combinations of high, intermediate and low fertiliser N and milk prices. Stocking density, milk and total sales from WC were approximately 0.90 of FN. In scenarios with high fertiliser N price combined with intermediate or low milk prices WC was more ($P<0.05$) profitable than FN. Based on milk and fertiliser N prices at the time, FN was clearly more profitable that WC between 1990 and 2005. However, with the steady increase in fertiliser N prices relative to milk price, the difference between FN and WC was less clear cut between 2006 and 2010. Projecting into the future assuming similar trends in fertiliser N and milk prices to that in last decade, this analysis indicates that WC will become an increasingly more profitable alternative to FN for pasture based dairy production.

6.1. Introduction

Over the last ten years, the farm-gate cost of fertiliser N in Ireland has been increasing at an annual rate of around 9% (Fig. 6.1a) due to growing demand worldwide and rising manufacturing costs (Prince *et al.*, 2009). In contrast, milk price in Ireland, while variable, has been relatively static (Fig. 6.1b). Hence, there has been a strong increase in the cost of fertiliser N relative to the farm-gate price received for milk (Fig. 6.1c). This is negatively impacting on profitability of pasture based systems of dairy production, which are highly reliant on input of fertiliser N. At the same time in the European Union and in other parts of the world there has been increasing regulatory pressure to lower N losses to water and to the atmosphere, for example, various national regulations stemming from the Nitrates Directive, Water Framework Directive and the National Emission Ceilings Directive (European Council, 1991; European Parliament and Council, 2000; 2001). In general, white clover based systems (WC) are associated with lower stocking densities, higher N use efficiency, lower surplus N per hectare, lower losses of nitrate to water and of ammonia and nitrous oxide (a potent greenhouse gas) to the atmosphere than N-fertilised grass based systems (FN) (Jarvis *et al*, 1996; Hooda *et al.*, 1998; Rochon *et al.*, 2004; Schils *et al.*, 2005; Humphreys *et al.*, 2008; Ledgard *et al.*, 2009). These differences can be largely attributed to lower N fluxes associated with the generally lower productivity of WC.

