IMPROVING NUTRIENT MANAGEMENT WITHIN INTENSIVE GRASSLAND-BASED SYSTEMS OF MILK PRODUCTION

Final Report for DARD (Project 0821) and for AgriSearch (Project D-29-06)

February 2015

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This research programme was funded jointly by DARD and by AgriSearch

STRUCTURE OF REPORT

This report begins with an 'Executive summary' which highlights key findings of the research undertaken. The body of the report is then presented in five distinct sections. Section 1 describes the dairy cow production study undertaken and examines the performance of the cows on each of the four experimental systems. Section 2 provides a detailed description of the establishment and management of the experimental 'farmlet' site used to measure nutrients losses. In Sections 3 and 4, details of nutrient losses measured within the farmlet site are presented. Finally, Section 5 provides a detailed description of four plot experiments which were undertaken to examine strategies by which to reduce phosphorus losses from applied slurry.

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EXECUTIVE SUMMARY

On 1 January 2007 the whole of Northern Ireland was designated a Nitrates Vulnerable Zone (NVZ) under the EU Nitrates Directive, thus establishing a stocking rate limit of 170 kg manure nitrogen per hectare (1.87 cows per hectare).

Northern Ireland applied to the EU Commission for a 'derogation' from the Nitrates Directive to permit some farmers to operate at higher stocking rates (up to 250 kg organic nitrogen per hectare: 2.74 cows per hectare). This application was successful.

However, the EU Commission indicated that research to improve nutrient utilisation within intensive grassland based milk production systems was required. This was stated in Article 8(6) of the EU Derogation Document: '*A study shall be conducted in order to collect, by the end of the derogation period, detailed scientific information on intensive grassland systems in order to improve nutrient management. This study will focus on nutrient losses, including nitrates leaching, denitrification losses and phosphate losses, under intensive dairy production systems in representative areas'*

To address this requirement, a research project was established at AFBI Hillsborough, the only dairy cow research facility within Northern Ireland, and one which is situated in an area which is broadly representative of dairying in large parts of Northern Ireland. The outcomes of this research project are presented in this report, under five separate sections.

Section 1: The performance of dairy cows within four contrasting grassland-based systems of milk production over three successive lactations

This study involved a comparison of four very different grassland-based milk production systems. Twenty cows were managed on each of four systems (Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx) for three successive years.

The **'Confinement'** system involved winter calving Holstein cows which were housed throughout the entire lactation and offered a mixed ration comprising grass silage, maize silage and concentrates. The concentrate : forage ratio in this mix was 55 : 45 during the first 180 days of the lactation, being reduced to 45 : 55 during the remainder of the lactation.

The **'WinterCalf'** system also involved winter calving Holstein cows. These were housed during the winter and offered the same diet as cows on the Confinement system. Cows on this system started grazing in early April and were offered approximately 4.0 kg concentrate/cow/day throughout the summer grazing season.

The remaining two systems were low input spring calving systems. One of these systems involved spring calving Holstein cows (**SpringCalf(H)**) and the other spring calving Jersey x Holstein crossbred cows (**SpringCalf(Jx)).** Cows on these two systems were offered grass silage plus 6.0 kg concentrate/cow/day from calving until turnout. During the grazing period these cows were supplemented with 1.0 kg concentrate/cow/day.

Each system was designed to operate at a high stocking rate (ie one which would require a derogation from the Nitrates Directive). In addition, each system was managed to minimise nitrogen and phosphorus loss to the environment. Cows on all systems were milked twice daily.

Total concentrate intakes over the full lactation were 3.5 t with the Confinement system, 2.5 t with the WinterCalf system and 0.85 t with the two SpringCalf systems. Silage intakes were greatest with the Confinement system as these cows were housed all year. Grass intakes were highest with the SpringCalf systems as cows on these systems had a long grazing period and were offered only a small amount of concentrates while grazing. Total intakes were greatest with the Confinement system and lowest with the SpringCalf systems. Holstein and Jersey crossbred cows in the spring calving system had similar intakes, thus demonstrating the high intake capacity of the smaller crossbred cows.

Cows on the Confinement system lost less body condition than cows on any other system and started to gain body condition by approximately week 16 of lactation. These cows completed the lactation with a higher body condition score (2.7) and a higher live-weight than cows on any other system.

Cows on the WinterCalf system had a similar liveweight change as those on the Confinement system until turnout. However, these cows lost live-weight post turnout and finished the lactation with a lower live-weight than those on the Confinement system. Unlike cows on the Confinement system, these cows did not gain body condition after week 16 of lactation.

Holstein cows on the SpringCalf system were approximately 60 kg heavier than the Jersey crossbred cows. Nevertheless, the Holstein and Jersey crossbred cows on this system lost similar amounts of live-weight in early lactation and gained similar amounts of live-weight in late lactation.

Although the management of cows on each of the systems differed in many respects, good quality silage and high quality pasture were offered throughout the experiment. Thus it is likely that concentrate feed level was the main driver of cow performance across the systems examined.

As expected, milk output was highest with the Total Confinement system. However, milk output with this system was less than 1000 kg higher than for the WinterCalf system, despite an additional tonne of concentrates being offered. This represents a milk yield response of 0.86 kg milk per kg concentrate.

Holstein cows on all systems produced milk with a high fat and protein content, a reflection of the sire selection programmes that have been in place at Hillsborough during the last decade. The higher milk fat content with the Confinement system is due to the greater proportion of silage in the diet with this system, with forage fibre a key driver of milk fat production.

The Holstein cows on the SpringCalf system produced 400 litres more milk than the Jersey crossbred cows. However, the Jersey crossbred cows produced milk with a higher fat and protein content than the Holstein cows. The overall effect was that fat + protein yield did not differ between these two cow genotypes. These results again demonstrate that the smaller crossbred cows must have had similar dry matter intakes as the larger Holstein cow in order to produce the same amount of milk solids.

‘Milk from forage’ values were lowest with the Confinement system and highest with the SpringCalf systems.

There was a general trend for crossbred cows within the SpringCalf system to have improved fertility compared to Holstein cows within this system, or indeed within any system. This improved fertility is likely due to hybrid vigour.

Within the systems involving Holstein cows, there was a general trend for cows within the Confinement system to have poorer fertility than those on the SpringCalf system. While overall conception rates at the end of the breeding season did not differ between systems, cows on the Confinement and WinterCalf systems had a 6-7 month breeding season, compared to a 14-week breeding season for cows on the Spring calving system.

Somatic cell counts tended to be higher in the systems which involved longer periods of housing (Confinement and WinterCalf). In addition, there were clear trends for Holstein cows on these two systems to have increased incidence of mastitis than Holstein cow on the SpringCalf systems. Clearly, cleanliness of cubicles and bedding versus cleanliness of pasture and grazing conditions, can all impact on the cell counts and mastitis risk for cows within these different systems.

The trend towards a lower incidence of mastitis with the crossbred cows compared to the Holstein cows is likely due to hybrid vigour. However, somatic cell count was not reduced by crossbreeding. Hybrid vigour has been shown to reduce mastitis incidence, while having little effect on somatic cell counts.

Hoof health problems are known to increase with increased duration of housing. In the current study there were trends for cows within the Confinement and WinterCalf systems to have higher levels of lameness compared to those within the SpringCalf systems, and this is likely to reflect the increased length of the housing periods and higher concentrate feed levels with these cows.

While the exposure of the hoof to slurry within a cubicle situation is likely to be a contributing factor to the incidence of hoof health problems, the hardness of the standing surface, increased lying times, and exercise are all likely to be additional contributing factors. The results of the study also suggest improved hoof health with crossbred cows compared to Holstein cows on the Spring calving system. In general crossbreeding improves hoof health, and Jersey crossbred cows have been shown to have harder hooves.

In order to calculate a 'whole farm' stocking rate, each of the four systems were 'scaled-up' to a 100-cow dairy herd. In addition, as the study did not involve young-stock, stocking rates and management practices presented within the Northern Ireland Farm Business Survey (2013) were used for young-stock.

Whole system stocking rates (including young-stock) were 2.5, 2.3, 2.4 and 2.4 for Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx), respectively. Thus all four systems would require a derogation under the NI Nitrates Directive action programme.

These whole system stocking rates were used when calculating whole farm phosphorus balances, greenhouse gas emissions and economic performance.

An objective of this experiment was to minimise P surpluses within each of the four systems. This was achieved by offering 'low' phosphorus concentrates and by not applying inorganic P fertiliser, the latter justified by the high P status of the soils within the study area.

Calculated P balances were 5.4, 0.6, -5.7 and -5.1 kg P per ha for Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx), respectively. For the Confinement system this P balance is likely to be sustainable in the longer term, provided slurry P is distributed across the entire grassland area of the farm. However, the other three systems are not sustainable long term with regards phosphorus. Indeed, the Spring-calving systems had a negative P balance which means that more phosphorus was removed from the farm in milk, in calves and in cull cows, than was imported onto the farm in concentrate feed or fertiliser. Thus additional P must be added to these systems if they are to sustainable long term.

This was examined by modelling the impact of applying 20 kg P_2O_5/ha (8.8 kg P) to all grassland areas within each system. In this scenario, the P balance with each system increased to 12.8, 8.7, 3.1 and 3.7 kg/ha, for Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx), systems, respectively. Under this scenario the Confinement system now has a P balance which is greater than the value of 10.0 kg/ha currently permitted for 'derogated farms' in Northern Ireland. This clearly demonstrates that with these high concentrate input systems there is relatively little scope to apply additional P as inorganic fertiliser, thus highlighting the need to distribute slurry nutrients evenly across the entire land area. The relatively small surpluses with the Spring-calving systems suggest that these systems may still not be sustainable even at a application rate of 20 kg P_2O_5/ha.

The Carbon footprint of each of the four systems was examined using the AFBI Dairy Systems Greenhouse Calculator. The carbon footprint of each of the Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx), systems were calculated to be 0.98, 1.00, 1.07 and 1.02 kg CO2-e per litre of milk, respectively. Thus system had relatively little impact on the carbon-footprint of the different systems, something that has been observed previously when systems are well managed.

The financial performance of the four systems was examined at a milk price of 32 pence per litre, adjusted for compositional bonuses. Differences between breeds in replacement rates, stillbirth rates, and calves and cull cows sold have been included within the calculations. Feed costs for the 'milking herd' were based on actual feed inputs measured within the study, with costs for grass silage, maize silage and grazed grass assumed as £105, £115, £60/t DM, respectively (CAFRE Forage costs, updated 2013). The cost of all concentrates was assumed as £275/t fresh.

Annual gross margins were calculated per cow, per ha and per litre of milk produced for each system. Gross margin per cow was highest with the WinterCalf system and lowest with the spring calving systems, while gross margin per ha was highest with the

Confinement system and lowest with the Spring calving systems. Gross margin per litre of milk produced was lowest with the Confinement system and highest with the Spring calving systems.

Fixed costs were obtained from CAFRE Benchmarking (2012 and 2013) for herds identified as ‘Fully Confined’ and ‘Conventional’ (farms producing at least 7500 litres/cow/year) and as ‘Spring Calving’. Across all farms within each of these three categories, mean fixed costs per litre of milk produced were 7.5, 6.6 and 8.3 pence per litre (£619, £536, £481 per cow), for Fully Confined, Conventional and Spring calving farms, respectively. Net margin values for each of the four experimental systems were then calculated by deducting fixed costs obtained from benchmarking from the gross margin values, described above.

The overall outcome of the economic analysis (milk at 32 pence per litre, concentrates at £275/t) was that net margin per cow and per ha was maximised with the WinterCalf system, while net margin per litre was maximised with the spring calving systems.

It is now recognised that volatility in milk prices and input costs are likely to remain a permanent feature of dairy farming for the foreseeable future, and as such, optimum systems are those that are likely to be robust and resilient over a wide range of milk price/concentrate cost scenarios. Thus this analysis was repeated at a milk price of 27 and 22 pence per litre (concentrate cost, £250/t). In general, the WinterCalf system was most profitable at milk prices of 27 ppl and greater, while at a milk price of 22 pence per litre the SpringCalving systems tended to be more profitable. This finding supports previous modelling work undertaken by AFBI which indicated that moderate input-moderate output autumn calving systems (approximately 8000 litres/cow/year), and high output Spring calving systems (approximately 7000 litres/cow/year) are the most robust systems for Northern Ireland.

The Spring calving systems involving Jersey crossbred cows were more profitable that those involving Holstein cows. This supports the findings of previous comparisons of these two breeds at AFBI Hillsborough.

In addition, when concentrate prices increased, net margins fell dramatically with systems involving higher concentrate inputs. Similarly, the increase in net margin associated with a reduction in concentrate cost was greatest with systems involving higher concentrate inputs.

Section 2: The establishment and management of a replicated 'farmlet' site to measure phosphorus, nitrous oxide and nitrate losses from three contrasting grassland based milk production systems over two successive years

Facilities were not available at AFBI to allow completely 'closed' systems to operate, that is, systems whereby nutrients produced by livestock on each system would be recycled within the land areas associated with that system. To overcome this problem, a replicated farmlet site (81 m x 93 m) was established to allow nutrient losses to be measured. Nutrient losses were measured from three out of the four systems examined: Confinement, WinterCalf and SpringCalf(Jx).

The nutrient loss site was nominally divided into four replicated blocks (A, B, C and D). Each block contained two mini-grazing paddocks, one each for cows on systems WinterCalf and SpringCalf(Jx), a block for growing maize silage and five silage plots.

The site was managed as described below during the three years of the study, with nutrient loss measurements confined to Years 2 and 3 of the project.

The mini grazing paddocks measured 0.043 ha (12.2 m x 35 m) for cows on system WinterCalf and 0.050 ha (14.3 m x 35 m) for cows on system SpringCalf(Jx). Mini-paddocks were grazed at the same frequency, and fertiliser applied at the same rates as for the main grazing plots. At the start of the second grazing season (Year 2), three static chambers were placed at randomly selected locations within each of the eight mini-grazing paddocks, with these chambers remaining in place for 24 months. These static chambers were used to measure nitrous oxide emissions.

Each of the four 'maize plots' had dimensions 13 m x 10 m, and maize was sown and managed, as per a normal maize crop. During Year 2, one static chamber was placed within each of the maize plots, and remained there for 24 months. Maize was harvested in late autumn.

Silage plots (each measuring 8.0 m x 2.0 m) were established within each of Blocks A–D, with these simulating 'silage areas' within systems Confinement, WinterCalf and SpringCalf(Jx). A fourth plot was treated as a zero N plot. Plots were treated with slurry (simulated trailing shoe: pre-first grass harvest, post-first grass harvest and post-second grass harvest, each year) and with inorganic fertiliser within 5-9 days (mean, 7.7 days) of slurry being applied. Herbage from all plots was harvested on four occasions each year. Nitrous oxide emissions were measured using static chambers within each plot during Years 2 and 3.

Slurry applied to the maize plots and silage plots was collected from cows on the appropriate systems, and application rates were based on the calculated total quantities of slurry produced during periods of confinement.

Although the experiment was conducted over a three-year period, this was not deemed sufficiently long to provide a robust examination of changes in soil properties over time. Nevertheless, a number of 'trends' were observed. Firstly, there was a 'trend' for soil P levels within the grass silage plots to fall over time, although soil N levels and soil total carbon levels remained unchanged. No such trends in soil P levels were observed within the grazing plots. Within the maize plots there was a clear trend for both soil P and soil K levels to increase over time. While soil N levels and total carbon levels did not appear to change during the three-year period within the maize plots, these tended to be lower than those recorded within the grass silage and grazing plots.

Maize yields during the study were poor, partly a consequence of the site (North facing) where the plots were established (chosen in order to have slopes suitable for runoff measurements), and due also to establishment problems due to 'pest attack'. The poor yields were reflected in the low recoveries of applied N and P.

Total yields of herbage harvested within the Confinement, WinterCalf and SpringCalf(Jx) grass silage plots were 14.1, 14.6 and 13.4 t DM/ha. As expected, the herbage yields with the Zero N plots were approximately half of those recorded with the treatments which received organic and inorganic N. Dry matter yields varied little over the three-year period that the experimental site was in operation. Across the Confinement, WinterCalf and SpringCalf(Jx) silage plots, proportionally 0.73 of N applied and 1.52 of P applied was recovered in crop harvested.

Section 3: Evaluating the Impact of Grazing on Phosphorus Loss from Grassland Soils

Two studies were conducted to examine the impact of grazing on phosphorus losses from grazing systems. In the first of these (Experiment 1), the cumulative impact of grazing on nutrient export in overland flow was examined at the end of the grazing season within the farmlet site. A second study examined the impact of grazing intensity during the spring period on soil associated nutrient losses in overland flow (Experiment 2).

Experiment 1 was undertaken within the WinterCalf and SpringCalf(Jx) grazing treatments. In addition, 12 m x 1.5 m exclusion plots were established in the centre of each grazing plot. The exclusion plots provided an untrampled (UT) treatment for comparative purposes.

Overland flow was simulated on 0.5 m^2 subplots within each treatment plot by hydrologically isolating each plot from overland and shallow sub-surface flow using stainless steel surrounds placed 0.05 m into the soil. Rainfall simulations were carried out on the WinterCalf, SpringCalf(Jx) and UT treatment plots over a two-day period in February 2010 and 2011 prior to grazing commencing at the site. In addition, simulations were also undertaken in late October 2010 and 2011, after the final grazing of the experimental plots had taken place. The aim of conducting rainfall simulations pre- and post-grazing was to determine the accumulative impact of treatments over a complete grazing season.

Soil from both of the grazed treatments (WinterCalf, SpringCalf(Jx)) had a significantly greater bulk density ($P<0.001$) and a lower total pore space ($P<0.001$) than soil from the UT treatment.

Grazing treatment had no impact on the concentrations of nutrients or sediment recorded in overland flow in the WinterCalf, SpringCalf(Jx) and UT treatment.

In contrast, grazing had a significant impact on soil structure and the generation of overland flow. The change in soil structure in both the WinterCalf and SpringCalf(Jx) treatments resulted in an increase in the volume of overland flow generated during rainfall simulation events. In the case of the SpringCalf treatment, this resulted in significant differences in the NH_4, NO_2, total soluble phosphorus, and total phosphorus loads exported when compared to the UT treatment.

However, the lack of a significant difference in nutrient and sediment loads or concentrations between the SpringCalf(Jx) and WinterCalf treatments suggest that if best practice is adhered to, adopting a system with a much greater reliance on grazed grass (longer grazing season, lower concentrate feed levels), does not significantly increase the risk posed to water quality at this site.

Experiment 2 was undertaken on a separate site located just outside of the grazing area used within this overall project.

Sixteen plots, each measuring 3.0 x 7.0 m, were established in a four block randomised block design. The boundary of each plot was marked by triple strand electrified fencing.

Four treatments were examined in the experiment, with each treatment replicated four times. Each treatment comprised of two short term ‘grazing events’, which took place on 23 February 2010 (G1) and on 6 April 2010 (G2). Treatments during G1 were: ungrazed (UG-), lightly grazed (LG-) or heavily grazed (HG-), while the fourth treatment also remained ungrazed (UG-). During G2 the first three treatments were grazed to a common grazing intensity (-G), while the fourth treatment again remained ungrazed (UG-UG).

Grazing was implemented as follows: During G1 and G2, ten lactating Holstein Friesian dairy cows (average live weight 650 kg) were given access to the experimental site at approximately 10.00 h, with cows not having had access to food during the previous two-hour period. Cows were fitted with excreta collection bags using a harness system, thus preventing contamination of the site. During G1 cows had access to treatment plots LG- and HG- until a residual sward height of approximately 5.0 cm had been achieved, but before sward/soil surface damage became apparent (approximately 90 minutes of grazing). Cows were returned to the experimental site at approximately 13.00 h on the same day, but were given access to the treatment HG- plots only (approximately 90 minutes of grazing). During the second grazing event (G2; 6 April) cows had access to the UG-G, LG-G, and HG-G plots for a single period of 120 minutes.

Overland flow was simulated on 0.5 m^2 subplots within each treatment plot by hydrologically isolating each plot from overland and shallow sub-surface flow using stainless steel surrounds placed 0.05 m into the soil, and using a rainfall simulation technique. Measurements were conducted on all treatments at two days (RD2) and sixteen (RD16) days post-grazing using rainfall simulators.

Grazing had no effect on the dissolved P factions recorded after both G1 and G2.

Increasing grazing intensity was associated with an increase in particulate P concentrations in overland flow at RD2 and RD16 following G1, and this was likely due to the combined effects of soil disturbance and vegetation removal.

Following G2, PP concentrations in overland flow at RD2 were significantly higher with the HG-G treatment, compared to the ungrazed treatments, while PP concentrations with the LG-G treatment did not differ from the ungrazed treatments.

Section 4: Nitrous oxide emissions and nitrogen loss by leaching from three contrasting grassland based milk production systems over two successive years

This part of the study was designed to quantify losses of N as nitrous oxide and via nitrate leaching over a period of two years, from 15 February 2010 to 20 February 2012, from three of the dairy production systems (Confinement, WinterCalf and SpringCalf(Jx)).

A static chamber method was used to measure N_2O emissions, which were verified using the DeNitrification-DeComposition (DNDC) computer simulation model. Residual N in the soil profile to 90 cm was measured in the autumn of 2010 and the DNDC model was used to estimate N leaching losses. The latter approach was adopted as previous research at AFBI Hillsborough has demonstrated that direct measurement of leaching losses using techniques such as ceramic cups or 'dip wells' can be extremely problematic in Hillsborough soils.

Daily fluxes of N_2O were measured within the silage, grazing and maize components of each of the three systems over a period of two years. Briefly, there was 1 chamber in each of the four silage treatments, which were replicated four times (16 chambers), three chambers in each of two grazing plots replicated four times (24 chambers) and one chamber in each maize plot replicated four times (four chambers). The total number of chambers sampled on each occasion was 44 and the N_2O fluxes in the chambers within each replicate were averaged to obtain a mean value for the treatment.