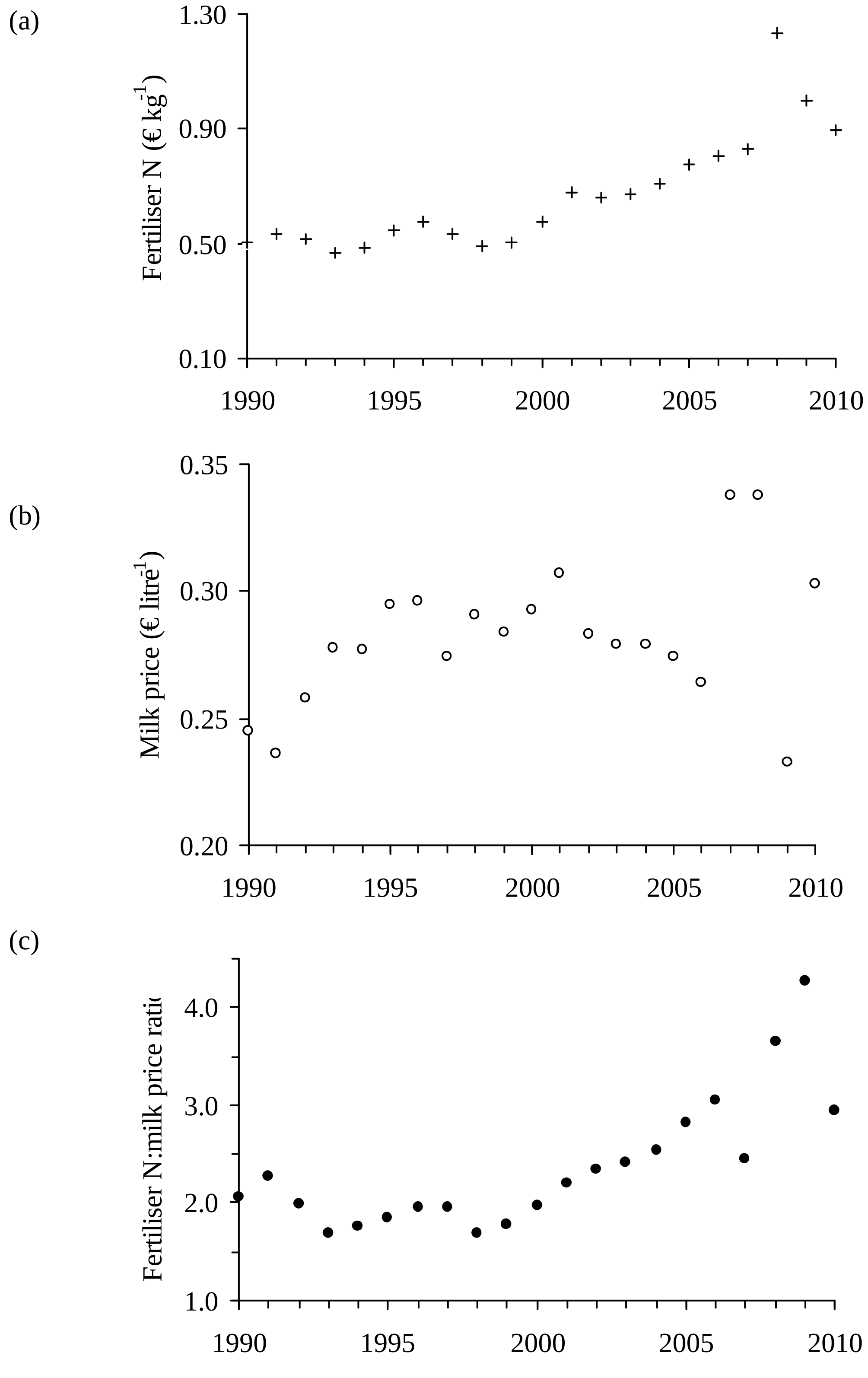


Fig. 6. 1. Prices in Ireland between 1990 and 2010 (CSO, 2010) of (a) fertiliser N (weighted average of calcium ammonium nitrate and urea); (b) milk and (c) the ratio between fertiliser N and milk price (Mihailescu E.).

In a review on grassland based dairy systems, Andrews *et al*. (2007) concluded that the herbage production of WC was approximately 0.70 of FN receiving 400 kg N ha^{-1}. Likewise, in studies in Scotland (Leach *et al*., 2000), Ireland (Ryan, 1986), and New Zealand (Ledgard *et al*., 1998), WC receiving no input of fertiliser N had approximately 0.80 of the milk production of FN receiving over 350 kg N ha^{-1}. In these studies, the net margins per hectare of WC were between 0.65 and 0.95 of higher stocked FN. However, these studies were conducted when the cost of fertiliser N was low relative to the farm-gate price received for milk. Furthermore, the levels of annual fertiliser N input in these studies are no longer permitted in many European countries. For example, in Ireland, under Statutory Instruments (SI) No. 610 (2010), stocking density on dairy farms is limited to 2 dairy cows per ha and the maximum allowed fertiliser N input per hectare is 200 kg ha^{-1} unless derogation has been granted subject to specific requirements, which allows individual farmers to carry higher stocking densities of dairy cattle per ha and fertiliser N use up to a maximum of 280 kg ha^{-1}.

Hence there is a need to evaluate the potential of white clover to replace fertiliser N and contribute to the profitability of pasture based systems of dairy production. In the present study, data from three system-scale studies conducted in Ireland (Humphreys *et al*., 2008; 2009; Keogh *et al*., 2010) were combined with recent farm gate input and output prices to determine the relative profitability of WC in the context of recent fertiliser N and milk prices and changing price conditions, and in the context of current and possible future restrictions on fertiliser N use and stocking densities on farms.

6.2. Materials and Methods

6.2.1. Site characteristics and weather conditions

Production data were derived from three previous studies (Humphreys *et al*., 2008; 2009; Keogh *et al*., 2010) conducted at Solohead Research Farm (52˚51'N, 08˚21'W) between 2001 and 2009. Soils on the farm are poorly drained Gleys (90%) and Grey Brown Podzolics (10%) with clay loam texture overlaying Devonian sandstone. Elevation ranges from 148.5 to 155.5 m OAD. Topographic relief causes variation in shallow ground water table depth (0 - 2.2 m below ground level (bgl)) and in drainage status in different parts of the farm. Much of the farm is seasonally wet and waterlogged during periods of high rainfall. The local climate is maritime with a long potential growing season. Rainfall was measured according to Fitzgerald and Fitzgerald (2004).

6.2.2. Economic evaluation of the systems

Details of the three previous studies (Humphreys *et al*., 2008; 2009; Keogh *et al*., 2010) are presented in Table 6.1. There were one WC and three FN in 2001 replicated in 2002, one WC and one FN in 2003 replicated in 2004, 2005 and 2006 and one WC in 2008 replicated in 2009. Therefore, for the present study, there were ten FN and eight WC involving a range of stocking densities of spring-calving Holstein-Friesian dairy cows, inputs of fertiliser N and concentrate feed (Table 6.1). For the purposes of this paper, the production data were compared on the basis of a farm area of 50 ha, with dairy replacements reared on the farm and grass-silage produced on the farm to meet the winter forage requirements. Replacement rate was on average 23 per cent. Surplus calves were sold once they were approximately 3 weeks of age and culled cows were sold off the farm at the end of lactation, in December each year. Approximately 0.90 of the diet was home-grown forage, approximately 0.65 grazed pasture, 0.25 grass-silage and 0.10 concentrate similar to that recorded in the above experiments.

Table 6. 1. Characteristics of the systems of dairy production based on N fertilised grass (FN) and grass-white clover (WC) grassland at Solohead Research Farm between 2001 and 2009 (Humphreys *et al*., 2008; 2009; Keogh *et al*., 2010). Data are means of two and four years[1].

	WC	FN			WC	FN	WC
Year	2001 to 2002				2003 to 2006		2008 to 2009
Stocking density (cows ha^{-1})	1.75	2.10	2.50	2.50	2.15	2.15	2.12
Fertiliser N (kg ha^{-1})	80	180	248	353	90	225	100
Concentrate (kg cow^{-1})	535	535	535	535	525	525	575
Milk output (kg cow^{-1})	6550	6275	6242	6375	6521	6526	6273
Milk output (kg ha^{-1})	11463	13178	15605	15938	14346	14357	13299
Milk fat (%)	4.05	4.20	4.08	4.19	4.17	4.20	4.31
Milk protein (%)	3.50	3.59	3.52	3.57	3.54	3.60	3.58
Milk Lactose (%)	4.78	4.77	4.74	4.79	4.72	4.74	4.72

[1]All data collated by Mihailescu E.

For the economic analyses and interpretation of the physical data, secondary data resources such as the Central Statistics Office of Ireland (CSO, 2010), results of the Teagasc 2009 profit monitor for spring milk farms (Ramsbottom and Clark, 2010), and Teagasc Management Data for Farm Planning (Anon., 2008) were used. The production data were valued using variable costs based on 2008 prices (Anon., 2008): €32 LU^{-1} (veterinary fees), €83 LU^{-1} (artificial insemination), €225 ha^{-1} (harvesting silage), €100 ha^{-1} (slurry spreading), and €122 LU^{-1} (other variable costs). The costs of harvesting silage and slurry spreading were included in contractor charges (Table 6.2). The concentrate feed was valued at €0.22 kg^{-1} (CSO, 2010). For the cost with fertiliser N was the weighted average for 2009 being in between the high and low prices used in the sensitivity analysis described below. The price per kg of N was €0.828 kg^{-1} for urea and €0.982 kg^{-1} for CAN.

For the present evaluation, surpluses of silage were sold each year and the deficits were met by purchased silage. Surpluses and deficits were calculated as the difference between preserved and consumed silage per system each year. The price of purchased silage DM was assumed to be €0.15 kg^{-1} (Anon., 2008). The cost associated with maintaining the white clover content of pastures by over-sowing with white clover seed was incurred only by WC as described by Humphreys *et al*., (2008; 2009).

The fixed costs were taken from Ramsbottom and Clark (2010), which was a report of the profitability of 1,100 commercial dairy farms in Ireland, due to unavailability of representative fixed costs for the experimental systems. The following fixed costs were used on the basis of milk sold (€ $litre^{-1}$): machinery, 1.23; car/electricity/telephone, 1.22; depreciation, 1.94; leases, 0.84; other miscellaneous fixed costs, 2.59.

From the total milk production each year, approximately 300 litres cow^{-1} were fed to the calves; the remainder was sold at a price of €0.288 $litre^{-1}$, which is the same as the intermediate milk price used in the sensitivity analyses described below. At Solohead Research Farm, in 2008 and 2009, the average price received was €350 $head^{-1}$ for culled cows and €120 $head^{-1}$ for calves. The net margin per hectare was calculated using a spreadsheet model developed in Excel.

6.2.3. *Sensitivity analysis*

Sensitivity analysis (Kleijnen, 1997) was carried out by using a spreadsheet model in Excel to investigate the response of the net margin of the 18 systems to changing prices of milk and fertiliser N. Variation in fertiliser N and milk prices was obtained from Central Statistics Office of Ireland (CSO, 2010). Three fertiliser N prices were used (€ kg^{-1} N): low (0.684 for urea and 0.856 for CAN), intermediate (0.828 for urea and 0.982 for CAN) and high (0.945 for urea and 1.347 for CAN) (CSO, 2010). The low prices were from 2007, the high from 2008, and the intermediate from 2009 (Fig. 6.1a). Similarly, for milk there were a low (€0.233 $litre^{-1}$), intermediate (€0.288 $litre^{-1}$) and high (€0.338 $litre^{-1}$) price (CSO, 2010). The low price was from 2009, the high was the average of 2007 and 2008, which were almost identical, and the intermediate was an average of the low and high values, which was almost identical to the average price received for milk in Ireland over the last ten years (CSO, 2010; Fig. 6.1b). A total of nine scenarios were investigated including all combinations of the three fertiliser N prices and three milk prices described above.

The sensitivity of the systems was assessed through the changes in the values of net margins relative to changing prices. Thus, the systems showing changes in net margin across the above price scenarios were assessed as being sensitive to volatile prices, whereas those showing very little or no changes were categorized as less sensitive or insensitive.

6.2.4. *Statistical analysis*

Data were subjected to analyses of variance to compare differences in production factors, sales, costs, gross and net margins per ha between the systems with data from systems in individual years as replicates in the model. The relationships between milk prices and net margins across the range of fertiliser N prices were examined using linear regression.