Residual mineral N in the soil profile to 90 cm was measured during November 2010 from each plot. Soil cores were taken from each plot and divided into 0-10, 10-20, 20-30, 30-60 and 60-90 cm depths and the sections bulked to form a composite sample at each depth.

The DNDC model is a computer simulation model of carbon and nitrogen biogeochemistry in agro-ecosystems. The model can be used to predict crop growth, soil temperature and moisture regimes, soil carbon dynamics, nitrogen leaching, and emissions of trace gases including nitrous oxide (N_2O), nitric oxide (NO), dinitrogen (N_2), ammonia (NH_3), methane (CH_4), and carbon dioxide (CO_2). The DNDC model was used to predict the patterns of nitrate loss from each of the dairy production systems examined. The DNDC outputs were validated by comparing the modelled data with field measurements of N_2O loss based on static chamber data and leaching of N based on the residual mineral N in the soil profile in the autumn.

The annual cumulative N_2O-N emission in 2010 for the silage plots associated with the Confinement, WinterCalf and SpringCalf(Jx) systems were 1.04, 1.03 and 1.30 kg

N/ha, respectively, while the values in 2011 were 1.65, 2.21, and 1.65 kg N/ha, respectively. Total N_2O emissions were significantly higher in 2011 (1.83 kg N/ha) compared to 2010 (1.14 kg N/ha), when averaged over all silage plots associated with each of the three systems.

For the Grazed paddocks in 2010 and 2011 there was no significant difference in N_2O-N emissions from paddocks associated with the WinterCalf and SpringCalf(Jx) systems. However, over both treatments, the total loss of N_2O-N was significantly higher in 2011 (3.28 kg N/ha) compared to 2010 (1.05 kg N/ha).

For the maize plots the cumulative loss of N_2O-N was 5.21 kg N/ha in 2010 and 7.13 kg N/ha in 2011, with these values not significantly different.

The average values of N_2O-N loss, as a percentage of N applied in 2010 for the Grazing paddocks, the Silage plots and the maize plots were 0.45, 0.30 and 2.7%, respectively and for 2011 were 1.41, 0.43 and 3.22%, respectively.

Total rainfall in 2011 was only slightly higher (922 mm) than in 2010 (885 mm), so total rainfall was not the driver of N_2O emissions, rather its distribution at a time when NO_3^- was present in the soil. Peaks in N_2O emissions occurred when rain fell immediately after calcium ammonium nitrate (CAN) was applied or during the autumn-winter period which coincided with the release of soil NO_3^- from mineralisation and nitrification processes. The high emissions from maize were most likely due to the poor crop growth in both years of this study.

The mineral N (NH_4^+-N and NO_3^--N) in the soil profile, at different depths to 90 cm, was measured in each plot during November 2010. Mineral N concentrations in the zero N Control silage plot tended to be lower than in the other plots, although this was only significant at the 10 to 20 cm depth.

The total mineral N in the soil profile in the maize plots and silage plots (Confinement, WinterCalf and SpringCalf(Jx)) was 61.4, 67.5, 68.7 and 67.0 mg N/kg, respectively. There was no significant difference in the residual mineral N content in the soil associated with the silage plots for systems Confinement, WinterCalf and SpringCalf(Jx).

The amount of mineral N in the grazing paddocks associated with WinterCalf and SpringCalf(Jx) was 49.5 and 83.9 mg N/kg, respectively. However there was no significant difference between the grazing treatments at any depth.

The trend with all the plots/paddocks was that most of the mineral N (NO_3^--N + NH_4^+-N) was in the top 30 cm of soil with the proportion of N decreasing with increasing depth from 0-10 to 20-30 cm. There was no significant difference between grazing paddocks and silage plots. There was some mineral N measured below 30 cm indicating that there had been some movement of NO_3^- down the soil profile.

However, coring down the soil profile only gives a snapshot of mineral N content at a particular time, which although useful to compare treatments is not an accurate measure of N leaching. The DNDC model was therefore used to predict leaching losses from each of the dairy production systems.

A comparison between measured and modelled temporal traces of daily N_2O fluxes for the grazing paddocks associated with the WinterCalf system in 2010 and 2011 demonstrated general agreement between the measured and modelled data.

Cumulative N leaching losses ranged from 7.7 kg N/ha (2% of applied N) for the SpringCalf(Jx) silage treatment in 2011 to 14.8 kg N/ha (6.7% of applied N) for maize silage cultivation in 2011. Most of the N leaching occurred post September for all systems, with losses higher in the maize plots due to the low maize yields.

In general modelled leaching losses from all systems were low compared to the amount of residual mineral N in the soil profile in the autumn of 2010.

The Inter-Governmental Panel on Climate Change (IPCC, 2007) estimate that 1% of applied N whether as slurry or synthetic fertilizer is lost as N_2O-N. In this study only the Maize system in 2010 and 2011 (3.0% average of 2 years) was above this default value when excretal N was accounted for in the N applied.

In this study, the residual N in the soil profile in autumn was not lost as N_2O, so this suggests that there may be another N loss process that was not measured, for example, the production of benign N_2. However, the production of N_2 gas is very difficult to measure against the large background in the atmosphere, without using expensive ^{15}N stable isotope techniques.

Section 5: Four plot scale experiments examining strategies to reduce phosphorus losses from applied slurry

While P losses within the experimental site as a result of grazing have been examined within Section 3, it was realised that large scale plots did not provide the optimum approach by which to examine factors influencing P losses from slurry applied to intensive grassland systems. For this reason, four small scale detailed experiments were conducted on an adjoining site to examine strategies by which to reduce P losses. These studies were undertaken as part of a PhD linked to the main study, with a brief overview of each of these experiments presented below:

Experiment 1 (Phosphorus losses from low emission slurry spreading techniques) was designed to investigate the effect of slurry application technique on slurry-associated phosphorus concentrations in runoff. Dairy cow slurry was applied to freshly harvested grassland stubble by hand to simulate splashplate, trailing shoe, and shallow injection spreading techniques. Both the trailing shoe and shallow injection techniques were applied 'across' the slope of the field, or 'down' the field slope.

Slurry application via the trailing shoe and shallow injection reduced dissolved reactive phosphorus (DRP) concentrations in runoff by 37 and 47%, respectively, relative to traditional splashplate spreading techniques. There was no effect of application direction (across or down slope) on P concentrations in runoff. In addition, slurry was also applied to a four-week regrowth, using the same slurry spreading techniques listed above. In contrast, slurry spreading technique had no effect (P>0.05) on P concentrations in runoff following this application. This was attributed in part to the very dry weather and soil conditions which resulted in problems generating runoff at this time. Nonetheless this experiment clearly demonstrated the potential of the trailing shoe and shallow injection slurry spreading techniques to reduce DRP concentrations in runoff, compared with the traditional splash plate technique.

The second experiment (**Experiment 2:** The impact of herbage regrowth interval on phosphorus losses in runoff post slurry application) was designed to investigate the effect of herbage mass on P concentrations in runoff, following slurry application with the trailing shoe technique. Slurry was applied by hand to plots with three levels of herbage cover: a zero-day regrowth, a 10-day regrowth, and a 20-day regrowth.

Dissolved reactive P concentrations in runoff were significantly reduced ($P<0.05$) following slurry application to the 10-day and 20-day herbage regrowth, relative to the zero-day regrowth treatment. In contrast, herbage regrowth had no significant effect on PP concentrations in runoff. Thus this experiment demonstrated that allowing a grass sward to recover for between 10 to 20 days following harvest before applying slurry, can be highly effective in reducing P losses in runoff.

Experiment 3 (The impact of slurry application method on phosphorus loss in runoff from grassland soils during winter and early spring) examined the effect of slurry application technique (Splashplate/Trailing shoe) and timing of slurry application (winter/early spring) on P concentrations in runoff. Slurry was applied by hand on four occasions during the winter/spring period (7 December, 18 January, 1 March, and 12 April) simulating either the splashplate or trailing shoe technique.

Following each application, DRP, PP, and total P concentrations in runoff were significantly greater ($P<0.05$) from the Splashplate treatment than from the Trailing shoe treatment. In addition, DRP concentrations in runoff from the Splashplate treatment were greater following the December and March slurry applications, than following the January and April applications, with the former application dates coinciding with periods of higher volumetric soil moisture content. In contrast, P concentrations in runoff from the Trailing shoe treatment did not differ between the four slurry application dates. While again highlighting the potential of the trailing shoe system to mitigate against P losses from applied slurry, this experiment also demonstrated that soil moisture content, and not season *per se*, was a significant driver of P losses.

The fourth experiment (**Experiment 4**: Phosphorus loss in runoff following the application of anaerobically digested slurry to grassland) was designed to investigate the effect of anaerobic digestion of slurry on P losses in runoff following slurry application to grassland. Both anaerobically digested (AD) slurry and undigested (UD) slurry were applied to grassland via a simulated splashplate spreading technique.

Despite AD slurry having a higher (P<0.001) water extractable P content than UD slurry, DRP concentrations in runoff were unaffected (P>0.05). In contrast, both dissolved unreactive P and PP concentrations in runoff from the AD slurry treatment were lower (P<0.05) than from the UD slurry treatment. The results of this experiment highlight that anaerobic digestion of slurry does not increase the risk of P being lost in runoff following slurry application.

INTRODUCTION

On 1 January 2007 the whole of Northern Ireland was designated a Nitrates Vulnerable Zone (NVZ) under the EU Nitrates Directive. This has had a number of implications for dairy farmers, including the implementation of ‘closed periods’ for fertiliser and manure spreading, the requirement to have 22 weeks of slurry storage capacity, and the introduction of a stocking rate limit. The latter is set at 170 kg manure nitrogen per hectare, which within Northern Ireland where a typical dairy cow produces 91 kg of manure nitrogen/year, is equivalent to 1.87 cows per hectare.

Nevertheless, many farms have traditionally operated at stocking rates of greater than 1.87 cows per hectare, and therefore Northern Ireland applied to the EU for a ‘derogation’ from the Nitrates Directive. This application was successful, and as a result dairy farmers with predominantly grassland-based systems (greater than 80% grassland) have been able to apply for a farm derogation to allow them to operate at a stocking rate of up to 250 kg organic nitrogen per hectare (2.74 cows per hectare).

However, this derogation is subject to periodic review by the EU. In order for a new derogation application to be approved, a number of conditions which were set by the EU must be met. One of these is a requirement to conduct research into intensive grassland-based milk production systems, with the aim of improving nutrient utilisation. Article 8(6) of the EU Derogation Document states that:

‘*A study shall be conducted in order to collect, by the end of the derogation period, detailed scientific information on intensive grassland system in order to improve nutrient management. This study will focus on nutrient losses, including nitrates leaching, denitrification losses and phosphate losses, under intensive dairy production systems in representative areas’*

To address this requirement, a study was established at AFBI Hillsborough, the only dairy cow research facility within Northern Ireland, and one which is situated in an area which is broadly representative of dairying in large parts of Northern Ireland.

The experimental systems examined within this study were chosen to represent the diverse range of milk production systems which are adopted within Northern Ireland. Two of the systems involved winter calving Holstein cows, with these systems defined as either 'Total Confinement' (cows housed throughout the entire lactation) or Conventional (winter housing, summer grazing). A third system involved spring calving dairy cows, with this defined as 'Low input'. The latter system was replicated using both Holstein and Jersey x Holstein crossbred cows. Each system was designed to operate at a stocking rate greater than 1.87 cows per hectare (ie. requiring a derogation), with each system designed to minimise nitrogen and phosphorus loss to the environment.

Cows remained on this study for three successive lactations. In addition to examining the effects of systems on cow performance, detailed nutrient loss measurements were conducted within three of the systems. In addition, in view of the difficulties associated with measuring phosphorus losses within a 'system', a series of detailed component studies were undertaken to examine strategies by which to reduce phosphorus losses from grazing systems.

This report provided a detailed examination of all the research undertaken within this project.

SECTION 1

The performance of dairy cows within four contrasting grassland-based systems of milk production over three successive lactations

INTRODUCTION

While the whole of Northern Ireland was designated a Nitrates Vulnerable Zone (NVZ) under the EU Nitrates Directive in 2007, dairy farmers with predominantly grassland-based systems (greater than 80% grassland) have been able to apply for a farm derogation to allow them to operate at a stocking rate of up to 250 kg manure nitrogen per hectare (2.74 cows per hectare). However, in granting this derogation the EU Commission established a number of monitoring requirements, one of which (Article 8(6) of the EU Derogation Document) states that: '*A study shall be conducted in order to collect, by the end of the derogation period, detailed scientific information on intensive grassland system in order to improve nutrient management. This study will focus on nutrient losses, including nitrates leaching, denitrification losses and phosphate losses, under intensive dairy production systems in representative areas'*

A wide range of milk production systems, some involving different dairy cow genotypes, are practiced on Northern Ireland (NI) dairy farms. For example, systems differ in terms of calving season (Autumn, Spring and 'all year'), annual per cow milk production (4,800-10,500 kg), stocking rate (1.0-3.5 cows/ha), annual concentrate feed level (0.5-4.0 t/cow), type of forage offered (grass silage, maize silage, whole crop silage and grazed grass), management regime (total confinement, partial confinement, traditional winter housing-summer grazing systems, and low input spring calving systems). In addition, while the Holstein-Friesian is the predominant dairy cow genotype within Northern Ireland, some of these systems operate with alternative cow genotypes.

The relative merits of each system are often debated, with data comparing physical and financial performance of different systems normally obtained from national data-bases or 'benchmarking'. However, robust detailed comparative scientific data on physical performance within contrasting systems is more difficult to obtain, with this due to the high cost associated with undertaking expensive systems-type research. This expense is increased in that systems-type research is often conducted over a number of years in order to mitigate against year to year variations in seasonal weather patterns.

Nevertheless, a number of system-type studies have been undertaken in recent years. For example, Ferris *et al*. (2003) compared four contrasting grassland-based systems of milk production involving winter calving cows over three successive lactations, while in a separate study, Ferris (2013) compared three contrasting spring calving milk production systems over three successive years. In a single lactation study, Vance *et al*. (2012) compared a total confinement system with a spring calving system. However, there appear to be no comparisons of total confinement vs conventional winter calving-summer grazing systems, or comparisons of winter vs spring calving production systems, with these identified as key knowledge gaps.

To address the monitoring requirements established by the EU Commission, a systems study was established at AFBI Hillsborough, the only dairy cow research facility within Northern Ireland, and one which is situated in an area which is broadly representative of dairying in large parts of Northern Ireland. The experimental systems were chosen to reflect the diversity of systems in place within Northern Ireland, and to help fill some of the systems 'knowledge gaps' identified earlier. Each system was designed to operate at stocking rates whereby a 'derogation' from the Nitrates Directive Action programme would be required, while in addition, components of each system were designed to minimise N and P surpluses, and nutrient loss to the environment. Cows remained on this study for three successive lactations.

This section of the report will examine cow performance, nutrient balances, the carbon footprint and economics of each of the four systems examined.

MATERIALS AND METHODS

Experimental Overview

This experiment was conducted at the Agri-Food and Biosciences Institute, Hillsborough (latitude 54º27'N; longitude 06º04'W) between October 2008 and April 2011. Cows were managed on one of four grassland-based milk production systems

(Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx) over three successive lactations.

Animals and allocation

The study involved eighty dairy cows each year. Sixty of these were high genetic merit Holstein cows, 20 cows on each of systems Confinement, WinterCalf and SpringCalf (H), while a further 20 Jersey x Holstein crossbred dairy cows were managed on system SpringCalf(Jx). Cows on systems Confinement and WinterCalf calved between October and April each year, and had mean calving dates of 18 November, 14 December and 24 December (Confinement) and 21 November, 12 December, and 7 December (WinterCalf) during Years 1, 2 and 3, respectively. With the SpringCalf systems cows calved between January and April each year, and had mean calving dates of 5 February, 1 February and 20 February (SpringCalf(H)) and 8 February, 7 February, and 11 February (SpringCalf(Jx)) during Years 1, 2 and 3, respectively.

Across the three years of the experiment winter calving Holstein cows on systems Confinement and WinterCalf had a mean predicted transmitting ability (PTA_{2010}) for milk and fat plus protein yield of 222 (s.d., 208.6) kg and 26 (s.d., 12.6) kg, respectively and a mean Profitable Lifetime Index (£PLI) of £88 (s.d., 48.4), while the respective values for Holstein cows on SpringCalf(H) were 98 (s.d., 280.5) kg, 18 (s.d., 15.3) kg and £62 (s.d., 53.3), respectively (November 2013, sire proof run). The Jersey x Holstein crossbred cows (F1) on system SpringCalf(Jx) were the offspring of a breeding programme involving randomly selected Holstein-Friesian cows from the AFBI-Hillsborough herd and Jersey sires of both Danish and New Zealand origin.

During Year 1, 15 multiparous cows and 5 primiparous cows were allocated to each of the four systems. Cows allocated to Confinement and WinterCalf were balanced according to calving date, parity, pre-calving live weight and body condition score (BCS), sire, and PTA for fat plus protein yield. Cows allocated to systems SpringCalf(H) and SpringCalf(Jx) were balanced according to calving date and parity, and dam PTA for fat + protein yield. Cows remained on the same management system for the duration of the experiment, or until removed from the experiment. Cows that were removed during or at the end of Years 1 and 2 were replaced at the start of Years 2 and 3, respectively. Replacement animals were largely primiparous, although on occasions multiparous cows were used as replacements. Replacement animals were balanced across systems according to the traits described above.

Housing

Cows were transferred to cubicle accommodation within 36 hours of calving. Cubicles were fitted with rubber mats (2.2 m x 1.25 m) which were scraped down daily, bedded with sawdust twice weekly and treated with lime weekly. Sufficient cubicles (approximately 10% excess) were available for all cows within the group. The concrete walkways within the house were scrapped a minimum of six times each day by an automatic scrapper system.

When confined, cows accessed their diets via a Calan gate feeding system (American Calan, Northwood, NH, USA). Each Calan gate was linked to an automatic cow identification system, which allowed cows to gain access to a feed box mounted on a weigh scale (Griffith Elder, Bury St Edmunds, UK), thus allowing individual food intakes to be measured. Cows on each of the four systems accessed their food via separate boxes, with an average three cows sharing each box.

Description of management systems

Key aspects of each of the four management systems are summarised in Table 1.1, with full details presented below. In addition, key dates within the experiment are presented in Table 1.2, while the ingredient composition of each of the concentrates offered during the experiment is presented in Table 1.4. Changes in the availability and cost of some ingredients meant that the ingredient composition of the concentrates offered throughout the experiment varied from year to year. Due to concerns about contaminated product being in circulation, sugar beet pulp was removed from all concentrates midway through Year 1 (June 2009) and was replaced with citrus pulp for the remainder of the year.

Systems Confinement and WinterCalf:

During the winter confinement period cows on systems Confinement and WinterCalf were managed as a single group, while accessing their food via separate feed boxes.

Cows on system Confinement were confined throughout the duration of the experiment. Once transferred to the experimental group following calving, cows were offered a diet containing concentrates and forage in a 55:45 DM ratio, with the forage component of the diet comprising grass silage and maize silage in a 80:20 DM ratio. In addition, all cows were offered 1.0 kg concentrate/day (0.5 kg at each milking) through the parlour. The ingredient composition of the concentrate offered is presented in Table 1.3. When cows on this treatment were a mean of 180 days calved, the concentrate:forage mix within the ration was changed to 45:55 DM ratio, with the proportions of grass silage and maize silage within the forage proportion remaining unchanged at 80:20. At this stage the ingredient composition of the concentrate was changed to that presented in Table 1.3. Cows remained on this diet until drying off.

Cows on WinterCalf were managed identically to those on Confinement until the commencement of the grazing period, which always occurred prior to the change in diet with the Confinement cows, as described above. Cows on this system commenced grazing on 29 March, 27 March and 29 March in Years 1, 2 and 3, respectively, initially grazing for approximately 8 hours per day (milking to milking), with full time turnout achieved on 16 April, 19 April and 14 April during Years 1, 2 and 3, respectively. During this transition period the proportion of concentrates included within the complete diet was reduced gradually, while 4.0 kg/day of a grazing concentrate (Table 1.3) was introduced into the diet, split between two equal feeds, and offered at milking. At full turnout, the concentrate feed level offered through the parlour was increased to the ‘target’ level of 5.0 kg/cow/day. On a number of occasions during the three-year experimental period, herbage shortages required that concentrate feed levels were increased to either 6.0 or 7.0 kg/cow/day, with these short term adjustments in feed levels lasting for between 1-3 weeks. Cows grazed full time until 19 October, 2 October and 9 October (Years 1-3 respectively), and were completely housed on 19 October, 1 November and 25 October (Years 1, 2 and 3, respectively). During this transition from full time grazing to full time re-housing, cows normally grazed during the day (between morning and evening milking) and were housed at night and offered the grass silage and maize silage in the same ratio as cows on the Confinement diet (while continuing to be offered their full grazing concentrate allowance through the parlour). Following full time re-housing, cows moved onto the same ration as was offered during late lactation to cows on system Confinement, as described previously.