6.3. Results

6.3.1. Comparison of net margin with intermediate prices for milk and fertiliser N

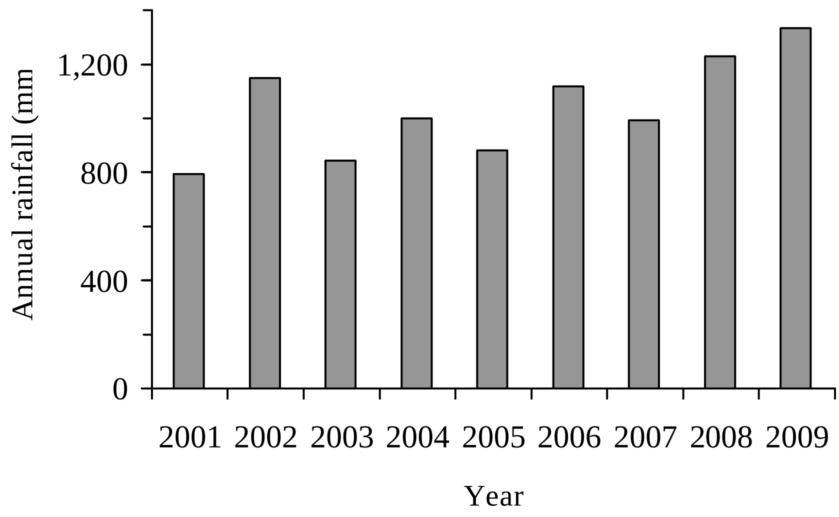
Stocking density, milk, cull cow, calf sales per ha from WC were approximately 0.90 ($P<0.05$) of FN (Table 6.2). Silage sales were of minor importance in both systems and total sales from WC were 0.91 ($P<0.05$) of FN. Fertiliser N use was substantially lower on WC than FN and the cost of fertiliser N per ha for WC was 0.34 ($P<0.05$) of FN. Concentrates and contractor changes accounted for approximately 0.45 of total variable costs on both systems. There was no difference ($P>0.05$) in the cost of concentrate between the systems whereas FN had higher ($P<0.05$) contractor charges. Total variable costs of WC were approximately 0.82 ($P<0.05$) of FN. There was no difference in gross margin between the systems. Fixed costs were higher ($P<0.05$) on FN. There was no difference in net margin between the systems for the scenario with intermediate milk and fertiliser N prices. There was considerable variation in the net margins of both systems from year to year, which was negatively correlated with annual rainfall (R^2 = -0.50; $P<0.01$; Fig. 6.2).

Table 6. 2. The economic performance of systems of dairy production based on N fertilised grass (FN) and grass-white clover (WC) grassland including sales, variable and fixed costs, gross and net margin per hectare[1].

System	FN	WC	*P* Value
Stocking density (LU ha^{-1})	2.28	2.04	<0.05
Fertiliser N (kg ha^{-1})	246	90	<0.001
Milk sales (€ ha^{-1})	3168	2875	<0.05
Total sales (€ ha^{-1})	3530	3205	<0.05
Fertiliser N (€ ha^{-1})	223	75	<0.001
Concentrate (€ ha^{-1})	312	275	NS
Contractor charges (€ ha^{-1})	299	253	<0.01
Total variable costs (€ ha^{-1})	1400	1146	<0.01
Gross margin (€ ha^{-1})	2131	2058	NS
Fixed costs (€ ha^{-1})	860	781	<0.05
Net margin (€ ha^{-1})	1271	1278	NS

[1] All data collated and analysed by Mihailescu E.

(a)



(b)

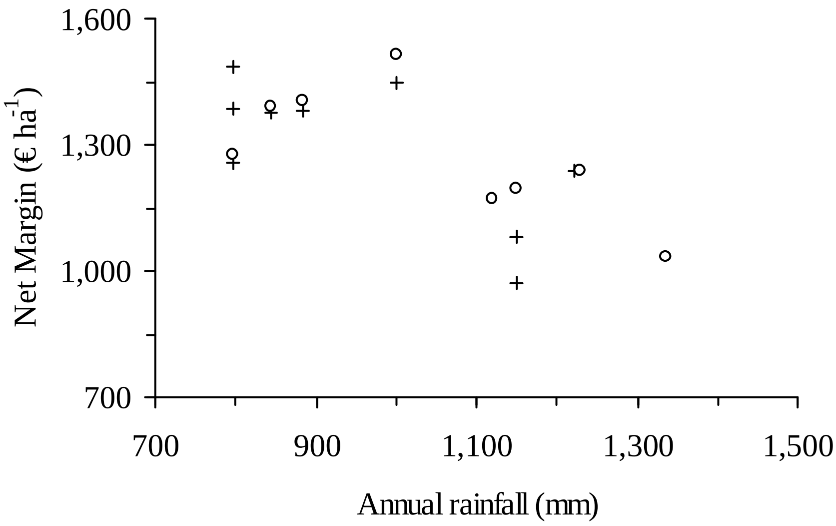


Fig. 6. 2. (a) Annual rainfall at Solohead Research Farm between 2001 and 2009, and (b) annual rainfall and net margins of systems of dairy production based on N fertilised grass (+) and grass-white clover (○) grassland with intermediate milk and fertiliser prices (Data analysed by Humphreys J.)