Systems SpringCalf(H) and SpringCalf(Jx):

From calving until the start of turnout, cows on these two systems were housed in a single group, but accessed their ration through separate Calan gates. The diet offered comprising grass silage and concentrates in a 70:30 DM ratio, together with 1.0 kg concentrate/cow/day offered through the parlour (0.5 kg at each milking). The ingredient composition of this concentrate is presented in Table 1.3. Cows on these two systems commenced grazing on 9 March, 9 February and 26 February during Years 1-3, respectively, with these cows moving to full time grazing on 2 April, 21 April and 14 April, respectively. During this transition period the duration of the daily grazing period increased from approximately two hours/day to approximately 12 hours/day (08.00-20.00 hrs), prior to full time grazing commencing. During this 'transition' grazing period cows were allocated sufficient herbage to allow them to graze to a residual sward height of approximately 5.0 cm, while during the non-grazing part of the day cows were initially offered grass silage *ad libitum*, together with their full daily winter concentrate allocations. However, part way through this transition period (21 March, 12 April and 30 March, in Years 1-3, respectively), the concentrate component of the complete diet was reduced, while 4.0 kg/day of a grazing concentrate was introduced into the diet, and offered through the parlour. Following full time turnout, concentrates offered in the parlour were reduced over a 10-20 day period until the 'target' concentrate feed level of 1.0 kg concentrate/cow/day was achieved. On a number of occasions, as a result of adverse weather or grass shortages (especially in late lactation), concentrate feed levels were increased above the target level, to 2.0 kg/cow/day. During the early grazing period (from full-time turnout until the risk period was deemed to have passed, normally late April/early May) paddocks were dusted with calcined magnesite (210 g/cow/day) on a daily basis to ensure cows had an adequate intake of magnesium.

Cows continued to graze full time until 19 October, 22 October, and 9 October (in Years 1-3 respectively), followed by a period of part time grazing (grazing by day and being offered grass silage by night), with full time re-housing achieved on 23 October, 1 November, and 9 November (Years 1-3, respectively). Following full time re-housing, these cows were offered a diet comprising grass silage and concentrate (75:25 DM ratio), plus 1.0 kg concentrate per cow per day through the parlour, until drying off.

Dry period

Cows with a BCS of ≥2.50 were dried off eight weeks pre-calving, or if average weekly milk yields fell below 5.0 kg/day. Cows with a BCS of 2.25 or ≤2.00 were dried off either 10 or 12 weeks pre-calving, respectively. Cows were dried off abruptly following morning milking and treated with long acting antibiotic tubes and a teat sealant. Cows that were non-pregnant remained on their experimental treatment for the same mean number of days as the pregnant cows within their experimental groups, after which they were removed from the experiment. Pregnant cows were moved to a dry cow group post drying off, and when possible, their intakes measured using a Calan gate feeding system as described earlier.

While confined, dry cows on all treatments were offered a grass silage (normally produced from secondary re-growth herbage) supplemented with 100 g/cow/day of dry cow mineral and vitamin supplement. Dry cows on Confinement were confined throughout the dry period. With WinterCalf, cows that were dried off early were grazed without supplementation within the grazing area for that system (mean of 17 grazing days/cow) until three weeks pre-calving, before being confined. All cows on WinterCalf that were dried off after mid October were housed immediately. Cows on the SpringCalf systems were dried off following re-housing and offered grass silage throughout the dry period.

Culling

Cows that were removed from the experiment due to health problems during the grazing season were replaced with ‘spare cows’ until the end of that grazing season, in order to maintain grazing group sizes (20 cows per group). Cows removed either during or at the end of Years 1 and 2 were replaced by new experimental cows at the start of the subsequent lactation. Cows were removed from the study as ‘infertile’ if they were not confirmed pregnant by the 31 December each year. These were subsequently replaced as described earlier.

Breeding programme

Throughout the three years of the experiment breeding for cows on Confinement and WinterCalf commenced during the first week of December, and continued until the first week of July. With the SpringCalf systems a 14-week breeding season was adopted, commencing start of April and finishing mid July. A voluntary waiting period of a minimum of 42 days prior to the start of breeding was adopted with all cows. Throughout the experiment cows were bred via artificial insemination approximately 12 hours after visual observation of oestrus. Holstein-Friesian cows were bred to Holstein sires while the crossbred cows were bred to sires of the Swedish Red and White breed. Pregnancy was diagnosed via rectal scanning at day 60 post AI. Cows were not treated with any fertility drugs until they were a minimum of 52 days post-calving. The exception to this were cows that displayed symptoms of uterine infection, in which case treatment was given as soon as the problem was identified. Cows which had not been observed on heat prior to day 52 post-calving were inspected by a veterinary surgeon, and treated as appropriate.

Diet preparation during the confinement periods

With Confinement and WinterCalf, sufficient silage (grass silage and maize silage) for these two systems was placed in a complete diet mixer wagon (Redrock, Co. Armagh, Northern Ireland) and mixed for approximately five minutes. The required quantity of concentrates was then added to the silage in the mixer wagon, and mixing continued for a further five minutes. This mixed ration was then deposited in the appropriate feed boxes for cows on these two systems.

With the SpringCalf systems, sufficient grass silage and concentrates for these two systems were placed in the mixer wagon, and mixed for approximately five minutes. This mixed ration was then deposited in the appropriate feed boxes for cows on each of these two systems. With all systems, the diet was offered at proportionately 1.07 of the previous day’s intake. Uneaten food was removed from the feed boxes daily at approximately 08:30-09.00 hours and fresh food offered at between 09:00 and 10:30 hours.

Silages offered

Grass silage offered was harvested from predominantly perennial ryegrass (*Lolium perenne*) based swards using a self-propelled precision chop forage harvester, treated with a bacterial inoculant at harvest, and ensiled in walled silos. With system Confinement, the grass silages offered were produced from primary growth, primary re-growth and secondary re-growth herbage, with these silages offered for approximately proportionally 0.40, 0.35 and 0.25 of the total cow feeding days within the experiment, respectively. These proportions represented herbage yields for primary growth, primary re-growth and secondary regrowth harvests within a 'three cut' grass silage system (Mayne and Gordon, 1986). Silages offered prior to turnout with the WinterCalf and SpringCalf systems was made from primary growth herbage, while silage offered following re-housing was produced from primary and secondary re-growth herbages. Maize silage offered was sown under plastic in spring each year, and harvested between mid October and early November.

Pasture Management

The grazing area for the cows on this experiment encompassed two 'field blocks' with a total area of approximately 22 ha. Swards were permanent pasture consisting of a perennial ryegrass-based sward. A rotational paddock grazing system was adopted throughout the grazing season, with the 'core grazing area' comprising 21 one-day paddocks for each of systems WinterCalf, SpringCalf(H) and SpringCalf(Jx), with paddock sizes being 0.17 ha, 0.20 ha and 0.20 ha, for each of these systems, respectively. Within each system paddocks were arranged in pairs, with pairs of paddocks from each system spread across the grazing platform, thus taking account of variations in sward quality and topography of the fields.

With systems WinterCalf and SpringCalf(Jx), on one occasion during each grazing cycle the 'one-day' paddock was subdivided into four mini-paddocks, to allow nutrient loss measurements to be undertaken, as described in Section 2. On the day when cows were due to graze this 'paddock', cows from each of these two systems were divided into four random groups (each of five cows) on a laneway adjacent to the paddock, with each group of five cows allowed to graze one of the mini-paddocks over a 24-hour period. With this exception, grazing for all three groups was managed as per a normal paddock rotational grazing system. Additional grazing paddocks were introduced into the rotation as the season progressed, while paddocks that were not grazed during a rotation (due to excess grass being available, as determined by a 'grass wedge' grassland management tool) were either grazed by a group of non-experimental cows, or cut and baled for silage.

Target fertiliser nitrogen (N) application levels were as follows: a pre-grazing application of urea (proportionally 0.46 N) across the grazing area prior to turnout at 28 kg N/ha, followed thereafter by calcium ammonium nitrate (CAN: proportionally 0.27 N) at a rate of 45, 30, 30, 30, 25, 25 and 20 kg N/ha following each of grazing cycles 1-7, respectively. Thus over the entire grazing season the target total fertiliser N application rate on the 21 core grazing paddocks was 238 kg N/ha. However, wet weather, delayed turnout dates and a number of other management issues meant that it was not always possible or desirable to follow this schedule. The entire grazing area was trimmed ('topped') to approximately 50 mm mid way through the grazing season.

Measurements

Animal measurements

Cows were milked twice daily between 06:00 and 08:00 hours and between 15:00 and 17:00 hours, with milk yields recorded automatically at each milking. Milk fat, protein and lactose concentrations were determined weekly using two consecutive (morning and evening) milk samples (Milkoscan, Model FT 120, Foss UK Ltd., Warrington, UK) while milk somatic cell count (SCC) was determined monthly using a Fossomatic 360 (Foss Electric, Hillerød, Denmark). Milk samples were preserved (lactab Mark III, Thompson and Cooper Ltd., Lydney, UK) and stored at 4ºC until analysed. Cow live weight was recorded automatically after each milking and an average weekly live weight subsequently calculated. Body condition score of lactating cows was assessed fortnightly by a trained operators using a five point scale (Edmondson *et al.,* 1989), where 1 = emaciated and 5 = extremely fat. Locomotion score was recorded fortnightly by a single trained operator using a five point scale (Manson and Leaver, 1988), where 1 = no unevenness in gait or tenderness and 5 = difficulty in walking and adverse effects on behaviour pattern. Blood samples were taken from the coccygeal vein of each cow between 06:30 and 08:30 hours at weeks 2, 6, 10 (±3 days), 20, 30 and 40 (±7 days) post-calving. Blood plasma was recovered via centrifugation and stored at -20ºC until analysis for β-hydroxybutyrate (BHB), non-esterified fatty acids (NEFA) (using a Wako kit, Wako Chemicals GMBH, Germany), glucose and urea (using Olympus kits, Olympus Life and Material Science Europa, Germany) using a Chemistry Immuno Analyser (Olympus AU640).

During the periods when cows on each of the four systems were housed, individual food intakes were measured daily using the Calan gate feeding system described previously. During the grazing season (from start of turnout to full-time re-housing) mean daily herbage DM intakes of lactating cows were calculated weekly for each cow from animal performance data, and total herbage intake over the grazing season subsequently calculated. Within this calculation, milk energy content was determined from weekly milk samples using the equations of Tyrrell and Reid (1965), while mean daily live-weight change over the grazing period (full time grazing period only) was determined by linear regression of weekly live-weight data. Total energy required for maintenance, production, tissue change, pregnancy (where appropriate) and walking (assumed as 2.0 km/day for cows grazing full time) was determined using the equations contained within 'Feed into Milk (FIM)', the UK dairy cow feed rationing system (Agnew *et al.*, 2004). The Metabolisable Energy (ME) content of herbage was measured using Near Infrared Reflectance Spectroscopy (NIRS), as described below, while the ME content of the grazing concentrates offered was assumed as 12.4 MJ/kg DM (based on published values for individual ingredients: FeedByte). During the part turnout and part re-housing periods, at the start and end of the grazing season respectively, the calculation took account of ME intake from grass silage, maize silage and the 'winter' concentrates offered at this time. However, no account was taken of live-weight change at this time due to large 'gut-fill' associated changes in live weight during these 'transition' times. Herbage intakes of non-lactating grazing cows were also determined using a similar technique, with the calculation taking account of the energy required for maintenance and pregnancy only. No account was taken of changes in live weight as live-weight changes during the non-lactating period were confounded by the substantial energy requirements of the growing foetus at this time.

Throughout the grazing season pre- and post-grazing sward heights were measured daily within the grazing area for each of systems Conventional, SpringCalf(H) and

SpringCalf(Jx) (40 measurements in a 'W' formation) using a rising plate meter (Jenquip, Feilding, New Zealand).

Throughout the study cows with health problems were treated by either a veterinary surgeon or by a member of Institute staff, as appropriate. All incidences of mastitis and lameness were recorded throughout the experiment with an incidence defined as one where antibiotic treatment was used.

Feed chemical analysis

Grass and maize silages offered were sampled daily and analysed for oven DM content, while dried samples were retained weekly, bulked for each 4-week period, and subsequently analysed for acid detergent fibre (ADF), neutral detergent fibre (NDF), ash and phosphorus concentrations. In addition, maize silage samples were dried at 60ºC on one occasion each week, bulked for each 4-week period, and analysed for starch content. Fresh silages were analysed weekly for gross energy (GE), nitrogen (N), pH, ammonia-N and volatile components, while the metabolisable energy (ME) content of fresh silage was estimated weekly using Near Infrared Reflectance Spectroscopy (NIRS), as described by Park *et al.* (1998). During the grazing season herbage pluck samples were taken within the grazing area for each system once weekly, and dried at 60ºC. Dried samples were bulked for each two-week period and analysed for ADF, NDF, N, GE, WSC, phosphorus and ash concentrations. In addition, a fresh sample of grass from the grazing area within each system was analysed weekly for metabolisable energy content using Near Infrared Reflectance Spectroscopy (NIRS) as described by Park *et al.* (1998) for grass silage, but using a calibration equation developed for fresh grass. A sample of each concentrate type offered was collected weekly, with samples bulked for each four-week period. Bulked samples were analysed for N, ADF, NDF, GE, ash and phosphorus content. The feedstuffs offered were analysed as described by Ferris *et al.* (1999), except for the GE content of silage which was determined on a fresh sample, as described by Porter (1992).

Statistical analysis

Data from 17 cows were excluded from the analysis due to mastitis/udder problems (n = 8), stomach/digestive problems (n = 3) and miscellaneous reasons (n = 7), with these cows treated as missing values within the analysis. Data were analysed using GenStat Version 11.1 (Payne *et al.*, 2008). Food intake, milk production data, parameters describing live weight and body condition score data at fixed time points, and continuous fertility data were analysed using Residual Maximum Likelihood (REML) analysis using a repeated measures mixed model. The model included the following terms as fixed effects: lactation number (1, 2, 3, 4+), year (1, 2 or 3), milk production system (Confinement, WinterCalf, SpringCalf(HF) and SpringCalf (Jx)), while cow + cow within lactation were included as random effects. Lactation length was not included within the model as differences in lactation length between systems were due in part to differences in performance within the systems. Weekly live weight data and fortnightly condition score data (until week 44 post calving) were analysed using REML analysis using a repeated measures mixed model, with the model containing the following terms as fixed effects: lactation number (1, 2, 3, 4+), year (1, 2 or 3), week of lactation, and system (Confinement, WinterCalf, SpringCalf(HF) and SpringCalf (Jx)) and system x week of lactation, while cow and cow within week of lactation were included as random effects. Monthly locomotion score data were analysed using the same model, except that week of lactation was replaced by month post calving within the model. Blood metabolite data were analysed using a similar model, with week of lactation defined as 2, 6, 10, 20, 30 and 40 post calving. Binomial fertility and health data were analysed using a Generalised Linear Mixed Analysis (Binomial Distribution and Logit Link Function) with lactation number (1, 2, 3, 4+), milk production system (Confinement, WinterCalf, SpringCalf(HF) and SpringCalf (Jx)), year (1, 2 or 3), and treatment x year included as fixed effects, and with cow as the random effect. The bootstrap method was used to generate SEMs, while significance of each fixed effect was assessed by comparing a Wald statistic against the appropriate Chi square distribution.

RESULTS

Grass silages offered within all systems (Table 1.4) were well preserved, and had similar chemical compositions and nutritive values (crude protein and metabolisable energy contents). Maize silage offered with the Confinement and WinterCalf systems had a mean DM content of 315 g/kg and a mean starch content of 268 g/kg DM.

Similarly, the herbage offered within systems WinterCalf, SpringCalf(H) and SpringCalf(Jx) had a mean crude protein and ME content of 227 g/kg DM and 11.7 MJ/kg DM, respectively (Table 1.5). Concentrates offered with the Confinement and WinterCalf systems had a mean P concentration of 3.8 (pre day 180) and 3.7 (post day 180) g/kg DM, while those offered with the SpringCalf systems when confined had a P content of 4.2 g/kg DM (Table 1.6). The grazing concentrate offered had a P concentration of 3.9 g/kg DM.

Mean pre- and post-grazing sward heights (across the three years of the experiment) were 8.7 and 5.1 cm with WinterCalf, 8.7 and 5.0 cm with SpringCalf(H) and 8.6 and 4.9 cm with SpringCalf(Jx), respectively (Table 1.7), while mean grazing stocking rates with these three systems were 5.09, 4.30 and 4.30 cows/ha, respectively.

The mean days in milk with systems Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx) were 326, 317, 303 and 302 ($P<0.001$), respectively, while the mean length of the dry period for cows on each of these systems was 70, 71, 80 and 77 days ($P=0.110$), respectively (Table 1.8). In the case of WinterCalf, on average 17.6 days of the dry period occurred while grazing, while with each of the other systems, cows were housed for the entire dry period.

During the lactation period, total silage intakes were higher with Confinement than with any other system ($P<0.001$), while intakes of maize silage were higher ($P<0.001$) with Confinement than with WinterCalf (Table 1.8). Full lactation concentrate DM intakes decreased from the Confinement through to the SpringCalf systems ($P<0.001$), while total herbage intakes were calculated as 2041, 2788 and 2692 kg DM/cow with systems WinterCalf, SpringCalf(H) and SpringCalf(Jx), respectively. Total DM intakes over the full lactation decreased from the Confinement through to the SpringCalf systems ($P<0.001$). Intakes did not differ between the SpringCalf(H) and SpringCalf(Jx) systems for silage, concentrates, grass or total DM intake.

Dry cows on the WinterCalf system were calculated to have a mean grass intake of 164 kg/cow. Total forage intakes (grazed grass plus grass silage) during the dry period were higher with the SpringCalf systems than with either the Confinement or WinterCalf systems ($P = 0.010$).

Full lactation milk yields were significantly higher with the WinterCalf system than with either of the spring calving systems, while yields with the Confinement system were higher than for the WinterCalf system (P<0.001) (Table 1.9). Milk fat and protein concentrations differed between systems, being higher with SpringCalf(Jx) than with any of the other systems (P<0.001). Yields of milk fat, protein and fat plus protein were highest with Confinement, and lowest with the SpringCalf systems (P<0.001), with energy corrected milk yield following a similar trend. Milk phosphorus concentrations were unaffected by system, while somatic cell score was lowest with the SpringCalf(H) system.

There was a significant effect of system and time, and a significant system x time interaction, on weekly live weights (Figure 1.1) and fortnightly condition scores (Figure 1.2) over the first 44 weeks of lactation (P<0.001). Holstein cows on SpringCalf(H) had a lower mean live weight than those on the Confinement or WinterCalf system, although these differences were not apparent at calving (Table 1.10). In addition, Holstein cows on SpringCalf(H) tended to be lighter at the nadir live weight, lost more live weight to nadir, and took longer to reach nadir than Holstein cows on any other system (P<0.001). The crossbred cows on SpringCalf(Jx) were lighter than cows on any other system throughout the lactation (P<0.001), while not differing from the cows on SpringCalf(H) in live-weight loss to nadir (P>0.05). While there was no difference between systems in the condition score of the cows at calving, cows on Confinement (H) had a significantly higher mean condition score and dry off condition score than cows on any other system.

Days to first observed heat was unaffected by system (P=0.099), while tending to be lowest with SpringCalf(Jx) (Table 1.11). Conception to first service and to first and second service was unaffected by treatment, although the letter tended to be highest with the SpringCalf(Jx) (P=0.114). The interval from calving to conception was lower (P<0.05) with SpringCalf(JX) than with Confinement. Neither pregnancy rate at the end of the breeding season nor calving interval differed between systems. System had a significant effect on the proportion of cows with one or more cases of mastitis, with cows on SpringCalf(Jx) having fewer cases of mastitis than cows on either Confinement or WinterCalf (P>0.05) while cows on SpringCalf(H) had fewer cases of mastitis than those on Confinement (P>0.05). While the proportion of cows with at least one case of lameness was unaffected by system, incidence tended to be lowest with SpringCalf(Jx) (P= 0.120). Mean locomotion scores (P<0.001) were 2.76, 2.65, 2.61 and 2.54 for Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx) respectively, with mean locomotion score decreasing from the Confinement to the SpringCalf(Jx) system (P<0.001). In addition, locomotion score increased with time post calving (P<0.001), while there was a significant system x time interaction (P=0.008: Figure 1.3).

Plasma NEFA concentrations (Figure 1.4) decreased with time post-calving (P<0.001), while there was a significant effect of system (P<0.001) on NEFA concentrations, and a significant system x time interaction (P=0.002). System had no effect on plasma glucose concentrations (P=0.194), while there was a significant time (P<0.001) and system x time interaction (P=0.002) (Figure 1.5). Plasma phosphorus concentrations (Figure 1.6) were unaffected by system (P=0.104), while increasing with time post calving (P<0.001). In addition, there was a significant system x time interaction (P=0.004) for plasma P concentrations. Plasma urea concentrations (Figure 1.7) tended to increase with time post calving (P<0.001), while there was a significant effect of system (P<0.001) on urea concentrations, and a significant system x time interaction (P<0.001).

DISCUSSION

Four contrasting grassland-based milk production systems were examined, with these systems differing in management strategy adopted, concentrate inputs, forage type and quality, and in the case of the SpringCalf systems, the genotype of the cows involved.

Food intake and cow performance

Differences in total DM intakes, and the very different contributions of the different feedstuffs to intakes within each of the systems, is highlighted in Table 1.8. While total DM intake is confounded to some extent by the longer lactation length with cows on Confinement and WinterCalf, compared to those on SpringCalf (18 days longer, on average), total concentrate intake (and the associated effects on milk output) is the primary driver of the differences in total DM intake between systems. As expected, the intake of conserved forages was significantly higher with the Confinement system, than with any other system, a reflection of the absence of a grazing period within this system. While intakes of conserved forage with the WinterCalf system might have been expected to be substantially higher than for the SpringCalf systems, due to the much longer confinement period with this system, the actual difference was relatively small (not significant for grass silage). This reflects the higher concentrate feed level with this treatment during the periods of confinement.