6.3.2. Comparative sensitivity of net margins

In relation to changing milk and fertiliser N prices the general trend was that the net margins of FN and WC were very sensitive to changing milk price and to a much lesser extent to changing fertiliser N price (Table 6.3). The net margin of WC was less sensitive to fertiliser N price than FN. In scenarios with high fertiliser N price combined with intermediate or low milk prices, or intermediate fertiliser N price combined with low milk price, WC was more ($P<0.05$) profitable than FN. In contrast where low fertiliser N price was combined with high milk price FN was more ($P<0.05$) profitable than WC. Based on the relationships between milk prices and net margins in this study (Table 6.3) the milk price at which the net margin of WC equalled that of FN across a range of fertiliser N prices was determined (Fig 6.3a). With higher fertiliser N prices a higher milk price is necessary for FN to be more profitable than WC.

Figure 6.3b shows the actual milk price (weighted average) for each year between 1990 and 2010 relative to the milk price at which the profitability of WC would have equalled FN based on fertiliser N prices during that period. In the fifteen years between 1990 and 2005, the milk price was high relative to fertiliser N price in each of these years to the extent that FN was clearly more profitable than WC. However, in the five years between 2006 and 2010 the situation was much less clear cut. In 2007 and 2010 fertiliser N and milk prices were such that FN was more profitable than WC. In 2009, a year that combined high fertiliser N with low milk prices (Fig. 6.1), WC was more profitable than FN. In 2006 and 2008 the actual milk price was close to the points where there was little difference in profitability between WC and FN. The milk prices were the profitability of WC equals FN was projected to 2020 based on the average increase in fertiliser N prices between 1997 and 2010 (Fig. 6.3b). This indicates that in future relatively high milk prices will be needed to sustain the profitability of FN relative to WC.

Table 6. 3. The impact of high (H), intermediate (M) and low (L) milk and fertiliser N prices on the net margins per ha of systems of dairy production based on N fertilised grass (FN) and grass-white clover (WC) grassland[1] and the relationships between milk prices and net margins per ha across the range of fertiliser N prices[2].

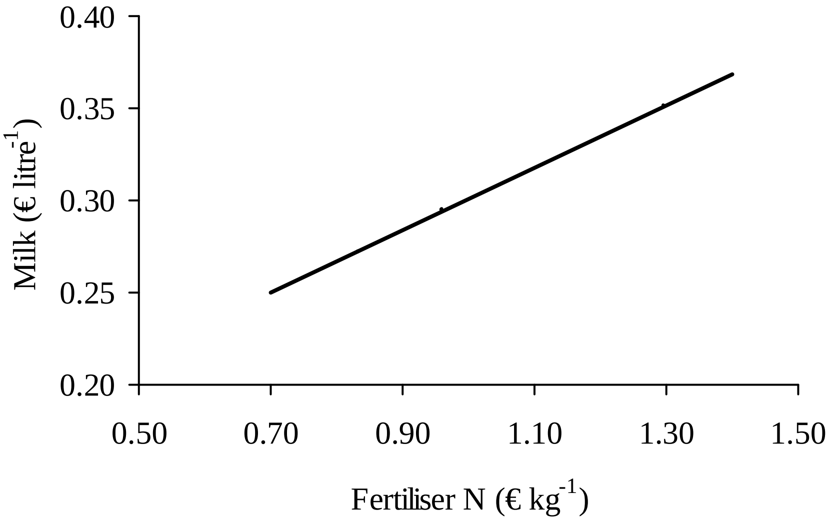
Scenario	Fertiliser N price	Milk price	FN	WC	P Value
S1	H	H	1,761	1,751	NS
S2	H	M	1,202	1,252	<0.05
S3	H	L	597	703	<0.01
S4	M	H	1,812	1,779	NS
S5	M	M	1,262	1,262	NS
S6	M	L	657	714	<0.05
S7	L	H	1,845	1,775	<0.05
S8	L	M	1,294	1,275	NS
S9	L	L	690	726	NS

Fertiliser N price	Intercept	Slope	s.e. slope	R^2	P value
			FN		
H	-1,966	11,001	649	0.911	<0.001
M	-1,906	11,001	668	0.906	<0.001
L	-1,873	11,001	677	0.904	<0.001
			WC		
H	-1,623	9,982	707	0.900	<0.001
M	-1,612	9,982	706	0.901	<0.001
L	-1,599	9,983	703	0.902	<0.001

[1] Analysis carried out by Mihailescu E.

[2] Analysis carried out by Humphreys J.