The different milk outputs across the systems are in agreement with what was expected, with Holstein cows on Confinement producing significantly more milk (890 kg) than those on WinterCalf, with these in turn producing significantly more milk (1979 kg) than those on SpringCalf(H). Holstein cows on Confinement and WinterCalf were balanced for genetic merit, and as such difference in performance between these two systems can be attributed entirely to the systems imposed. However, Holstein cows on SpringCalf had a numerically lower PTA for milk yield and fat plus protein yield (124 kg and 8 kg lower, respectively), than those on the other two systems, and this may have contributed to a small extent to the lower level of performance observed with this system. Nevertheless, it is likely that the primary driver of the difference in performance between these three systems was the different concentrate inputs. For example, when full lactation energy corrected milk yield (Y) for each of these three systems is plotted against full lactation concentrate intake (x: kg fresh basis), the following relationship was identified: $Y = 1237\ x + 5663$ ($r^2 = 0.998$), the high r^2 value demonstrating the high degree of linearity of the response. The mean milk yield response to concentrate feeding across systems was 1.23 kg milk/kg concentrate offered.

Other factors that may have contributed to differences in performance between these three systems include concentrate composition, the quality and type of the forage offered (grass silage, maize silage, grazed grass) and management system (grazing vs confinement), with the latter confounded by weather effects. With regards forage type and quality, most research indicates that the inclusion of quality maze silage in the diet of confined cows (included for Confinement and WinterCalf in the current study) will result in improved intakes and milk yields. However, the quality of the conserved forages relative to grazed grass, together with grazing management, is also likely to be a key driver of differences in performance. The ME content of the grazed grass within the current study was on average approximately 0.4 MJ higher than that of the grass silage, while the crude protein content was also substantially higher. In addition, feeding behaviour associated with grazing vs confinement systems have been demonstrated to be very different, cows at pasture grazing for longer than confined cows offered a conserved forage-based diet (Roca-Fernandez *et al*., 2013). The combined impacts of these diverse factors on performance are not always easy to separate and quantify, with few studies having examined performance within confinement and grazing systems at a common concentrate feed level. In the comparisons which do exist, conflicting outcomes have been observed, with authors finding performance to be improved (AbuGhazal *et al*., 2007), unaffected (Purcell *et al*., 2014) or reduced with confinement systems (Mohammed *et al*., 2009). Similar inconsistencies have been observed in studies in which confined cows were offered higher concentrate levels than grazing cows, with authors finding performance to be improved (White *et al*, 2001: Bargo *et al*., 2002: Vahmani *et al*., 2013), unaffected (Kennedy *et al*., 2005: Boken *et al*., 2005) or reduced with confinement systems (Rego *et al*., 2004). While improved performance was observed in the current study, these inconsistent findings across published studies are likely to reflect differences in forage quality, climatic effects and management differences between systems within the different studies. In general (although not always), in studies where food intakes were presented the presence or absence in milk production responses have been mirrored in similar effects on total DM intake.

The high milk constituents within the Holstein cows in the current study reflect the long term focus on milk constituents within the research herd. Nevertheless, there was a general trend for milk fat content to increase with increasing levels of confinement, with this reflecting the increasing proportion of forage fibre in the diet. A similar effect was observed by Purcell *et al*. (2014), who observed a lower milk fat content with grazing cows, than with either fully or partially confined cows, despite similar levels of concentrates being offered with all treatments. Similarly, the lower milk protein levels with the SpringCalf(H) than with the Confinement and WinterCalf systems are likely to reflect the improved energy status of cows on the latter systems, and the higher starch content of the diets offered, a reflection of the higher concentrate levels adopted, and the inclusion of maize silage in the diets. While previous studies have indicated that the inclusion of grazed grass in the diet normally results in improved milk protein concentrations (Ferris *et al*., 2007; Purcell *et al*., 2013), these studies were undertaken at a common concentrate feed level within grazing and confinement systems, while the confinement systems did not involve maize silage forages.

The difference in performance between the Holstein and crossbred cows within the SpringCalf systems, namely a lower milk volume but improved milk composition with the crossbred cows, resulted in no overall effect on milk solids yield. This is in agreement with previous studies comparing these two genotypes within low and moderate concentrate input systems (Prendiville *et al*., 2009; Prendiville *et al*., 2010; Vance *et al*., 2013). The similar outputs of milk solids can be explained by the similar dry matter intakes with each of the two genotypes, which occurred despite the crossbred cows being 62 kg lighter, on average, than the Holstein cows. The similar intake capacity of these two genotypes have been demonstrated in previous studies (Prendiville *et al*., 2009; Vance *et al*., 2012; 2013a, 2013b), with both Prendiville *et al*. (2010) and Vance *et al*. (2013) observing that crossbred cows modified their feeding/grazing behaviour in order to achieve similar intakes as larger Holstein cows.

Body tissue reserves

Cows on Confinement completed the lactation with a higher body condition score than cows on any of the other systems, reflecting the fact that these cows appeared to lose less body condition in early lactation, and began to gain body condition from approximately week 12 of lactation onwards. This trend was also reflected in the live-weight change data with this system, with these cows tending to lose less live weight to nadir, reach nadir live weight earlier (at day 69 post calving), and gain substantially more live weight from nadir to drying off. While part of this trend may be ‘rumen fill’ related, a consequence of the more fibrous nature of the forage component of the diet offered, the condition score data clearly demonstrates that these cows had an improved energy status compared to those on any other treatment. This is supported by the trend for these cows to have higher plasma glucose concentrations, with glucose a key driver of milk production. However, perhaps surprisingly plasma NEFA concentrations remained high with this system throughout the lactation. The lower urea concentrations with this treatment throughout the study reflects the lower protein content of the conserved forages offered, and the lower protein concentrates adopted, the latter made possible by the inclusion of ‘meta-smart’, which supplied methionine, one of the first limiting amino acids for milk production.

Although cows on the WinterCalf system were managed identically to those on the Confinement system in both early and late lactation, the impact of the grazing period is evident from both the live-weight and condition score data. For example, while the live-weight change pattern of cows on this study followed a similar trend to cows on the Confinement system until approximately week 20 post calving, their live weights ‘decreased’ thereafter, reflecting these cows having access to grazing. This decrease is likely to reflect ‘rumen-fill’ effects associated with grazing, condition score data indicating little change after this time. The latter suggests that while these cows may not actually have been in negative energy balance at this time, they were certainly gaining little body tissue. As with the Confinement system, these cows also had a higher plasma NEFA concentration throughout the lactation than cows on either of the SpringCalf systems.

In contrast, the condition score curves for the Holstein and crossbred cows on the SpringCalf systems follow almost identical trends, with cows appearing to lose body condition until late lactation, with a small gain observed thereafter. These trends are almost identical to those observed with these two genotypes in a study involving similar systems (Vance *et al*., 2013), with cows of both genotypes showing little evidence of condition score gain even in late lactation. The live-weight data presented in Figure 1.1 clearly highlights the lower live weights of crossbred cows compared to Holstein cows. Similarly, the live-weight data demonstrates the impact of a predominantly grazing system, with Holstein and crossbred cows reaching nadir live weight at days 145 and 113 of lactation respectively, and gaining live weight thereafter, until the end of lactation. The lower plasma glucose concentrations observed with these cows in early lactation is reflected in lower milk yields at this time, a reflection of their lower concentrate feed levels prior to turnout.

Fertility performance and cow health

While this study involved twenty cows per system each year over a three-year period, this is inadequate to robustly assess the effects of system on fertility or health performance. Nevertheless, a number of observations can be made.

In agreement with findings of previous studies (Auldist *et al*., 2007; Prendiville *et al*., 2009; Vance *et al*., 2013), there was a general trend for crossbred cows within the SpringCalf system to have improved fertility compared to Holstein cow within this system, or indeed within any system. The improved fertility performance of Jersey crossbred cows is normally attributed to hybrid vigour (Lopez-Villalobos, 1998). In contrast, within the systems involving Holstein cows, there was a general trend for poorer fertility with the Confinement system than the SpringCalf(H) system, the exception being conception at the end of the breeding season which tended to be higher with the Confinement system. With regards the latter, cows on the Confinement and WinterCalf systems had a 6-7 month breeding season, compared to a 14-week breeding season for cows on the SpringCalf system. In a recent review, Mee (2012) concluded that compared to confined cows, cows on pasture systems had increased luteal and oestrous activity, and lower early embryonic mortality, although evidence of the impact of management system on conception metrics was conflicting.

There were clear trends (although not significant for SpringCalf(H) vs WinterCalf), for Holstein cows on Confinement and WinterCalf to have increased incidence of mastitis than Holstein cow on the SpringCalf system. Previous studies have found an increased incidence of mastitis with confined cows, compared to grazing cows (White *et al*., 2002; Washburn *et al*., 2002). In addition, as was observed in the current study, higher somatic cell counts have also been observed with predominantly housed cows compared to grazing cows, although this has not been universal observed. Clearly, cleanliness of cubicles and bedding vs cleanliness of pasture and grazing conditions, can all impact on the cell counts of cows within these different systems. The trend towards a lower incidence of mastitis with the crossbred cows compared to the Holstein cows (significantly lower than for Confinement and WinterCalf), despite no difference in somatic cell score within these systems, is again in agreement with the findings of Vance *et al*. (2013). Hybrid vigour has been demonstrated to result in lower incidence of mastitis, while having little effect on somatic cell counts.

The non-significant trend for higher levels of lameness in cows within the Confinement and WinterCalf systems, compared to the SpringCalf systems, is likely to reflect the increased duration of the housing periods and higher concentrate feed levels with these cows. This agrees with findings of a number of studies (Olmos *et al*., 2009) and observational studies, that the incidence of hoof health issues increase with increased duration of housing. While the exposure of the hoof to slurry within a free stall situation is likely to be a contributing factor to the incidence if hoof health problems, the hardness of the standing surface, increased lying times, and exercise are all likely to be additional contributing factors. The results of the study also suggest improved hoof health with crossbred cows compared to Holstein cows on the SpringCalf system, with these results in agreement with the trends observed by Vance *et al*. (2012). A number of studies have compared the hoof health of Jersey cows with a second breed, and have suggested the former have improved hoof health (Alban, 1995; Huang *et al*., 1995), with this likely due to Jersey cows having harder hooves.

Stocking rates

While actual stocking rates were measured within the grazing components of WinterCalf and the two SpringCalf systems (5.1 and 4.3 cows per ha, respectively; mean across the three years of the study), the grass silages and maize silages offered within the study were not produced specifically for this study, and as such, yield data were not available. In order to allow overall stocking rates to be calculated, actual intakes of grass silage and maize silage within each of the systems were used, while a yield of 13.0 t DM/ha was adopted for grass harvested for grass silage production (mean yield from small scale replicated silage plots associated with each of the systems: Section 2 of this report) and a yield of 10.0 t DM/ha was assumed for maize (typical yield of maize silage in NI). In addition, in-silo losses and feed out losses of 13.4% (Mayne and Gordon, 1986) and 4.0%, respectively, were adopted. On this basis, whole system stocking rates were calculated as 2.67, 2.38, 2.41 and 2.45 for Confinement, WinterCalf, SpringCalf(H) and SpringCalf(J), respectively. Thus, all systems were highly stocked, and would require a derogation under the EU Nitrates Directive.

For subsequent calculations (P balance, economic performance and GHG emissions), a ‘whole farm’ stocking rate (including young stock) is required. Stocking rates for young stock were calculated assuming a replacement rate of 30% for each of the three systems involving Holstein cows, and 25% for the system involving crossbred cows, with this based on the outcomes of an on-farm comparison of Holstein and Jersey crossbred dairy cows (Ferris *et al*., 2012). This calculation took account of still births (assumed as 12% and 9% for Holstein and Jersey crossbred heifers at first calving, and 3% for all cows at subsequent calvings: Ferris *et al*., 2012)), and a 14.5% ‘loss’ of young stock between birth and calving at two years of age (Wathes *et al*., 2008). All male calves, and female calves which were surplus to breeding requirements, were assumed sold at birth. Young stock ‘cow equivalents’ (ce) were assumed as 0.4 ce for animals between birth and 12 months of age, and 0.6 ce for animals between 13 and 24 months of age, while a stocking rate of 2.23 ce/ha was assumed (DARD Farm Business Survey Data, 2013). Using these assumptions, whole farm stocking rates (milking and young stock) were determined as 2.53, 2.33, 2.35 and 2.38 ce/ha for systems Confinement, WinterCalf, SpringCalf(H) and SpringCalf(J), respectively.

Phosphorus balance

One of the objectives of this experiment was to minimise P surpluses within each of the four systems. This was achieved through the adoption of low phosphorus concentrates and by not applying inorganic P fertilisers. The latter was justified in that the soil status of all grassland areas within the study were $\geq$ index 3. To examine the effect of system on P balance, a ‘whole farm’ P balance was calculated for each of the systems, these calculations being undertaken for a 100-cow dairy herd, plus young stock. Whole farm stocking rates for the ‘milking herd’ and young-stock were as described above.

Phosphorus inputs from inorganic fertiliser were assumed as zero across all land areas. Concentrate inputs to young stock were assumed as 395 kg/year for heifers associated with the Confinement and WinterCalf systems, and 385 kg/year for heifers associated with the SpringCalf systems (DARD Farm Business Survey Data, 2013). The P content of this concentrate was assumed as 4.25 g/kg fresh, based on the content of commercial concentrates being offered at AFBI Hillsborough. Concentrate inputs for the dairy enterprise were based on actual concentrate intakes recorded during the study, adjusted to an annual basis, and the actual P content of these concentrates. Straw was assumed to have a P content of 1.0 g/kg fresh, while usage was assumed as 150 kg/heifer during the rearing period (Farm Business Survey Data, 2013), and 30 kg/cow/year.

Phosphorus exports in milk were based on milk outputs, adjusted to an annual basis, and the actual P content of milk produced. Phosphorus exports in cull cows were based on the replacement rates defined previously, and the assumed P content of a cull cow (3.96 kg) (NI Nitrates Directive Guidance Booklet, 2010). Phosphorus exports in young stock were determined by assuming all surplus young stock were removed from the farm at birth, with these animals having an assumed P content of 0.33 kg each (NI Nitrates Directive Guidance Booklet, 2010).

Based on these values, P balances were calculated as 5.4, 0.6, -5.7 and -5.1 kg P per ha for Confinement, WinterCalf, SpringCalf(H) and SpringCalf(J), respectively (Table 1.12). While this P balance is likely sustainable for the Confinement system, provided slurry P is distributed across the entire grassland area of the farm, it is unlikely to be sustainable for the WinterCalf system, and is clearly unsustainable for the SpringCalf systems. However, if a P application of 20 kg P_2O_5/ha (8.8 kg P) is assumed for all grassland within each system, the P balance with each system increases to 12.8, 8.7, 3.1 and 3.7 kg/ha, for Confinement, WinterCalf, SpringCalf(H) and SpringCalf(J), respectively (Table 1.13). In this scenario, the P surplus with the Confinement system exceeds the 10.0 kg surplus currently allowed for ‘derogated farms’ under the NI Derogation. The relatively small surpluses with the SpringCalf systems suggest that these systems are still unlikely to be sustainable at this P application level (8.8 kg P/ha). Thus it is clear that with both moderate and low input grassland-based systems, a certain amount of inorganic P will be required to maintain the sustainability of these systems. Increasing the P application on all grassland within each system to 40 kg P_2O_5/ha (17.6 kg P) increases the P balance to 20.3, 16.8. 11.9 and 12.5 kg P/ha, for Confinement, WinterCalf, SpringCalf(H) and SpringCalf(J), respectively. At this level of P application, none of the systems would be able to meet the 10.0 kg P balance permissible for derogated dairy farms in Northern Ireland. This emphasises the need for farmers, especially those operating lower concentrate input systems, to undertake soil tests on a regular basis, and to apply inorganic P when required, so as to maintain soil indexes at the agronomic optimum.

Carbon footprint of the three systems

There is increasing evidence that changes in global climate patterns are linked to emissions of greenhouse gases, with agriculture an important source of these gases. Within Northern Ireland approximately 27% of total greenhouse gas emissions are from agriculture, compared to only 7% within the UK as a whole. There are a number of reasons why this is important. Firstly, governments are setting targets by which greenhouse gases emissions should be reduced, and there is increasing pressure to meet these targets. Secondly, supermarkets are increasingly interested in being able to demonstrate that the produce they sell has a low carbon footprint, and in the future may seek to source milk from farmers who are able to demonstrate that their production systems have a low carbon-footprint.

The main greenhouse gases emitted from agriculture are carbon dioxide, methane and nitrous oxide. Carbon dioxide is produced from the burning of fossil fuels (ie diesel in tractors, electricity production, fertiliser manufacture), methane is produced by ruminant livestock and from slurry during storage, while nitrous oxide is produced by bacteria within the soil and within slurry stores. Methane and nitrous oxide are much more 'potent' as greenhouse gases than carbon dioxide, and have a 'global warming potential' that is approximately 25 and 298 times greater (respectively) than carbon dioxide.

In order to examine GHG emissions from each of the systems, data collected were inputted into the AFBI Dairy Systems GHG calculator. As with previous calculations in this report, all information was 'scaled up' to simulate a farming system comprising a herd of 100 dairy cows, and associated young stock, as described previously.

The different qualities of diet offered within each of the systems will have resulted in different methane emissions per kg of feed intake. For example, Yan *et al*. (2000) demonstrated that as forage proportion in the diet of cattle decreases, methane production as a proportion of energy intake decreases. It is also recognised that total feed intake is the major factor determining enteric CH_4 emissions in cattle (Yan *et al*., 2000; Mills *et al*., 2003; Ellis *et al*., 2007). In the present study, total DM intake was highest with the Confinement system and lowest with the SpringCalf systems, suggesting that the higher intake was the primary driver of the higher total enteric CH_4 emissions observed with Confinement. An even larger difference in enteric methane emissions between the systems might have been expected in view of the sizeable difference in milk yield. However, the impact of this difference was 'diluted' by enteric emissions associated with heifer rearing and dairy cow 'maintenance', with emissions associated with these components largely equal across systems. The exception to this was SpringCalf(Jx) for which emissions associated with heifer rearing and maintenance were lower. Nevertheless, these results clearly demonstrate that even with diverse milk production systems, enteric methane emissions represent the predominant source of total emissions. This is in agreement with other grassland-based dairy production systems where the main source of total GHG emissions was also enteric fermentation (Schils *et al*., 2005; Gibbons *et al*., 2006; O'Brien *et al*., 2010). As a result of the higher emissions associated with concentrate production and manufacture with Confinement and WinterCalf, the relative contribution of enteric methane emissions to total GHG emissions was lower with these systems (42 and 43%, respectively), than with the SpringCalf systems (average, 46.5%). Previous studies examining enteric emissions from dairy systems also reported lower contributions with confinement systems than with grazing systems (Nagel *et al*., 2003; O'Brien *et al*., 2012a).

The higher total emissions from manure management with Confinement and WinterCalf were, like enteric emissions, largely driven by the higher DM (and OM) intakes associated with these system, while differences in diet quality and the N content of the diet will have had a lesser effect. However, as a percentage of total emissions, the relative contribution of manure management was relatively similar for all systems. This is in agreement with the observations of O'Brien *et al*. (2012a) and predictions by Bell *et al*. (2011), with the former attributing emissions of 14% and 15% to manure management with a grazing and confinement system, respectively while the latter calculated that manure and soils were responsible for 32% of total emissions within both a grazing and confinement system.

The higher emissions associated with fertiliser production and application with the SpringCalf systems, reflects the much smaller contribution of concentrates (and consequently greater reliance on forage) with these systems, and the fact that the Confinement and WinterCalf systems involved maize, which received only small quantities of inorganic nitrogen. These results are in agreement with the findings of others dairy systems where greater contributions were also calculated for a grazing system than for a confinement system (for example 16% and 9% respectively in Bell *et al*. (2011) and 23% and 5% in O'Brien *et al*. (2012a)).

The experimental farm did not have facilities to record fuel and electricity use associated with each of the management systems, and default values within the calculator were thus adopted. These values relate emissions associated with fuel and electricity to the volume of milk produced, thus accounting for the marginally higher total emissions obtained with Confinement and WinterCalf, than with the SpringCalf systems. However, it is likely that this methodology does not fully account for the large management differences between systems within the current study. For example, with Confinement all forage offered throughout the lactation were harvested, and then offered on a daily basis, while all slurry produced during the lactation was removed and spread. In addition, there is likely to be increased electricity usage (scraper systems/lighting) associated with a fully confined system. While the contributions of the emissions arising from fuel/electricity use with SpringCalf (3%) were similar to those (4.5%) presented by O'Brien *et al*. (2012a) for a pasture-based system in Ireland, the contributions from fuel/electricity use with Confinement (also 3%) were lower than those reported by O'Brien *et al*. (2012a) from a confinement system (9%), suggesting a likely underestimation of the emissions from this source in the current study. The calculator developed in the present study does however provide the option to calculate emissions from actual fuel and electricity consumptions when these are available, thus allowing this process to be accounted for with improved accuracy.

The higher total emissions/cow with Confinement and WinterCalf compared to SpringCalf reflects the higher milk production and associated higher intakes with the former systems. This agrees with previous LCA studies where it was shown that intensification of milk production systems causes GHG emissions per hectare of land to increase (Basset-Mens *et al*., 2009).

In contrast to total GHG emissions, emissions per kg of ECM produced were relatively similar across the four systems, although marginally higher for the SpringCalf system involving Holstein cows. In contrast, Bell *et al*. (2011) predicted emissions of 1.12 and 1.32 kg CO_2e/kg ECM for a Confinement and grazing system respectively, and similarly, Lovett *et al*. (2006) predicted an increase in GHG emissions with decreasing use of concentrate supplementation for spring calving milk production systems typical of those practiced in Ireland. In contrast, O'Brien *et al*. (2012a) found that emissions from a grazing system were lower than those from a confinement system (0.87 and 1.03 kg CO_2e/kg fat and protein corrected milk, respectively). Reasons for these inconsistencies between systems may be due to differences in inputs and performance between systems in different studies, as well as differences between calculators in the assumptions used. It is indeed recognised that direct comparison of carbon footprints obtained from different GHG calculators is difficult due to variations in the methodologies and assumptions used within individual calculators (IDF, 2010).

Across both systems, taking account of carbon sequestration reduced GHG emissions by 14%. The greater reduction in GHG emissions per kg of ECM produced with the SpringCalf systems, compared to the Confinement system reflects the greater grassland area associated with the SpringCalf systems, and the corresponding increased potential for sequestration to take place. These results agree well with previous studies, where the carbon footprint was reduced to a greater extent on pasture-based systems than on confinement systems when carbon sequestration was included (Schonbach *et al*., 2012; O'Brien *et al*., 2012a). The sensitivity of the carbon footprint to carbon sequestration within this and previous studies (Dollé *et al*., 2011; O'Brien *et al*., 2012a) demonstrates the importance of developing GHG calculators that offer the option to include or exclude this process when calculating the carbon footprint of dairy systems. It also highlights the need for a better standardisation of approaches when taking carbon sequestration into account.

Within the SpringCalf systems there was a tendency for the system involving crossbred cows to have lower emissions than the system involving Holstein cows. As energy corrected milk yield was similar with each of the two systems, this difference is likely a combination of the lower maintenance energy requirements of the smaller crossbred cows, combined with their assumed lower replacement rate. Results presented in Table 1.14 highlight that these higher emissions were primarily due to increased emissions from enteric fermentation and manure management. Our results are in close agreement with the outputs of a whole farm simulation model which indicated that a reduction in replacement rate of 5 to 9% on intensive dairy farms in The Netherlands (5% in the current study) would reduce the carbon footprint of milk production by 2 to 4% (Vellinga *et al*., 2011). While the assumed difference in replacement rates between genotypes within the current study was relatively small, modelling work on UK and other European dairy farms predicted that reducing replacement rates by approximately 25% would reduce methane emissions by 4-5% per cow (Garnsworthy, 2004) and GHG emissions (per kg of ECM) by 5-7% (Weiske *et al*., 2006).

Financial performance of the four systems

The financial performance of the four systems has been compared in Table 1.15 for a 100-cow herd, plus young stock. Lactation milk yields were adjusted to an annual milk output basis. The analysis was initially undertaken at a milk price of 32 pence per litre, with milk price adjusted for compositional bonuses. Differences between breeds in replacement rates, stillbirth rates, calves sold, and cull cows sold have been included within the calculations, based on the assumptions highlighted earlier. The values of the Holstein calves sold were assumed as £100 (bull) and £150 (heifer), while the value of Jersey crossbred calves sold were assumed as £50 (bull) and £150 (heifer). Holstein cull cows were assumed to have a value of £600, while crossbred cull cows were assumed to have a value of £470 (based on actual values of cows of these two breeds sold from the AFBI-Hillsborough herd over a two-year period). The value of replacement heifers was assumed to be the same for both breeds (£1300, CAFRE benchmarking). Feed costs for the 'milking herd' were based on actual feed inputs (adjusted from a lactation basis to an annual basis) measured within the study, with costs for grass silage, maize silage and grazed grass assumed as £105, £115, £60/t DM, respectively (CAFRE Forage costs, updated 2013), and the cost of all concentrates assumed as £275/t fresh. Sundry costs were based on Farm Business Survey Data (2013). Veterinary/medicine and AI costs were assumed to be 20%

lower with the crossbred cows due to their improved health and fertility, with total annual sundry costs assumed as £145/cow and £121/cow for Holstein and Jersey crossbred cows, respectively.

Annual gross margins were calculated per cow, per ha and per litre of milk produced for each of Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx), with these £1368, £1411, £1218 and £1235 (gross margin per cow), £3655, £3363, £2941 and £3020 (gross margin per ha), and 15.5 pence, 17.3 pence, 19.1 pence and 20.7 pence (gross margin per litre), respectively.

As Hillsborough is operated as a research farm, it is not possible to obtain 'representative' fixed costs for the systems examined. Thus, in order to examine the net margin of each of the systems, fixed costs were obtained from CAFRE Benchmarking. During 2012 and 2013, herds participating in Benchmarking were categorised, with three of the categories adopted ('Fully Confined', 'Conventional' and 'Spring Calving'), corresponding to the Confinement, WinterCalf and SpringCalf systems, respectively. Within the Benchmarking data sets, data were excluded for herds with Automatic Milking Systems, while for each of Fully Confined and Conventional sub-groups, data were excluded for herds with milk yields of less than 7500 litres/cow/year. This left 11, 44 and 8 farms within the Fully Confined, Conventional and Spring Calving categories respectively in 2012 and 8, 50 and 19 farms within each of these three categories during 2013. Across all farms within each of these three categories, mean fixed costs per litre of milk produced were calculated over the two year period (7.5, 6.6 and 8.3, for Fully Confined, Conventional and Spring calving, respectively), with these then expressed on a per cow basis using milk yields within benchmarking (£619, £536, £481 for Fully Confined, Conventional and Spring calving, respectively). These fixed costs included machinery depreciation and running costs, contractor costs, building depreciation, property charges, paid labour, conacre and finance and miscellaneous charges. Net margin values for each of the four experimental systems were obtained by deducting fixed costs obtained from benchmarking from the gross margin values, described above.

The overall outcome of the economic analysis (milk at 32 pence per litre, concentrates at £275/t) was that net margin per cow was maximised with the WinterCalf system

(£875/cow), while being relatively similar with each of the other three systems (£749, £737 and £754 for Confinement, SpringCalf(H) and SpringCalf(Jx), respectively (Table 1.15). Similarly, net margin per ha was maximised with the Conventional system (£2086/ha), intermediate with the Confinement system (£2001/ha), and lowest with the Spring calving systems (£1780 and £1843/ha with SpringCalf(H) and SpringCalf(Jx), respectively. Net margin per litre increased from the Confinement system (8.5 pence) through to the SpringCalf(Jx) system (12.7 pence).

This analysis was repeated at a milk price of 27 and 22 pence per litre (concentrate cost, £275/t), with the outcomes of this presented in Figure 1.8. The analysis was further repeated across these same milk prices at a concentrate cost of £225/t (Figure 1.9). Consistent across both concentrate cost scenarios is the dramatic reduction in net margin with falling milk price, with this reduction being greatest with the systems involving higher levels of milk production. Similarly, the increase in net margin associated with a reduction in concentrate cost was greatest with systems involving higher concentrate inputs.

The relative profitability of each of the different systems varied as milk price and concentrate cost changed. In general, the WinterCalf system was most profitable at milk prices of 27 ppl and greater. At a milk price of 22 pence per litre the SpringCalving systems tended to be more profitable, especially at higher concentrate costs. The impact of a low milk price and high concentrate costs is particularly evident with the Confinement systems, with this system experiencing a negative net margin of -£133/cow/year in this scenario.

It is now widely recognised that volatility in milk prices and input costs are likely to remain a permanent feature of dairy farming for the foreseeable future, and as such, optimum systems are those that are likely to be robust and resilient over a wide range of milk price/concentrate cost scenarios. In general, across the range of scenarios examined, the WinterCalf system tended to have the highest net margin. This finding supports the modelling work of Anderson *et al*. (2010) which indicated that a moderate input-moderate output autumn calving system (approximately 8000 litres/cow/year) is one of the most robust systems for Northern Ireland. Next most profitable were the spring calving systems, followed by the Confinement system, reflecting the susceptibility of the system to low milk prices and high concentrate costs.

Nevertheless, it must be recognised that within any milk price-concentrate cost scenario, there was a ‘relatively small’ difference in net margin across the four systems, when compared to actual differences in net margin within similar systems in practice. It is suggested that all four systems operated at relative high levels of efficiency, when defined as milk produced per kg of concentrate offered. This is demonstrated in Figure 1.10, which presents the milk yield response to concentrates on Northern Ireland dairy farms (CAFRE Benchmarking data, 2012), with performance of each of the four systems (annual basis) superimposed on this Figure (all milk yields corrected to a common milk energy content). This figure clearly demonstrates that in terms of efficiency (milk output/kg concentrate offered), the Hillsborough systems lie along the upper end of the data set, indicating that these systems operated with similar levels of efficiency as the top Northern Ireland dairy farms operating similar systems. The relatively narrow range of net margins across the four systems highlight that a range of systems can operate with high net margins in Northern Ireland, provided high levels of technical efficiency are achieved within each systems, with this in agreement with the findings of CAFRE benchmarking over many years. It must also be noted that the ranking in net margin within the current study is impacted by many factors, and the margins can be relatively sensitive to changes in the assumptions made. Thus individual farmers, under different circumstances, and with different efficiencies from those within the current study, may have very different net margins from those determined.

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Table 1.1 Summary of key aspects of each of the four systems examined

	Confinement	WinterCalf	SpringCalf (H)	SpringCalf (Jx)
Calving period	October – April	October – April	January – April	January – April
Genotype	Holstein-Friesian	Holstein-Friesian	Holstein-Friesian	Jersey x Holstein-Friesian crossbred
Winter management	Total confinement system. From calving until 160 days post calving, diet contained 35% forage and 65% concentrate (DM basis). Thereafter, diet contained 40% concentrate and 60% forage (DM basis)	Confined. From calving until turnout diet contained 35% forage and 65% concentrate (DM basis)	Confined. From calving until turnout diet contained 70% forage and 30% concentrate (DM basis)	Confined. From calving until turnout diet contained 70% forage and 30% concentrate (DM basis)
Grazing management		Rotational grazing system. Offered 5.0 kg concentrate/cow/day	Rotational grazing. Offering 1.0 kg concentrate/cow/day	Rotational grazing. Offered 1.0 kg concentrate/cow/day
Late lactation management		Confined. Offered diet containing 40% concentrate and 60% forage (DM basis)	Confined. Offered grass silage plus 3.0 kg concentrate/cow/day	Confined. Offered grass silage plus 3.0 kg concentrate/cow/day

Table 1.2 Summary of key dates within the study

		Confinement	WinterCalf	SpringCalf (H)	SpringCalf (Jx)
Year 1	Calving date	18/11/2008	21/11/2008	04/02/2009	08/02/2009
	Start part time grazing	NA	29/03/2009	09/03/2009	09/03/2009
	Start full time grazing	NA	16/04/2009	02/04/2009	02/04/2009
	Start part time re-housing	NA	NA	19/10/2009	19/10/2009
	Start full time re-housing	NA	19/10/2009	23/10/2009	23/10/2009
	Dry off date	08/10/2009	08/10/2009	11/11/2009	01/12/2009
Year 2	Calving date	14/12/2009	12/12/2009	01/02/2010	07/02/2010
	Start part time grazing	NA	27/03/2010	09/02/2010	09/02/1020
	Start full time grazing	NA	19/04/2010	21/04/2010	21/04/2010
	Start part time re-housing	NA	02/10/2010	22/10/2010	22/10/2010
	Start full time re-housing	NA	01/11/2010	01/11/2010	01/11/2010
	Dry off date	06/11/2010	07/10/2010	10/12/2010	12/12/2010
Year 3	Calving date	24/12/2010	07/12/2010	20/02/2011	11/02/2011
	Start part time grazing	NA	29/03/2011	26/02/2011	26/02/2011
	Start full time grazing	NA	14/04/2011	14/04/2011	14/04/2011
	Start part time re-housing	NA	09/10/2011	09./10/2011	09/10/2011
	Start full time re-housing	NA	25/10/2011	09/11/2011	09/11/2011
	Dry off date	14/10/2011	15/10/2011	09/12/2011	04/12/2011

Table 1.3 Ingredient composition of concentrate feedstuffs offered (kg/t)

	Confinement concentrates									Grazing concentrates		
	Confinement and WinterCalf						SpringCalf(H) and SpringCalf(Jx)					
	Year 1		Year 2		Year 3		Year 1	Year 2	Year 3	Year 1	Year 2	Year 3
	Pre day-180	Post day-180	Pre day -180	Post day-180	Pre day-180	Post day-180						
Barley	160	160	175	160	175	175	150	150	150	190	190	190
Wheat												
Maize meal	160	175	175	175	175	175	150	150	150	190	190	190
Sugar beet pulp	155	155†					150			170†		
Citrus pulp	155	170	310	325	155	155	150	295	295	170	340	340
Soya hulls					155	155						
Soya bean	155	140	140	140	140	140	260	260	260	170	170	170
Rape meal	155	140	140	140	140	140	80	80	80	40	40	40
Megalac	20	20	20	20	20	20	20	20	20			
Metasmart	2.5		2		2.5							
No phos minerals	14	14	15	14	15	15	20	25	25	30	30	30
Salt	4	4	4	4	4	4	5	5	5			
Di calcium phosphate												
Limestone												
Calcined Magnesite	4.5	4.5	4	4.5	4	4	5	5	5	10	10	10
Molaferm	15	17.5	15	17.5	15	17	10	10	10	30	30	30
Water												

† Part way through Year 1 (June 2009) sugar-beet pulp was removed from the post day 180 concentrate and from the grazing concentrate, and was replaced by additional citrus pulp

Table 1.4 Mean chemical composition of grass silage and maize silage offered during the three years of the experiment (g/kg DM, unless stated otherwise)

	Confinement	**s.d.**	**WinterCalf**	**s.d.**	**SpringCalf(H) and SpringCalf(Jx)**	**s.d.**
Grass silage						
Volatile corrected DM (g/kg)	265	60.1	269	59.6	280	60.3
Crude protein	149	25.2	146	27.5	142	24.5
Ammonia N (g/kg total N)	78	33.8	75	37.8	76	35.3
pH	3.80	0.287	3.79	0.263	3.79	0.264
Lactate	121	48.6	129	46.1	122	45.2
Acetate	17.7	10.71	16.6	10.31	15.1	7.87
Acid detergent fibre	285	26.2	285	24.8	282	25.2
Neutral detergent fibre	489	42.9	490	43.5	484	46.4
Ash	87	11.2	86	12.1	88	13.7
Gross energy (MJ/kg DM)	19.2	1.61	19.2	1.67	19.3	1.33
Phosphorus	2.88	0.514	2.98	0.533	2.98	0.473
Metabolisable energy (MJ/kg DM)†	11.3	0.68	11.4	0.70	11.2	0.72
Maize silage						
Volatile corrected DM (g/kg)	303	43.0	327	36.0		
Crude protein	81	10.1	79	7.6		
Ammonia N (g/kg total N)	94	22.8	91	19		
pH	3.68	0.218	3.65	0.146		
Lactate	52	26.8	52	22.4		
Acetate	29	13.6	26	11.2		
Acid detergent fibre	236	32.9	229	36.6		
Neutral detergent fibre	454	52.9	441	57.7		
Ash	35	3.3	33	3.2		
Gross energy (MJ/kg DM)	19.3	1.3	19.7	1.21		
Starch	266	60.9	269	62.7		
Phosphorus	2.25	0.285	2.29	0.300		
Metabolisable energy (MJ/kg DM)†	11.1	0.73	11.4	0.55		

† Predicted using NIRS

Table 1.5 Mean chemical composition of grazed grass offered during the three years of the experiment (g/kg DM, unless stated otherwise)

	WinterCalf	**s.d.**	**SpringCalf (H)**	**s.d.**	**SpringCalf (Jx)**	**s.d.**
Dry matter (g/kg)	178	0.3	179	0.3	179	0.3
Crude protein	225	52.2	233	89.1	223	52.0
Acid detergent fibre	219	29.7	211	20.0	21.3	19.9
Neutral detergent fibre	467	62.8	441	38.7	456	35.9
Ash	91	9.9	89	12.8	91	21.4
Water soluble carbohydrate	147	44.9	148	47.7	155	46.7
Gross energy (MJ/kg DM)	18.7	0.48	18.8	0.40	18.7	0.58
Phosphorus	3.3	0.52	3.3	0.53	3.3	0.48
Metabolisable energy (MJ/kg DM)[†]	11.7	0.38	11.7	0.41	11.7	0.41

[†] Predicted using NIRS

Table 1.6 Mean chemical composition (g/kg DM, unless stated otherwise) of concentrate feedstuffs offered during the three years of the experiment (standard deviation in brackets)

	Confinement concentrates						**Grazing concentrates**	
	Confinement and WinterCalf				**SpringCalf(H) and SpringCalf(Jx)**			
	Pre day 180		**Post day 180**					
Crude protein	185	(17.4)	184	(14.0)	211	(18.4)	189	(24.1)
Acid detergent fibre	129	(29.1)	125	(17.8)	120	(21.7)	122	(27.4)
Neutral detergent fibre	238	(40.5)	220	(21.7)	226	(43.8)	218	(47.8)
Ash	79	(8.9)	74	(7.5)	80	(9.20)	78	(10.8)
Gross energy (MJ/kg DM)	18.0	(0.27)	18.1	(0.21)	18.2	(0.18)	18.7	(3.0)
Phosphorus (g/kg DM)	3.8	(0.46)	3.7	(0.32)	4.2	(0.39)	3.9	(0.35)

Table 1.7 Pre and post grazing grass heights, total fertiliser nitrogen inputs and grazing stocking rates within systems where cows grazed

		WinterCalf	SpringCalf (H)	SpringCalf (Jx)
Year 1	Pre grazing sward height (cm)	8.7	8.6	8.7
	Post grazing sward height (cm)	5.0	4.8	4.8
	Fertiliser nitrogen applied to grazing areas (kg N/ha)	233	230	230
	Grazing stocking rate (cows/ha)	5.09	4.38	4.38
Year 2	Pre grazing sward height (cm)	8.7	8.9	8.5
	Post grazing sward height (cm)	5.3	5.3	5.2
	Fertiliser nitrogen applied to grazing areas (kg N/ha)	240	244	244
	Grazing stocking rate (cows/ha)	4.98	4.26	4.26
Year 3	Pre grazing sward height (cm)	8.8	9.1	8.7
	Post grazing sward height (cm)	5.0	5.0	4.7
	Fertiliser nitrogen applied to grazing areas (kg N/ha)	235	233	233
	Grazing stocking rate (cows/ha)	5.19	4.27	4.27

Table 1.8 Effect of system on days on study and total dry matter intakes (kg/cow) during the lactation and dry periods

	Confinement	WinterCalf	SpringCalf (H)	SpringCalf (Jx)	SED	P-value
Days on study						
Lactation period	326^{b}	317^{b}	303^{a}	302^{a}	6.7	<0.001
Dry period	70	71	80	77	4.7	0.110
Lactation period intakes (kg DM/cow)						
Silage	2527^{c}	1159^{ab}	1053^{a}	1066^{a}	60.9	<0.001
Maize	672^{b}	397^{a}	0	0	27.1†	<0.001†
Concentrate	3080^{c}	2175^{b}	722^{a}	760^{a}	55.1	<0.001
Grass	0	2041^{a}	2788^{b}	2692^{b}	81.2 *	<0.001*
Total	6362^{c}	5763^{b}	4563^{a}	4473^{a}	136.6	<0.001
Dry period intakes (kg DM/cow)						
Total	723^{a}	739^{a}	878^{b}	812^{ab}	51.1	0.010

† Comparison between Confinement and WinterCalf only
* Comparison between WinterCalf, SpringCalf(H) and SpringCalf(Jx) only
Means with the same superscripts within rows are not significantly different (P>0.05)

Table 1.9 Full lactation milk production performance associated with each of the four systems

	Confinement	WinterCalf	SpringCalf(H)	SpringCalf(Jx)	SED	P-value
Milk yield (kg/lactation)	9333[c]	8443[b]	6464[a]	6049[a]	240.6	<0.001
Milk composition (g/kg)						
Fat	44.9[b]	43.3[ab]	42.8[a]	49.0[c]	1.02	<0.001
Protein	34.6[b]	34.9[b]	33.6[a]	36.3[c]	0.48	<0.001
Lactose	46.0[b]	45.5[a]	45.1[a]	45.3[a]	0.25	<0.001
Milk solids yield (kg/lactation)						
Fat	419[c]	365[b]	277[a]	294[a]	11.3	<0.001
Protein	323[c]	295[b]	218[a]	220[a]	7.6	<0.001
Lactose	430[c]	384[b]	291[a]	274[a]	11.3	<0.001
Fat plus protein	741[c]	660[b]	495[a]	514[a]	18.3	<0.001
Energy corrected milk yield (kg/lactation)	9934[c]	8817[b]	6640[a]	6775[a]	242.5	<0.001
Phosphorus content (mg/litre)	924	915	NA	938	15.7	0.348
Somatic cell count (x 1000/ml)	222	209	114	183		
Somatic cell score (x 1000/ml $\log_e$)	11.76[b]	11.82[b]	11.34[a]	11.70[b]	0.176	0.031

NA - not analysed
Means with the same superscripts within rows are not significantly different (P>0.05)

Table 1.10 Effect of milk production system on body tissue reserves

	Confinement	WinterCalf	SpringCalf(H)	SpringCalf(Jx)	SED	P-value
Live weight (kg)						
Mean	602[c]	581[c]	540[b]	478[a]	11.0	<0.001
At calving	588[b]	590[b]	570[b]	506[a]	12.1	<0.001
At drying off	662[d]	627[c]	591[b]	524[a]	13.5	<0.001
Nadir	542[c]	534[c]	502[b]	444[a]	10.5	<0.001
Loss to nadir	48[a]	56[ab]	68[b]	61[ab]	6.7	0.023
Days to nadir	69[a]	130[bc]	145[c]	113[b]	14.7	<0.001
Gain from nadir to drying off	125[a]	94[a]	92[a]	80[a]	8.5	<0.001
Condition score						
Mean	2.55[b]	2.41[a]	2.38[a]	2.38[a]	0.048	<0.001
At calving	2.62	2.58	2.60	2.64	0.046	0.577
At drying off	2.70[b]	2.40[a]	2.31[a]	2.37[a]	0.066	<0.001

Means with the same superscripts within rows are not significantly different (P>0.05)