(a)



(b)

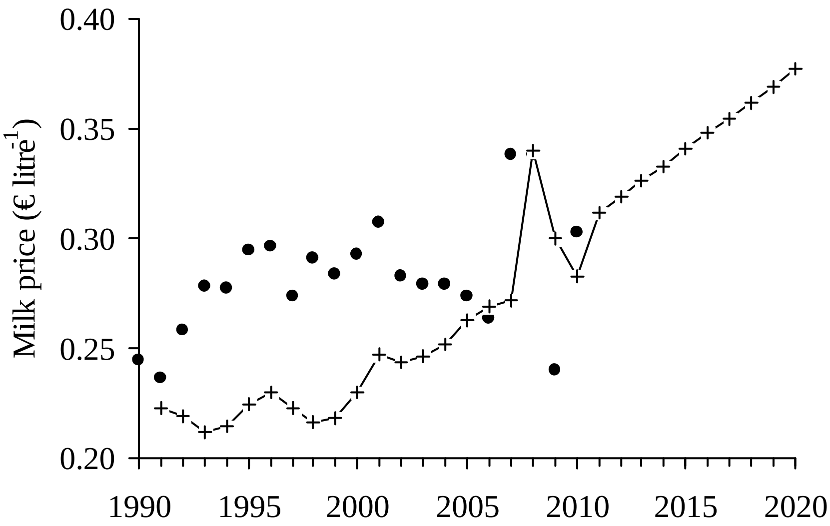


Fig. 6. 3. (a) The combination of fertiliser N and milk prices at which the profitability of dairy production based on grass-white clover (WC) equals that based on N fertilised grass (FN): Above the line FN was more profitable and vice versa, and (b) actual milk price (●) and the milk price (+) at which the profitability of WC would have equalled FN between 1990 and 2010 and projected to 2020 based on the increase in fertiliser N price between 1997 and 2010 ($R^2 = 0.77$; $P < 0.001$) (Humphreys J.)

6.4. Discussion

6.4.1. Inter-annual variation of profitability

There was considerable variation in the net margins of both systems, which can largely be attributed to differences in rainfall between years (Fig. 6.2). Under high summer rainfall there was a lower herbage response to applied fertiliser N, particularly at the higher N fertilisation rates and more difficult grazing conditions leading to lower annual milk sales (Humphreys *et al.*, 2008). Furthermore, in wet years, there was higher supplementation with concentrates, hence higher costs, and longer indoor housing with associated higher costs similar to that described in other studies (Sayers and Mayne, 2001; Dillon *et al.*, 2002; Kennedy *et al.*, 2005).

One of the weaknesses of this study was that there were more FN than WC in the earlier years (2001 and 2002) when rainfall was relatively lower than the later years (Table 6.1), which would have favoured the relative profitability of FN in this study. Moreover, in some instances there was a poor match between the herbage production and stocking densities (Humphreys *et al.*, 2009) leading to sizeable silage sales especially in 2003 and 2005 from FN and in 2001, 2002 and 2003 from WC. This herbage could have been more profitably converted into milk. Nevertheless, overall sales of silage were fairly evenly divided between WC and FN and would have had a relatively small impact on the extent of the differences in net margins of the systems.

6.4.2. Comparison of profitability of the systems

The lower total sales reflected the generally lower stocking densities on WC. However, variable costs were also lower on WC. Variable costs were lower due to lower fertiliser N costs associated with the replacement of fertiliser N by biologically fixed N in WC and also due to the smaller scale of production on WC leading, for example, to lower contractor charges. Overall variable costs on WC were 0.82 of FN and 0.58 of this difference was due to differences in the fertiliser N input; the remainder was mostly due to differences in scale. Consequently, there tended to be little difference in the gross margins between the two systems with intermediate milk and fertiliser N prices. The fixed costs tended to be marginally higher on FN, which was attributable to activities associated with higher stocking densities and higher milk output such as electricity use, labour and repayments on capital investments, which in general, tend to increase with increasing scale of the enterprise (Ramsbottom and Clarke, 2010).

In the sensitivity analysis there was a very clear trend for WC to be more profitable with higher fertiliser N prices. This can be offset by higher milk prices (Fig.6.3a) although, as pointed out in the introduction the price of fertiliser N has been increasing at a higher rate than milk price and, hence, the increasing ratio between fertiliser N and milk price in Fig. 6.1c. As can be seen in Fig. 6.3b, FN has been consistently more profitable than WC between 1990 and 2005, which is in general agreement with many previous studies (Doyle *et al*., 1984; Ryan, 1988; Penno *et al*., 1996; Schils *et al*., 2000; Leach *et al*., 2000). However, with the steady increase in fertiliser N prices relative to milk price, the difference between FN and WC was less clear cut between 2006 and 2010. Projecting into the future assuming similar trends in fertiliser N and milk prices to that in last decade, this analysis indicates that WC is likely to become an increasingly more profitable alternative to FN for pasture based dairy production.