Table 1.11 Effect of milk production system on fertility performance and cow health

	Confinement	WinterCalf	SpringCalf(H)	SpringCalf(Jx)	SED	P-value
Fertility performance (proportional basis unless stated otherwise)						
Days to first observed heat	54	52	48	40	6.1	0.099
Days to first serve	65	89	74	70	9.9	0.066
Conception to first service (proportion)	0.24	0.27	0.36	0.38	0.094	0.419
Conception to first and second service (proportion)	0.41	0.45	0.53	0.68	0.094	0.114
Interval from calving to conception (days)	116^{b}	108^{ab}	98^{a}	90^{a}	9.4	0.032
Pregnancy rate at end of breeding season (proportion)	0.81	0.73	0.69	0.83	0.081	0.673
Calving interval (days)	397	390	382	376	10.0	0.147
Health parameters						
Proportion of cows with one or more cases of mastitis	0.42^{c}	0.41^{bc}	0.24^{ab}	0.13^{a}	0.083	0.007
Proportion of cows with one or more cases of lameness	0.20	0.28	0.11	0.05	0.120	0.120
Mean locomotion score	2.76^{c}	2.65^{b}	2. 61^{ab}	2.54^{a}	0.0511	<0.001

Means with the same superscripts within rows are not significantly different (P>0.05)

Table 1.12 Effect of system on the phosphorus balance for a 100 cow dairy herd, plus replacements

	System			
	Confinement	Conventional	SpringCalf (H)	SpringCalf (Jx)
Total P inputs (kg)				
Fertiliser	0	0	0	0
Concentrates	1247	931	406	404
Straw	13	13	13	11
Sum	**1260**	**944**	**419**	**415**
Total P outputs (kg)				
Milk	839	769	601	576
Culls	119	119	119	88
Calves	20	20	20	22
Sum	**978**	**908**	**740**	**685**
Total P surplus on farm (kg)	282	36	-321	-270
P surplus/ha	5.4	0.6	-5.7	-5.1

Table 1.13 Effect of system on the phosphorus balance for a 100 cow dairy herd, plus replacements

	System			
	Confinement	Conventional	SpringCalf (H)	SpringCalf (Jx)
No inorganic P fertiliser applied to grassland	5.4	0.6	-5.7	-5.1
If 20 kg P_20_5 applied to all grassland	12.8	8.7	3.1	3.7
If 40 kg P_20_5 applied to all grassland	20.3	16.8	11.9	12.5

Table 1.14 Effect of management system on whole farm emissions (t CO_2e/year), total emissions allocated to milk production (per farm, per cow and per hectare: t CO_2e/year) and total emissions per kg of milk produced (kg CO_2e/kg milk†) (excluding and including carbon sequestration)

	System			
	Confine-ment	**Winter-Calf**	**Spring-Calf (HF)**	**Spring-Calf (J × HF)**
Excluding carbon sequestration (CO_2e)				
Whole farm emissions (t) (before allocation) *	1059	995	847	788
% of emissions allocated to milk production	90	89	87	90
Total emissions allocated to milk production (t)	954	890	739	709
Source of emissions (CO2e/kg milk) (% of total emissions in brackets)				
Enteric fermentation	407 (42)	433 (43)	500 (46)	473 (47)
Concentrate production and transportation	215 (22)	173 (17)	85 (8)	88 (9)
Manure management	189 (19)	208 (21)	244 (23)	228 (22)
Fertiliser manufacture and application	85 (9)	117(12)	180(17)	170 (17)
Land use	15 (2)	10(1)	0 (0)	0 (0)
Fuel and electricity use	33 (3)	25 (3)	35 (3)	28 (2)
Other sources	32 (3)	33 (3)	32 (3)	31(3)
Total emissions per cow (t/cow)	9.54	8.9	7.4	7.1
Total emissions per ha (t/ha)	20.3	17.5	15.1	14.8
Total emissions per kg of ECM (kg/kg ECM)	0.98	1.00	1.07	1.02
Including carbon sequestration (CO_2e)				
Total emissions allocated to milk production (t)	852	770	613	587
Total emissions per cow (t/cow)	8.5	7.7	6.1	5.9
Total emissions per ha (t/ha)	18.1	15.2	12.5	12.2
Total emissions per kg of ECM (kg/kg ECM)	0.87	0.86	0.89	0.84

* Whole farm emissions = Total emissions allocated to milk production + Total emissions allocated to meat production (both from the dairy enterprise).
† Energy corrected milk production.

Table 1.15 Effect of management system on economic performance (concentrate cost, £275/t; milk price, 30 ppl)

	Confinement	WinterCalf	SpringCalf (H)	SpringCalf (Jx)
Milk sold (litres/cow/year)	9084	8404	6572	6137
Outputs (£/cow/year)				
Milk sold @ 32 ppl)	3035	2788	2144	2141
Calves sold	66	66	66	53
Cull cows sold	180	180	180	117.5
Less replacement charge (£/cow/year)	418	418	418	349
Total outputs (£/cow/year)	2863	2616	1972	1962
Variable costs (£/cow/year)	1496	1205	753	728
Gross margin				
£/cow/year	1368	1411	1218	1235
£/ha/year	3655	3363	2941	3020
Pence/litre/year	15.5	17.3	19.1	20.7
Fixed costs (£/cow/year)	619	536	481	481
Net margin				
£/cow/year	749	875	737	754
£/ha/year	2001	2086	1780	1843
Pence/litre/year	8.5	10.7	11.6	12.7

Based on CAFRE Benchmarking Data (2012 and 2013). Thanks are due to Jason McFerran for providing the anonomised data, and for help with its interpretation.

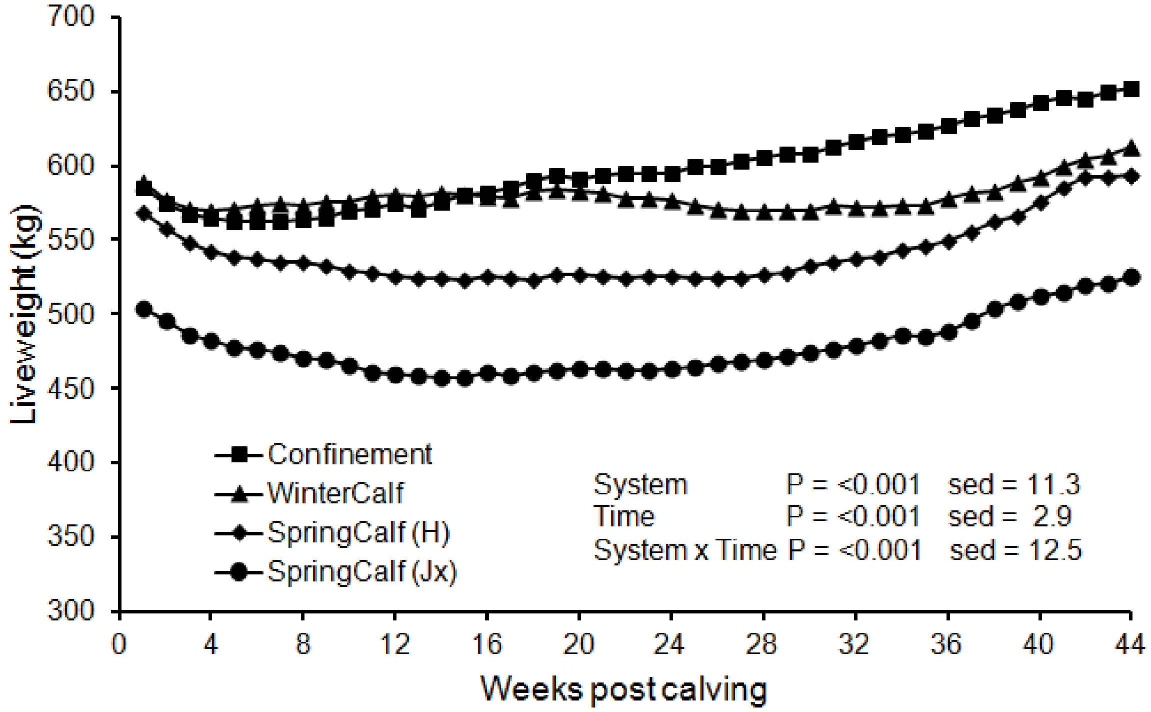

Figure 1.1 Effect of management system on live weight change during the first 42 weeks of lactation

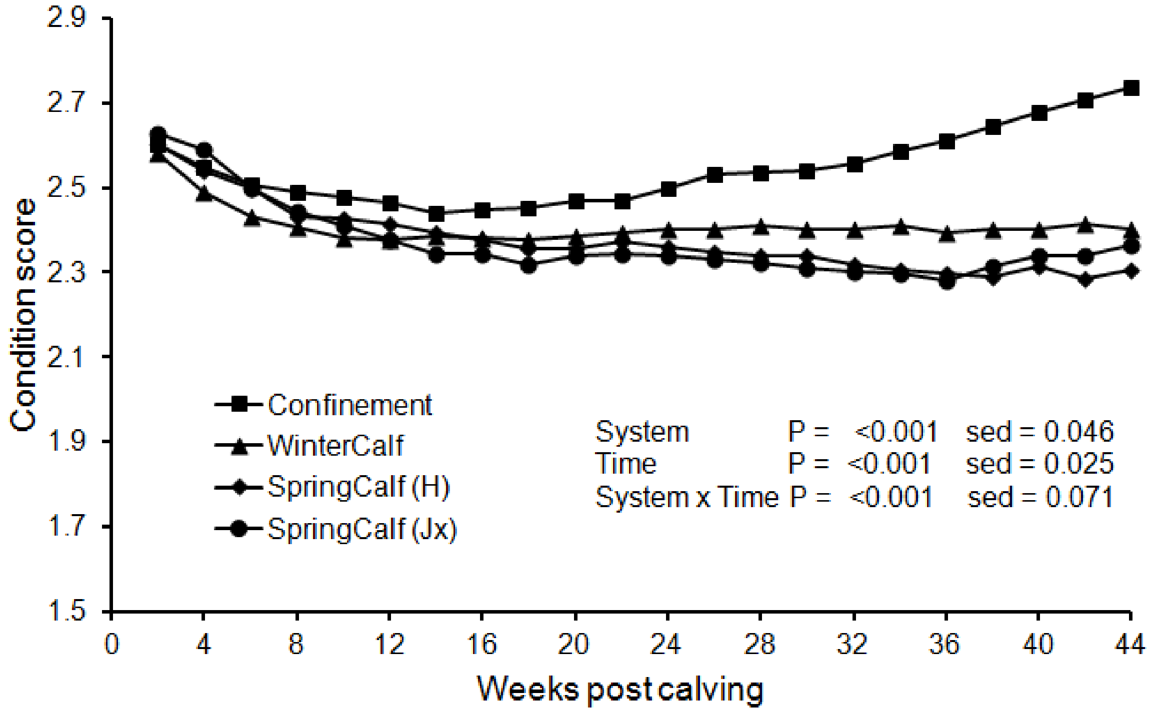

Figure 1.2 Effect of management system on condition score change during the first 42 weeks of lactation

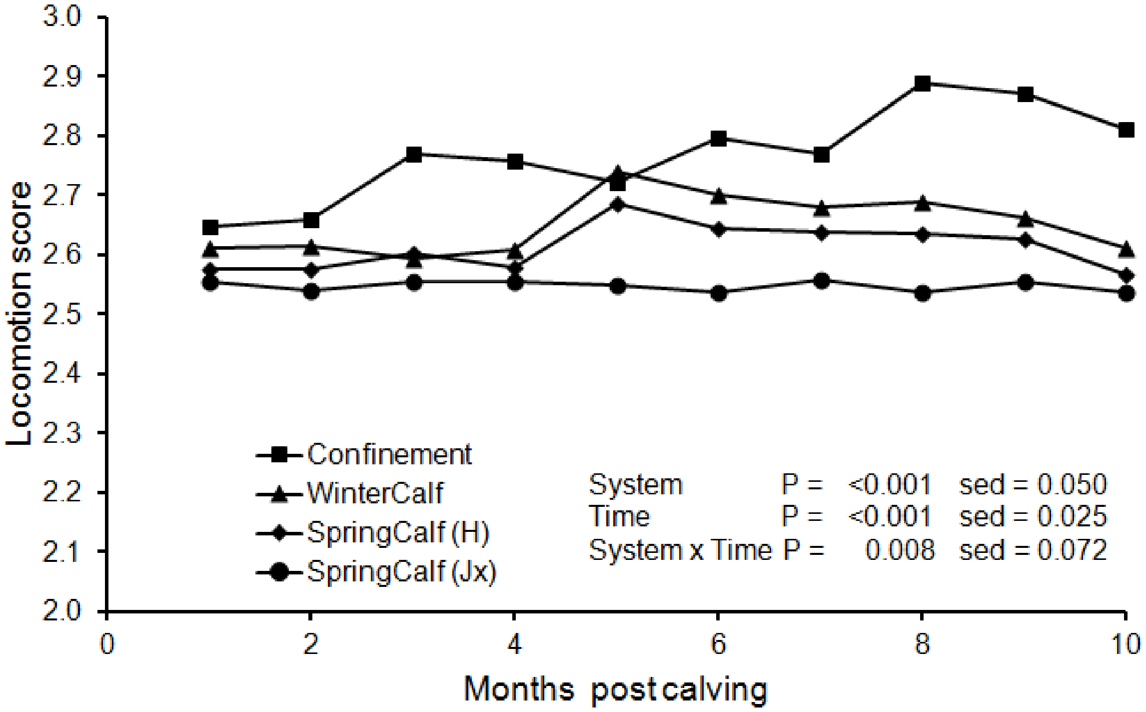

Figure 1.3 Effect of management system on locomotion score change during the first 10 months of lactation

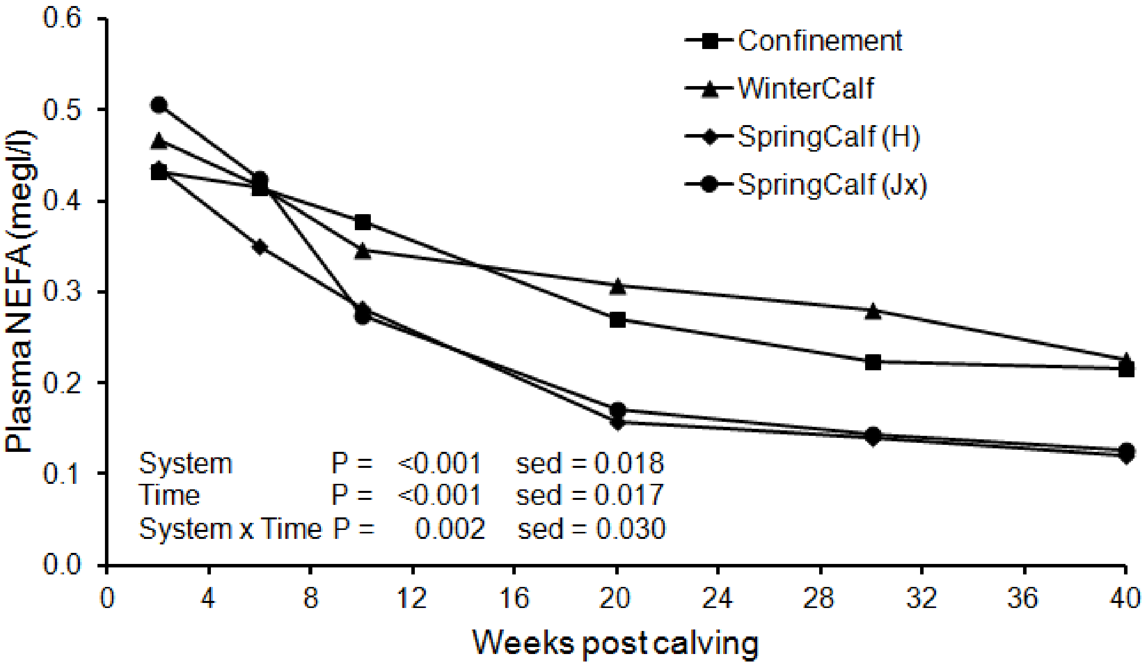

Figure 1.4 Effect of management system on plasma NEFA concentrations during the first 40 weeks post calving

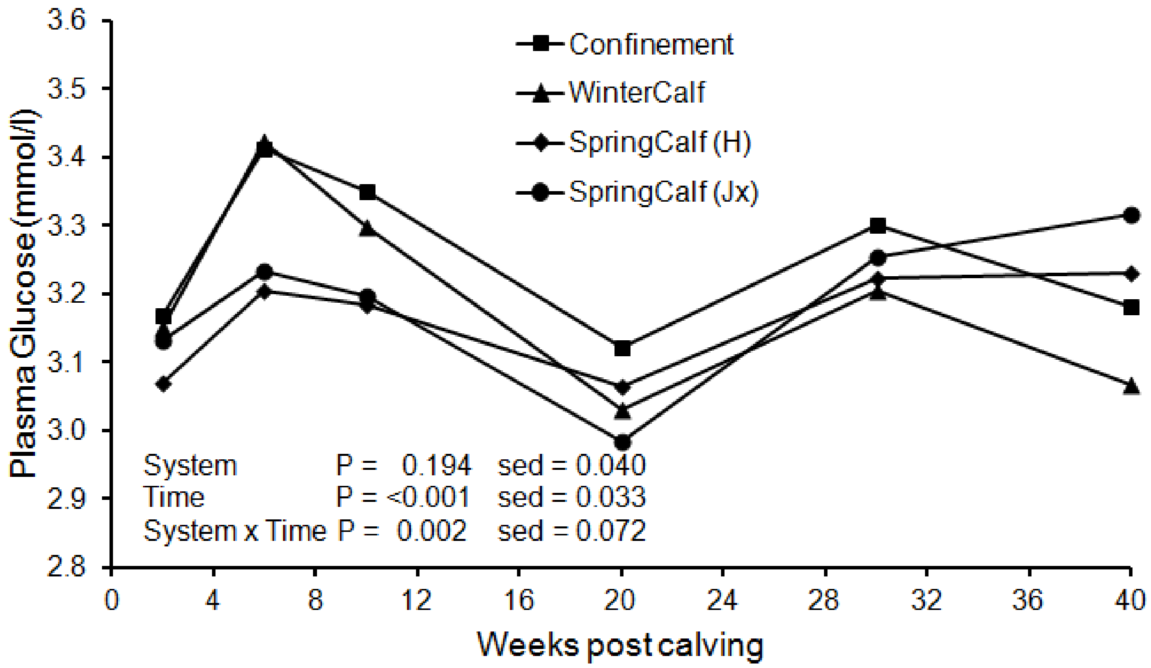

Figure 1.5 Effect of management system on plasma glucose concentrations during the first 40 weeks post calving

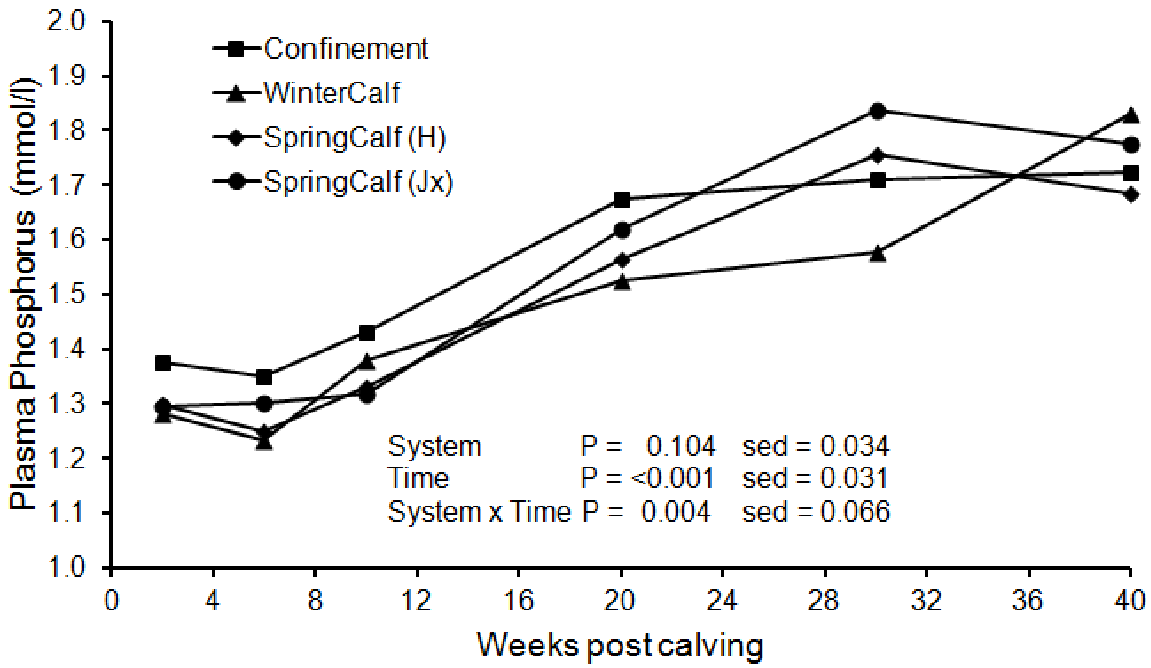

Figure 1.6 Effect of management system on plasma phosphorus concentrations during the first 40 weeks post calving

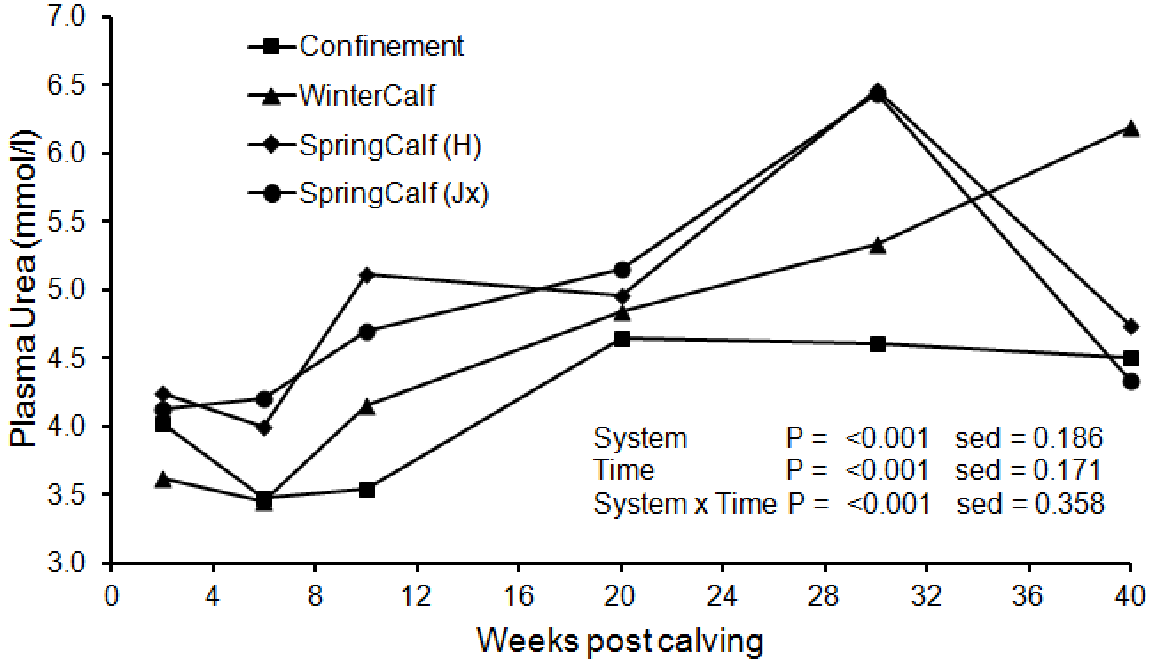

Figure 1.7 Effect of management system on plasma urea concentrations during the first 40 weeks post calving

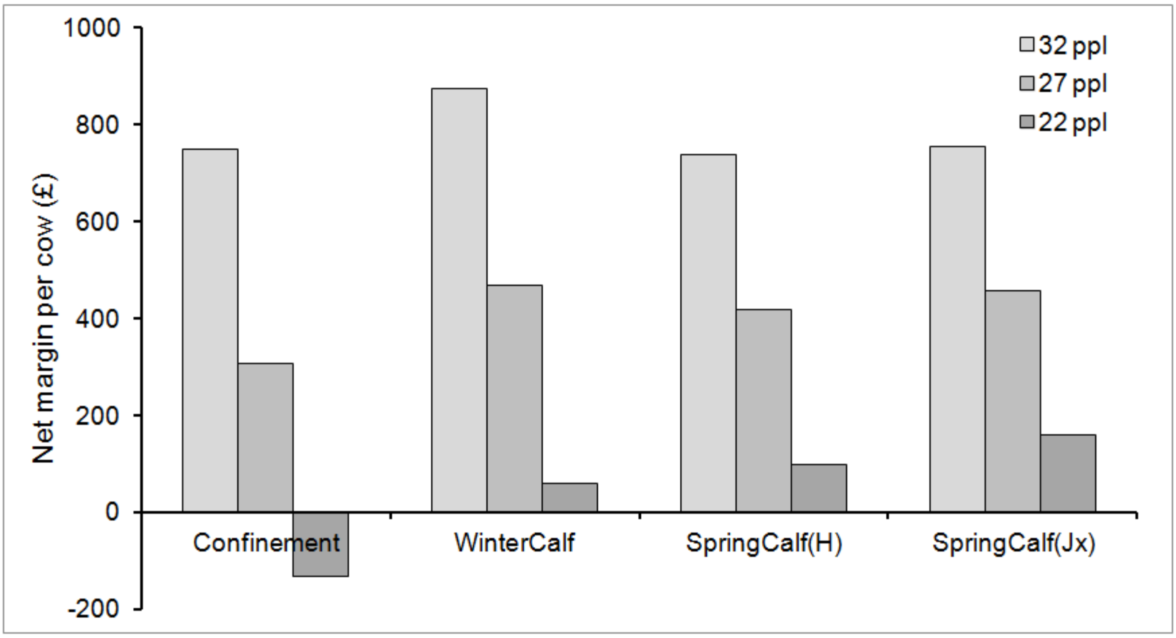

Figure 1.8 Effect of system on net margin per cow (£) across a range of milk prices, when concentrates are costed at £275/t,

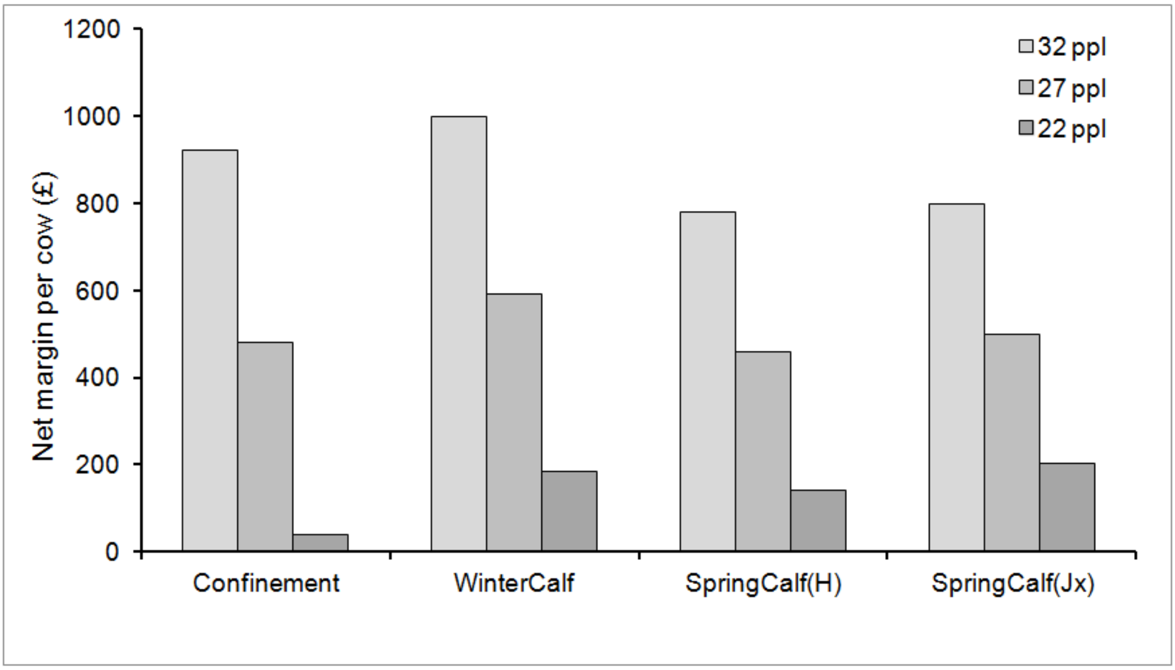

Figure 1.9 Effect of system on net margin per cow (£) across a range of milk prices, when concentrates are costed at £225/t

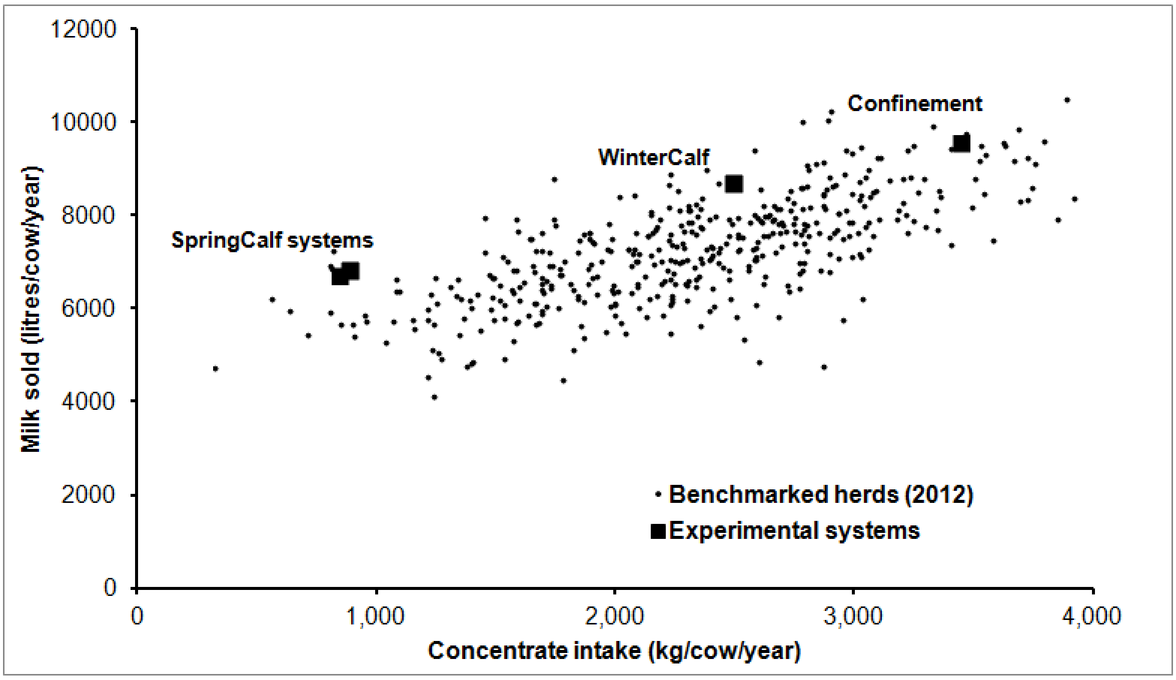

Figure 1.10 Milk yield response to concentrates on 430 Northern Ireland benchmarked farms in 2012 (CAFRE, benchmarking), with performance of the four systems examined in this study superimposed.

SECTION 2

The establishment and management of a replicated 'farmlet' site to measure phosphorus, nitrous oxide and nitrate losses from three contrasting grassland-based milk production systems over two successive years

INTRODUCTION

Part 1 of this report presented animal performance data associated with four contrasting grassland-based systems of milk production. Each of these systems were designed to achieve high levels of animal performance, while having reduced nutrient inputs, and as such, low nutrient losses. Unfortunately, facilities were not available at AFBI to allow completely ‘closed’ systems to operate, that is, systems whereby nutrients produced by livestock on each system would be recycled within the own land areas associated with that system. To overcome this problem, and to achieve the requirements set out by the European Commission in Article 8(6) of the EU Derogation Document, namely *‘this study will focus on nutrient losses, including nitrates leaching, denitrification losses and phosphate losses, under intensive dairy production systems ...’*, a replicated farmlet site was established to allow nutrient losses to be measured. This part of the report describes the establishment of this replicated farmlet site, the management of the site, and crop production within the site. Details of nutrient losses recorded within the site are presented in subsequent sections.

MATERIALS AND METHODS

Overview

This experiment was conducted at the Agri-Food and Biosciences Institute, Hillsborough (latitude 54º27’N; longitude 06º04’W) between February 2009 and March 2011. Part 1 of this report describes dairy cow performance associated with four contrasting grassland-based systems of milk production, namely systems Confinement, WinterCalf, SpringCalf(H) and SpringCalf(Jx). This part of the report describes nutrient losses associated with three of these systems (Confinement, WinterCalf and SpringCalf(Jx).

Nutrient loss measurements were conducted on a site (81 m x 93 m) which was located within the area grazed by cows on the study. The site was established at the same time that grazing paddocks within the main study were established, with the entire site fenced with a double strand of electric fencing to prevent unplanned animal access. The undrained drumlin hill slope on which the site was situated had a northerly aspect, and an average slope of 5 degrees. The soil type was a clay loam (44% sand, 33% silt and 23% clay) overlying Silurian shale parent rock, while the Hydrology of Soils Types (HOST) classification was 24, which is indicative of poorly drained soils with a high capacity for runoff generation. This HOST classification represents approximately 46% of Northern Ireland soils.

The field where the site was located had been re-seeded in 1980 with perennial ryegrass (*Lolium perenne* var. Talbot) and had been grazed by dairy cattle during each subsequent year. During this period the field received only occasional applications of slurry, while during the 4-5 year period prior to the start of the experiment, inorganic fertiliser nitrogen had been applied at rates between 200 and 250 kg N per ha per year.

The layout of the site is presented in Figure 2.1, with the site nominally divided into four replicated blocks (A, B, C and D). Each block contained two mini-grazing paddocks, one each for cows on systems WinterCalf and SpringCalf, a block for growing maize silage and five silage plots. Areas outside of these grazing, maize and silage areas were trimmed using a ride-on lawnmower at fortnightly intervals during the growing season. Prior to the commencement of the experiment (February 2009) soil samples were taken from across the site to a depth of 7.5 cm, with each of Blocks A-D (20 samples per block) sampled separately, and samples subsequently analysed. Across the four blocks the soil was found to have mean concentrations of P, K, Mg, Ca and S (mg/l soil) of 35.7, 193, 242.2, 1101.5 and 12.7, respectively, and a mean pH of 5.93. Carbon lost on ignition was 14.1% of the dry weight of the soil.

The site was managed as described below during the three years of the study, with nutrient loss measurements confined to Years 2 and 3 of the project.

Mini-grazing paddocks

Within the main dairy cow production experiment described earlier, cows on systems WinterCalf and SpringCalf(Jx) grazed within a rotational paddock grazing system, with cows grazing each paddock for a 24-hr period. However, on one occasion during each grazing cycle cows on each of these two systems grazed the mini-paddocks described above. On days when cows were due to graze these mini-paddocks, the 20 cows from each system (fewer in early and late season) were randomly divided into four sub-groups (maximum of five cows per sub-group) following each milking, with each sub-group allowed to graze one of the mini-paddocks over a 24-hr period.

The mini-paddocks measured 0.043 ha (12.2 m x 35 m) for cows on system WinterCalf and a 0.050 ha (14.3 m x 35 m) for cows on system SpringCalf(Jx), with each mini-paddock proportionally 0.25 of the size of the 'one-day' paddocks grazed by cows on these two systems (0.17 and 0.20 ha, respectively) during the remaining days within each grazing cycle. Cows accessed the mini-paddocks via temporary lane-ways which ran along each side of the experimental site. Each mini-paddock was fitted with a water drinker and was fenced using a double strand of electrified fencing. These mini-paddocks were treated identically to the 'one day' grazing paddocks in terms of fertiliser application rates and topping, except that fertiliser was sown by hand, while topping was undertaken using a ride-on lawnmower.

Target inorganic fertiliser nitrogen application levels within the grazing areas within the main study were as follows: a pre-grazing application of urea (proportionally 0.46 N) across the grazing area prior to turnout at 28 kg N/ha, followed thereafter by calcium ammonium nitrate at rates of approximately 45, 30, 30, 30, 25, 25 and 20 kg N/ha following each of grazing cycles 1–7, respectively. Thus over the entire grazing season the target total fertiliser N application rate on the 21 core grazing paddocks was 238 kg N/ha. However, wet weather, delayed turnout dates and a number of other management issues meant that it was not always possible or desirable to follow this schedule. In view of the high stocking density associated with these systems, the moderate soil nitrogen status, and the good grass growth site class, this target fertiliser application rate was considerably lower than the value of 340 kg N/ha outlined in RB209 (7^{TH} Edition), and indeed lower than the maximum permitted value of 272 kg N/ha outlined in the Northern Ireland Nitrates Action Programme.

Dates on which the mini-paddocks were grazed, timing of fertiliser application and fertiliser application levels, are presented in Table 2.2. Until late March, fertiliser applications were in the form of urea (46% N), while all subsequent applications were in the form of calcium ammonium nitrate (27% N in year 1, and 26.6% N and 12% SO_3

in Years 2 and 3). Based on RB209, 7TH Edition, no phosphorus or potassium were applied during the study, plots having P and K indexes of 3 and 2, respectively.

During February 2010 (start of the second grazing season: year 2), three static chambers (described in full later) were placed at randomly selected locations within each of the eight mini-grazing paddocks, with these chambers remaining in place for two full years (until February 2012). Cows had free access to these chambers during periods when cows grazed the mini-grazing paddocks. On the occasions when inorganic fertiliser was being applied to the mini-grazing paddocks, lids were placed on the static chambers to prevent fertiliser being applied within these. The lids were then removed and fertiliser applied to these chambers at the same rate as was applied to the entire mini-paddock.

Maize plots

Each of the four ‘maize plots’ had dimensions 13 m x 10 m. Plots were treated with herbicide at a rate of 3.0 litres per ha each spring (17 April, 12 April and 5 April: Years 1–3, respectively) to kill grass and weeds. Plots were subsequently surface dressed with slurry (49, 32.2 and 32.2 m^3/ha) and ploughed within two hr of slurry being applied (30 April, 20 April and 13 April: Years 1 - 3, respectively). The slurry applied was collected from cows on system Confinement (as described later), and poured over the surface of the plots before being spread using a brush. Plots were then triple harrowed (11 May, 21 April and 18 April: Years 1 - 3, respectively). Prior to the final pass of the harrow, fertiliser nitrogen as calcium ammonium nitrate (61 and 56 kg N/ha: Years 2 and 3, respectively), and potash as Muriate of potash (72 kg K_20/ha during each of Years 1 - 3, respectively) was applied by hand. Maize seeds (varieties Gladi CS, Gladd-ES and Benecia: Years 1 - 3, respectively) were then sown under plastic mulch using a 3.2 m wide seed drill on the same date as harrowing took place. A total of 24 rows of maize seed was sown within each plot (three passes of the seed drill). A pre-emergent herbicide was applied at sowing, while a post-emergence herbicide was applied annually, in late June. Following application of this herbicide, maize plots were surrounded by triple strand electrified fencing (10 cm spacing) in an attempt to prevent badgers accessing the plots. Plots were subsequently harvested by hand on 19 October in Years 1 and 2 and on 31 October in Year 3.

Crop yields were estimated by harvesting six randomly selected sections of maize rows (each 3.0 m long) at a height of approximately 20 cm about ground level, and recording the total weight of fresh maize within each section. One plant harvested from within each of these 3.0 m sections was then chosen at random (6 plants in total) and these six plants subsequently chopped in their entirety and dried to determine oven dry matter content. The entire dried sample was then milled and thoroughly mixed, and a subsample removed and analysed for nitrogen and phosphorus content. Details of slurry applied, and yields of crop harvested, and nutrient recovery rates are presented in Table 2.3.

During February 2010, one static chambers was placed at a randomly selected site within each of the maize plots, prior to ploughing. Chambers were removed at the time of ploughing and subsequently replaced (between two rows of plastic) after the maize crop had been sown, and remained there until the maize was harvested.

Silage plots

The five silage plots (each measuring 8.0 m x 2.0 m) within each of Blocks A-D were randomly allocated to one of 'five' treatments, with three of these simulating 'silage areas' within systems Confinement, WinterCalf and SpringCalf(Jx). A fourth plot was treated identically to WinterCalf in Year 1, while a fifth plot was treated as a zero N plot in Year 1, with the treatments applied to these latter two plots being reversed during Year 2, and again reversed (back to the Year 1 treatments) during year 3. The upper 5.0 m section of each plot was used to measure herbage yields, while during Years 2 and 3, nitrous oxide emissions were measured from static chambers placed within the bottom 3.0 m section of a number of the plots.

Plots were treated with slurry (the entire length: 8.0 m) on three occasions (pre-first grass harvest, post-first grass harvest and post-second grass harvest) during each growing season (2009, 2010 and 2011). On each occasion slurry was applied to each plot by hand using a long necked pouring jug to simulate a trailing shoe application system. Within each plot a total of 40 strips of slurry were applied across the plot, with the space between strips being approximately 20 cm. In the case of the first application (pre-first harvest) herbage was parted by hand prior to application to ensure that slurry was placed at the base of the sward, while during the two subsequent applications, slurry was applied through to the base of the stubble following harvest of herbage. Slurry applied to each treatment plot was collected from cows managed on that treatment, as described later. Inorganic fertiliser was applied to the plots by hand, normally between 5 and 9 days (mean, 7.7 days) after slurry had been applied. Fertiliser applied during spring was in the form of calcium ammonium nitrate (27% N in year 1 and 26.6% N plus 12% SO_3 during Years 2 and 3), while inorganic fertiliser applied post-first and second grass harvests contained proportionally 0.22 N and 0.11 K_2O during year 1, and proportionally 0.245 N and 0.05 K_2O during years 2 and 3). Details of slurry and fertiliser application dates and rates, are presented in Table 2.4, while the mean chemical composition of the slurry applied, weighted for the volume of slurry applied at each application date, is presented in Table 2.5.

Herbage from all plots was harvested on four occasions each year (Table 2.6) using an Agria mower (5400, Agria, Möckmühl, Germany). Yield was assessed by harvesting a 1.0 m wide strip of herbage (target residual height, 5.0 cm) along the centre line of each plot (5.0 m strip), and recording the weight of fresh herbage removed. A sample of the herbage harvested was taken, analysed for oven dry matter content, and the dry sample subsequently analysed for nitrogen and phosphorus content. Herbage dry matter yields and recoveries of nitrogen and phosphorus in harvested crops are presented in Table 2.6.

Nitrous oxide emissions were measured using static chambers which were placed at approximately the centre point of the bottom 3.0 m section of some plots during Years 2 and 3. At the start of the second grazing season (Year 2: February 2010) these static chambers were placed in the Confinement, WinterCalf and SpringCalf(Jx) systems plots, with these remaining in place for two full years (until February 2012). In addition, at the start of Year 3, chambers were also placed in the plots designated as the zero-N treatment that year. Slurry and fertiliser were applied to these chambers at the same time and at the sample application rate as was applied to the remainder of the plot. Slurry for these static chambers was applied in two rows (20 cm apart) across the inside of the chamber. Herbage within the static chambers was harvested to the same height as herbage was harvested in the remainder of the plot using battery powered hand shears (Gardina, Accu 6; Kress and Kastner, Weiterstadt, Germany), and disposed of. The remaining herbage within the lower section of each plot (surrounding the static chamber) was then harvested and disposed of.