6.4.3. Wider Implications

In the present study WC had a stocking density and milk output of approximately 0.90 of FN and had similar profitability to FN. As pointed out above, the lower N fluxes associated with the lower productivity of WC are generally associated with lower losses of nitrate to water and of ammonia and nitrous oxide to the atmosphere than FN (Ledgard *et al*., 2009). Hence, the wider adoption of WC on farms offers potential to meet the twin goals of a sustainable income for dairy farmers in the context of rapidly rising cost of fertiliser N while better meeting environmental targets for lowering N losses from pasture based dairy farming.

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7. General discussion

This research was conducted on a sample of 21 farms which was not representative of the Irish dairy industry as a whole. Seventeen of the 21 investigated farmers participated in REPS (Rural Environment Protection Scheme) (DAFM, 2013a), therefore closely following the guidelines of GAP Regulations introduced in Ireland in 2006 (European Communities, 2010). Hence, the results cannot be extrapolated to all Irish dairy farmers. Also, the collected data were not checked through direct measurements of N and P inputs in mineral fertilisers, feeds (concentrates and fodders) and livestock and of N and P outputs in milk and livestock. This might be the reason for the generally high level of uncertainty in the results, especially in Chapters 4 and 5.

In Chapter 3 it was found that optimised use of mineral N and concentrate N inputs can most efficiently contribute to maintenance of herbage and milk yields, on one hand, and decreases in N balances and increases in NUE, on the other hand. This optimisation can be achieved through matches between the animals' N requirements with concentrate N inputs (Steinshamn *et al.*, 2004; Buckley and Carney, 2013), as well as between the grass plants' N requirements with mineral N inputs (Hennessy *et al.*, 2008). Other Irish (Treacy *et al.*, 2008) or European (Groot *et al.*, 2006; Nevens *et al.*, 2006; Raison *et al.*, 2006; Roberts *et al.*, 2007; Cherry *et al.*, 2012; Oenema *et al.*, 2012) studies emphasized the reduction of mineral N and concentrate N inputs associated with decreases in farm-gate N balances and NUE. Nielsen and Kristensen (2005) and Gourley *et al.* (2012) mentioned improved N recycling between soil-plant-animal-manure compartments as a contributor to increases in NUE on dairy farms.

Differently than other studies (Ryan *et al.*, 2011; Buckley *et al.*, 2013), in the current study SR had a weak impact on farm-gate N balances. This implies that SR can be increased without considerably increasing N surplus. This has important implications in the context of achieving a 50 % increase in dairy production as is envisaged in the Food Harvest 2020 report for Ireland (DAFM, 2013b).

In Chapter 4 it was found that 18 farmers applied lower amounts of mineral P fertilisers than P requirements of grass plants during a growing season (minimum 14 kg P ha^{-1} year^{-1}; Ryan and Finn, 1976). While this contributes to decreases in the proportion of soils with high STP (>8 mg litre^{-1}) and in the risk of P loss to water, continuous monitoring of STP levels is needed to maintain soil fertility in the long term (Lalor *et al.*, 2010; Bourke *et al.*, 2008; Wall *et al.*, 2012).

Similar to N, efficient use of P inputs from mineral fertilisers and feeds was found to contribute to increases in PUE. In Ireland, P inputs from mineral fertilisers can be efficiently used when contributing to STP levels between 5.1 and 8.0 mg litre^{-1}, which are considered optimal for herbage yields (Schulte and Herlihy, 2007). However, in the current study, STP levels were not estimated each year due to high costs for the farmers. Therefore, the correspondence between fertiliser P applications and STP levels was not monitored for the whole period of study.

In the current study, the use of feed P inpus can be optimised by increasing the number of days grazing and accounting for harvested grass in animal diet. An extended grazing season (from February to November; Humphreys *et al.,* 2009) may be one important reason for much lower feed P inputs on the farms in the current study (7.62 kg P ha^{-1}) compared to Dutch (24.00 kg P ha^{-1}; Aarts, 2003), Danish (22.00 kg P ha^{-1}; Nielsen and Kristensen, 2005) and daiy farms on the Atlantic seaboard (between 11.88 kg P ha^{-1} and 66.00 kg P ha^{-1}; Raison *et al*., 2006). Compared to Ireland, the length of the grazing season varied from zero in Galicia and North Portugal (Raison *et al*., 2006) to six months in The Netherlands (Groot *et al.,* 2006), and nine months in Brittany (Raison *et al*., 2006).

It is important to note here that due to the role of both N and P inputs from mineral fertilisers and feeds in supporting herbage and milk production and driving N and P surpluses on the dairy farms in the current study, integrated N and P improved management needs to be undertaken, in order to contribute to increased economic (farms' ability to generate sufficient funds to sustain their production potential in the long run; European Comission, 2001) and environmental (indicated by N and P surplus; Jarvis and Aarts, 2000; Schröder *et al.,* 2003; Carpani *et al.,* 2008) sustainability of these farms.

Comparison to similar studies completed before the introduction of the Nitrate Directive in Ireland (Mounsey *et al.,* 1998; Treacy *et al*., 2008; Treacy, 2008) indicated considerable decreases in N and P surpluses and increases in NUE and PUE, mostly due to decreased mineral N and P inputs and improvements in N and P management, with a shift towards spring application of organic fertilisers. These results indicate a positive impact of the GAP regulations on dairy farm-gate N and P surplus and NUE and PUE and an improvement in N and P recycling on these farms, as envisaged in the Food harvest 2020 report (DAFM, 2013b). However, the farms involved in these studies were intensive dairy farms, therefore not being fully representative of all Irish dairy farms, which requires caution in interpreting these results.

In Chapter 5, it was found that an increase in net profit could be achieved by increasing milk receipts and decreasing mineral fertiliser expenditure. As discussed in Chapters 3 and 4, it is possible to decrease the amounts of mineral N and P inputs and associated expenditures on dairy farms, through optimised use of these inputs. Consequently, it can be expected that controlling these expenditures while maintaining milk yields, and associated receipts, would be more effective for maintaining net profit of the dairy farms in the current study. However, among EU dairy specialist producers (Belgium, Denmark, France, Germany, Italy, The Netherlands, England), Ireland has expenditure disadvantage in terms of mineral fertilisers (6.3 % of total output, compared to 2.3 %, on average, for the EU producers) (Donellan *et al*., 2011). This indicates high reliance on mineral fertilisers, and therefore high associated expenditures, of grassland-based Irish dairy farms. In this context, an increase in spring application of organic fertilisers, associated with a decrease in annual requirement for mineral fertilisers, as found in Chapters 3 and 4, can be considered as an effective way of controlling mineral fertiliser expenditures.

Increased net profit was also associated with SRs above 2 LU ha^{-1} for nine farms under derogation conditions (European Communities, 2010). Similarly, Patton *et al.* (2012) reported higher net profit associated with SR above the 2.0 LU ha^{-1} limit required by GAP Regulations (European Communities, 2010). In contrast, McCarthy *et al.* (2007) found that an increase in SR above this limit was not associated with an increase in net profit. These differences partially support the argument of Brennan and Patton (2010) that in grazed grass-based dairy production systems, increases in SR can determine increases in farm profitability as long as there is a good match between SR and the grass growing potential of the farm, to allow increased grass utilisation, with no major additional imports of either concentrate or mineral fertilisers and associated increased expenditures.

The most significant relationship in Chapter 5 indicated that mean feed expenditure increased with mean SR and feed inputs and decreased with total utilised agricultural area (TUAA). This highlights the importance of matching SR (and animal feed requirements) with the feed imports on grass-based dairy farms when there is limited availability of grassland area (McCarthy *et al.*, 2007), as an effective strategy to decrease farm-gate N and P balances, and to control feed expenditures, with potential positive impact on net profit. This finding is important in the context of generally low availability and high cost of agricultural land in Ireland (Donnellan *et al.,* 2011; Patton *et al*., 2012; Kelly *et al.,* 2013).

The results of the sensitivity analysis indicated that high input dairy farms are more sensitive than low input dairy farms to increases in mineral fertiliser price and decreases in milk price. This has important implications in the context of increasing mineral fertiliser prices (CSO, 2013) and milk quota abolition in 2015, which is expected to determine increased milk price volatility (Kelly *et al.,* 2012; Geary *et al.,* 2012).

Under these circumstances, the introduction of white clover in grazed grass-based dairy systems can contribute to lower mineral N expenditures (Humphreys *et al.,* 2012), due to the replacement of mineral N fertiliser by biologically fixed N_2 via white clover (Humphreys *et al.,* 2008). In Chapter 6 it was found reduced sensitivity of white clover/grass-based dairy systems (WC) compared to mineral N fertilised grass-based dairy systems (FN) in the scenarios with low milk price and high mineral fertiliser price. Similarly, Moreau *et al.* (2012) found very low sensitivity to variation in milk and mineral fertiliser N prices for clover/grass-based French dairy production systems relying on N inputs from biological N fixation via white clover.

Another opportunity for reducing mineral N input and associated expenditure on dairy farms in Ireland is to account for the nitrogen fertiliser replacement value (NFRV) of slurry applied to grassland (Lalor, 2008), with potential positive impacts on the farm-gate N balance and net profit, as discussed in Chapter 5. The potential fertiliser N cost savings associated with spring slurry application may have represented one important reason for increased proportion of slurry applied in spring, as discussed in Chapters 3 and 4.

It can be concluded that grass-based Irish dairy farms offer real opportunities to optimise N and P inputs in the form of organic and mineral fertilisers and feeds, with positive impacts on farm-gate N and P balances and potential N and P losses, as well as on farm profitability. However, caution needs to be taken to maintenance of herbage and milk production to optimal economic levels.

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