Details of slurry collection for application on the experimental plots

During each of Years 1 and 2, slurry applied to the maize and silage plots was collected in an underground tank (approximately 2.5 m x 2.5 x 1.5 m) located within a separate section of the main cow shed, while during Year 3 slurry was collected directly from the floor of the cow shed. Slurry applied during Year 1 was collected during Spring 2009, before any experimental cows were being offered either the mid-lactation or the non-lactating diets, and thus was representative of early lactation diets only. With systems Confinement and WinterCalf, a group of 12 cows (6 from each system) which were being offered the common early lactation diet associated with these systems, were moved to the part of the house fitted with the underground collection tank, and slurry collected over a five-day period (approximately 5.0 m^3). Slurry produced by these cattle was scraped into this tank via an automatic scraper system, with slurry entering the tank via a slatted area over the tank. Thus a common slurry was applied for plots associated with Confinement and WinterCalf in Year 1. Similarly, approximately 10 cows being managed on system SpringCalf were moved to this part of the house and slurry collected over a five-day period (approximately 2.0 m^3). After sufficient slurry had been collected in the tank, the contents of the tank were mixed a number of times using a vacuum slurry tanker, and the contents removed and transferred to a series of 1.0 m^3 'Intermediate Bulk Container' (IBC) storage tanks. These cubes were sealed and stored in an atrium area outside a cold store room where the temperature was typically between 4 and 8ºC, until slurry was required. However, during Years 2 and 3, in an attempt to ensure that slurry collected was 'representative' of that produced by cows during the entire period when cows were confined, slurry was collected from cows being offered early lactation, late lactation and dry period diets, with the number of 'cow slurry collection days' with each diet being in approximate proportion to the number of days that each diet was offered within each of systems Confinement, WinterCalf and SpringCalf(Jx). Thus slurry collection days for the early, late and dry period diets were proportionally 0.55, 0.35 and 0.10 for Confinement, 0.70, 0.15 and 0.15 for WinterCalf, and 0.35, 0.35 and 0.30 for SpringCalf(Jx). With each system, cows from each stage of lactation (and being offered the associated diet for that stage of lactation) were moved in turn to the part of the shed where the collection took place, with cows remaining there until slurry had been collected in proportion to the total number of confined cow feeding days (CFD) associated with that diet. During the final year of the experiment slurry was collected directly from the floor of the cow shed. This was achieved by switching off an automatic slurry scraping system, and blocking off the cow standing passages using wooden boards and sandbags. Slurry which accumulated in the standing area was then collected and stored in IBC's. As in Year 2, slurry was collected from cows in early and late lactation, and from non-lactating cows in proportions to the number of cow feeding days that cows were on each diet.

Assumptions used to determine slurry and fertiliser application rates within the replicated trial site

As outlined above, the dairy unit at AFBI is not equipped with facilities to allow all slurry produced by cows on different systems to be collected, measured and stored separately. Consequently, it was necessary to estimate the total quantities of slurry (and slurry nitrogen) excreted by cows on each system, so as to allow slurry to be applied to silage and maize plots at rates applicable to each system. Calculations and assumptions used while making these estimates are detailed below for systems Confinement, WinterCalf and SpringCalf(Jx). With each of the three systems slurry nitrogen produced during periods when cows were confined was assumed to be available for spreading on land used to produce grass silage, and, in the case of systems Confinement and WinterCalf, maize silage. Based on a herd size of 20 cows per system, and predicted/actual calving dates, the number of lactating and non lactating cows was estimated during each month of the year. Daily intakes of conserved forages (grass silage and maize silage) and concentrates were subsequently estimated for each month during which cows (either lactating or non-lactating cows) were expected to be housed, with these intake estimates taking account of differences in diet type and quality, expected milk output and stage of lactation. Total monthly nitrogen intakes during the housed periods associated with each system were subsequently calculated based on the assumed/actual nitrogen content of each feed offered, and the annual nitrogen intakes for confined periods then calculated. During periods when animals were partially confined (i.e. grazing for part of the day) half of the nitrogen excreted in slurry was assumed to be available for subsequent spreading, with the other half assumed to have been excreted during periods of grazing. Total annual nitrogen excretion (faeces and urine combined) during housed periods were subsequently determined based on an assumed nitrogen use efficiency of proportional 0.4 (proportionally 0.6 of nitrogen consumed excreted in faeces). This value was assumed in view of the lower protein content of the diets offered, together with preliminary results from nitrogen utilisation studies undertaken as part of this experiment.

Maize: Based on the estimated intake of maize silage within each of systems Confinement and WinterCalf, an assumed yield of harvested maize of 9.0 t DM/ha, and an assumed in-silo and feeding loss of 15%, the area of forage maize required to sustain systems Confinement and WinterCalf was determined. Nutrients were applied to maize crops using the framework of requirements set out within RB209 (7^{TH} Edition). As maize was planted following ploughing out of grassland, Soil Nitrogen Supply (SNS) Indices were assumed as 2, 2 and 1 during Years 1-3, respectively (Medium soil, 3-5 year grassland, managed on a low N system). On this basis, slurry only (49 m^3/ha) was applied to maize plots during Year 1. However, there was evidence of nitrogen deficiency within the growing crop during Year 1, and as such inputs of slurry nitrogen were reduced and inorganic nitrogen levels increased during Years 2 and 3, with the objective of providing 120 kg available N/ha during each of these two years. No inorganic P was applied as the soil P index (3) meant that the P requirement of the crop could be met by slurry alone. In view of the soil potash status of 2, a K_2O requirement of 180 kg/ha was determined. After a deduction for K_20 provided by slurry, muriate of potash (130 kg/ha) was used to supply 78 kg K_20/ha.

Grass silage: Based on the land area required for growing maize with each of systems Confinement and WinterCalf, and deducting the quantity of slurry nitrogen applied to the maize crop, the quantity of slurry N remaining for use on grass silage within each of these two systems was determined. In the case of system SpringCalf, all slurry calculated as produced during confinement periods was assumed to be available for grass silage, this system involving no maize silage. Within each of these systems, approximately proportionally 0.5, 0.3 and 0.2 of the remaining slurry nitrogen was assumed to be available for application prior to first, second and third harvests of grass silage. Nutrients applications were based on the requirements set out within RB209 (7^{TH} Edition), with nominal targets of 120, 100 and 80 kg N/ha prior to first, second and third cuts of silage respectively (moderate soil N status). The silage system operated was managed as a three-cut system, with the residual harvest (late October/early November) assumed to be equivalent to a late grazing, with no fertiliser or slurry applied prior to this harvest. While the good grass growth class would have allowed these figures to be increased by up to 40 kg/ha, this was not adopted. Availability of slurry nitrogen was again assumed to be proportionally 0.4, with the remaining nitrogen requirements of the crop met from inorganic fertiliser nitrogen. No inorganic P fertiliser was applied in view of the P index of the soil and P supplied by slurry, while the compound fertiliser applied on each occasion allowed the K requirements of the crop to be met.

Site where nutrient losses were monitored within a replicated plot study (Spring 2011).

Cows grazing within the experimental site, with silage plots in the foreground and maize plots in the background (Summer 2011).

Static chambers within grazing plots.

Maize plots during Year 1.

Applying slurry to silage plots using a ‘simulated’ trailing shoe technique.

Silage plots after slurry was applied.

Silage plots, with static chambers in place.

RESULTS AND DISCUSSION

Mean monthly temperature and rainfall during the four-year period covered by the experiment are presented in Figures 2.2 and 2.3, respectively. Hillsborough has a cool maritime climate and for the two main measurement years the mean annual rainfall was 884 mm (Year 2: 2010) and 922 mm (Year 3: 2011). Similarly, the mean annual temperature for Years 2 and 3 was 11.7ºC and 13ºC. During Year 2 (2010) average monthly rainfall was lower than the 10-year average in April, May June, October and December, while being higher in July, September and November. During Year 2 (2011) average rainfall was lower than the 10-year average in April, July and August while being higher in February and October. During Year 1 average monthly temperatures were higher than the 10-year average during June and lower during December, while in Year 2 temperatures were higher in April and November.

Although the experiment was conducted over a three-year period, this was not deemed sufficiently long to provide a robust examination of changes in soil properties over time. Nevertheless, a number of ‘trends’ were observed within the data (Table 2.1). Firstly, there was a ‘trend’ for soil P levels within the grass silage plots to fall over time, although soil N levels and soil total carbon levels remained unchanged. No such trends in soil P levels were observed within the grazing plots. Within the maize plots there was a clear trend for both soil P and soil K levels to increase over time.

While soil N levels and total carbon levels did not appear to change during the three-year period within the maize plots, these tended to be lower than those recorded within the grass silage and grazing plots.

Maize yields during Years 1 and 2 were extremely poor, while there was a total crop failure during Year 3 (Table 2.3). These poor yields can be attributed to a number of factors, including the fact that the experimental plots were located on a northerly facing slope. However, 'pest attack' was the primary reason for the poor crop performance, especially during Years 2 and 3. Crops were attacked by birds at the germination stage and at the young seedling stage, while the growing crop was attacked by badgers. Despite repeated efforts to protect the crops, the location of the small maize plots (in a large grassland field) close to a large area of woodland made them particularly vulnerable to pest attack. The poor yields were reflected in the low recoveries of applied N and P.

Total yields of herbage harvested within the Confinement, WinterCalf and SpringCalf(Jx) grass silage plots were 14.1, 14.6 and 13.35 t DM/ha (Table 2.6), with these similar to yields (15.8 t) recorded previously within grass plots within a four-harvest system by Ferris *et al*. (2002). The trend for a lower yield with the SpringCalf(Jx) system likely reflects the slightly lower total N application with this system. As expected, the herbage yields with the Zero N plots were approximately half of those recorded with the treatments which received organic and inorganic N. Dry matter yields varied little over the three-year period that the experimental site was in operation. Across the Confinement, WinterCalf and SpringCalf(Jx) silage plots, proportionally 0.73 of N applied and 1.52 of P applied was recovered in crop harvested.

REFERENCE

Ferris, C.P. Gordon, F.J. and Patterson, D.C. (2002). Effect of harvesting frequency on herbage dry matter production. *Proceedings of Agricultural Research Forum of the Irish Grassland and Animal Production Association*, Tullamore, Ireland. Page 70.

Table 2.1: Some soil properties determined during the early winter periods in each of Years 1 (2009), 2 (2010) and 3 (2011)†

Land use	Experimental system	Year	P (mg/l soil)	K (mg/l soil)	Mg (mg/l soil)	Ca (mg/l soil)	S (mg/l soil)	Soil pH	Loss on ignition (% DM)	Total nitrogen (% DM)	Total carbon (% DM)
Silage	Confinement	1	30.6	174	288	1153	12.3	5.9	15.1	0.6	6.5
		2	31.1	184	287	1220	14.2	5.9	15.5	0.6	6.4
		3	29.5	151	293	1296	13.7	5.9	15.7	0.6	6.6
	WinterCalf	1	37.4	192	295	1247	11.9	6.0	15.5	0.6	6.6
		2	34.6	219	279	1274	14.1	6.0	15.7	0.6	6.7
		3	31.7	151	296	1379	13.4	6.0	14.9	0.6	6.4
	SpringCalf	1	32.1	139	272	1224	10.9	5.9	15.2	0.6	6.3
		2	33.8	159	268	1215	13.7	5.9	15.8	0.6	6.8
		3	28.7	120	250	1231	12.5	5.8	15.2	0.6	6.7
	Zero N	1	32.4	138	251	1157	12.5	5.9	15.0	0.6	6.3
		2	32.7	164	278	1125	15.2	5.8	15.3	0.6	6.4
		3	29.4	122	233	1200	12.8	5.8	14.7	0.6	6.3
Grazing	WinterCalf	1	33.6	176	264	1122	13.2	5.9	15.9	0.6	6.8
		2	32.4	190	251	1110	16.4	5.8	16.1	0.6	6.8
		3	33.8	173	234	1100	15.1	5.8	16.1	0.6	7.0
	SpringCalf	1	31.4	170	248	1078	12.1	5.8	15.1	0.6	6.5
		2	31.5	182	238	1057	15.6	5.7	15.3	0.6	6.5
		3	30.9	157	218	1081	13.9	5.7	15.8	0.6	6.7
Maize	Confinement and WinterCalf	1	30.8	153	190	1192	10.2	6.0	11.4	0.4	4.1
		2	35.7	236	177	1091	11.8	6.0	11.5	0.4	4.5
		3	36.6	269	169	1077	10.5	5.9	11.1	0.4	4.3

+ Mean analysis of samples taken from the four experimental plots across Blocks A, B, C and D

Table 2.2: Dates on which mini-grazing plots were grazed, timing of inorganic fertiliser nitrogen applications, and fertiliser application rates

	Year 1 (2009)		Year 2 (2010)		Year 3 (2011)	
	WinterCalf	SpringCalf	WinterCalf	SpringCalf	WinterCalf	SpringCalf
Date of pre-grazing application	17 February†	17 February†	22 March†			
N applied (kg/ha)	19.6	19.6	27.9			
Date of 1st grazing	02 April	29 March	03 April	19 February†	13 April	22 March†
Date of fertiliser application	03 April	30 March	08 April	22 March	14 April	24 March
N applied (kg/ha)	44.8	45.0	43.9	45.0	44.6	28.5
Date of 2nd grazing	30 April	30 April	26 April	26 April	04 May	18 April
Date of fertiliser application	05 May	05 May	28 April	03 May	05 May	19 April
N applied (kg/ha)	30.0	30.0	29.7	29.9	29.7	44.8
Date of 3rd grazing	26 May	26 May	20 May	13 May	23 May	10 May
Date of fertiliser application	28 May	28 May	21 May	14 May	25 May	10 May
N applied (kg/ha)	30.0	30.0	29.7	29.9	29.7	29.9
Date of 4th grazing	10 June	09 June	10 June	03 June	13 June	31 May
Date of fertiliser application	15 June	15 June	11 June	08 June	15 June	01 June
N applied (kg/ha)	30.0	30.0	29.7	29.9	29.7	29.9
Date of 5th grazing	08 July	08 July	01 July	29 June	04 July	21 June
Date of fertiliser application	09 July	09 July	02 July	30 June	06 July	21 June
N applied (kg/ha)	30.0	30.0	24.8	24.9	24.8	24.9
Date of 6th grazing	12 August	13 August	26 July	22 July	25 July	14 July
Date of fertiliser application	18 August	18 August	29 July	22 July	28 July	16 July
N applied (kg/ha)	24.8	24.9	24.8	24.9	24.8	24.9
Date of 7th grazing	17 Sept	17 Sept	19 August	10 August	15 August	08 August
Date of fertiliser application	14 Sept	14 Sept	20 August	12 August	17 August	10 August
N applied (kg/ha)	24.8	24.9	24.8	24.9	24.8	24.9
Date of 8th grazing			09 Sept	02 Sept	05 Sept	30 August
Date of fertiliser application				03 Sept	07 Sept	08 Sept
N applied (kg/ha)				24.9	24.8	24.9
Date of 9th grazing			06 October	25 Sept	20 Sept	21 Sept
Date of 10th grazing					21 October	21 October
Total inorganic nitrogen applied (kg N/ha)	234	234	235	234	233	233

† Applied as urea nitrogen (46% N); all other applications as CAN (27% N in Year 1, 26.6% N in Years 2 and 3)

Table 2.3: Details of slurry and fertiliser applied to maize plots, and nutrient recovery in crops

	Year		
	1	2	3
Date slurry applied	30/4/09	20/4/10	13/4/11
Application rate (t/ha)	49.0	32.2	32.2
Composition of slurry applied			
Dry matter (g/kg fresh)	82.4	101.1	126.7
pH	7.16	7.12	7.59
Nitrogen (g/kg fresh)	3.51	4.12	5.11
Ammonia (mg/kg)	2215	2175	2422
Phosphorus (g/kg DM)	5.92	5.85	6.23
Total N applied from slurry (kg/ha)	172.3	132.4	164.4
Total ammonia applied in slurry	108.7	69.9	77.9
Date CAN applied	NA	26/4/10	18/4/11
N applied via CAN (kg/ha)	0	61	56
Total N applied	172	193	221
Total available N applied	109	131	134
Composition of maize harvested			
Dry matter (g/kg)	260 (8.3)	293 (28.7)	ND
Nitrogen (g/kg DM)	13.6 (0.63)	12.9 (0.65)	ND
Phosphorus (g/kg DM)	2.70 (0.243)	2.19 (0.164)	ND
Starch (g/kg DM)	139 (18.3)	230 (33.9)	ND
DM yield (kg DM/ha)	8.2 (4.37)	5.4 (0.66)	ND
Recovery of applied phosphorus in crop (proportion)	0.92 (0.475)	0.64 (0.169)	ND
Recovery of applied nitrogen in crop (proportion)	0.64 (0.315)	0.38 (0.090)	ND

ND, not determined due to crop failure

Table 2.4: Details of slurry and fertiliser nitrogen applications to silage plots with systems Confinement WinterCalf and SpringCalf during years 1-3

	YEAR 1 (2009)			YEAR 2 (2010)			YEAR 3 (2011)		
	Confinement	WinterCalf	SpringCalf	Confinement	WinterCalf	SpringCalf	Confinement	WinterCalf	SpringCalf
Pre first harvest									
Slurry application rate (m^3/ha)	38.1	43.8	25.6	29.4	36.3	24.4	35.6	31.9	21.9
Slurry N applied (kg/ha)	141	162	93	103	128	78	214	186	89
Slurry ammonia N content applied (kg/ha)	82.1	94.2	54.2	57.4	61.7	42.5	82.7	82.3	46.6
Fertiliser N applied (kg/ha)	54.0	45.6	70.9	47.4	39.7	59.2	43.1	47.7	68.5
Total N applied (kg N/ha)	195.4	207.9	163.9	149.9	167.3	136.7	257.2	233.2	157.1
Total available N applied (kg N/ha)	136	140	125	105	101	102	126	130	115
Date of 1st harvest	28 May	28 May	28 May	02 June	02 June	02 June	25 May	25 May	25 May
Post first harvest									
Slurry application rate (m^3/ha)	23.1	26.3	15.0	21.4	25.0	16.9	19.9	17.9	12.1
Slurry N applied (kg/ha)	73	83	47	75	96	77	88	87	42
Slurry ammonia N content applied (kg/ha)	53.1	60.2	29.7	55.3	62.2	42.5	54.0	48.0	31.0
Fertiliser N applied (kg/ha)	56.4	52.3	67.4	59.0	50.5	58.2	54.8	55.1	73.2
Total N applied (kg N/ha)	129.5	135.2	114.2	133.8	146.5	134.9	142.9	142.3	115.2
Total available N applied (kg N/ha)	109	112	97	114	113	101	109	103	104
Date of 2nd harvest	22 July	22 July	22 July	29 July	29 July	29 July	18 July	18 July	18 July

Table 2.4 (continued)

	YEAR 1 (2009)			YEAR 2 (2010)			YEAR 3 (2011)		
	Confinement	WinterCalf	SpringCalf	Confinement	WinterCalf	SpringCalf	Confinement	WinterCalf	SpringCalf
Post second harvest									
Slurry application rate (m^3/ha)	21.3	23.8	22.5	9.1	10.7	7.3	8.5	7.7	5.2
Slurry N applied (kg/ha)	57	63	38	36	41	28	39	36	18
Slurry ammonia N content applied (kg/ha)	46.4	51.8	44.9	20.3	23.7	16.7	23.2	19.7	12.9
Fertiliser N applied (kg/ha)	37.1	34.4	44.0	46.2	43.9	49.8	43.9	44.1	51.6
Total N applied (kg N/ha)	93.9	97.8	82.3	82.2	84.9	77.7	83.1	80.3	70.0
Total available N applied (kg N/ha)	84	86	89	67	68	66	67	64	64
Date of 3rd harvest	15 Sept	15 Sept	15 Sept	21 Sept	21 Sept	21 Sept	15 Sept	15 Sept	15 Sept
Residual harvest									
Date of residual harvest	28 Oct	28 Oct	28 Oct	26 Oct	26 Oct	26 Oct	14 Nov	14 Nov	14 Nov
Whole season nitrogen applications									
Total N applied (kg/ha)	419	441	360	366	399	349	483	456	342
Total available N applied (kg/ha)	329	338	311	286	282	269	302	297	284

Table 2.5: Chemical composition of slurry applied to silage plots with systems Confinement, WinterCalf and SpringCalf during years 1-3 (weighted according to quantities applied prior to Harvests 1-3)

	YEAR 1 (2009)			YEAR 2 (2010)			YEAR 3 (2011)		
	Confinement	WinterCalf	SpringCalf	Confinement	WinterCalf	SpringCalf	Confinement	WinterCalf	SpringCalf
Dry matter (g/kg)	61.4	61.5	55.7	81.8	80.8	84.9	99.1	105.2	78.2
pH	8.02	8.02	8.44	7.70	8.06	7.71	7.82	7.55	7.85
Nitrogen (g/kg)	3.29	3.29	2.82	3.56	3.68	3.75	5.33	5.37	3.81
Ammonia (mg/kg)	2201	2200	2041	2223	2052	2093	2498	2609	2310
Phosphorus (g/kg DM)	5.74	5.74	5.71	5.80	5.45	5.88	6.22	5.80	6.38

Table 2.6: Effect of system on herbage DM yields and organic matter within silage plots, on herbage composition, and on the proportion of applied nitrogen and phosphorus applied apparently recovered in harvested crop

Harvest		Treatment				SEM	Year			SEM	Significance		
		Confine.	Winter Calf	Spring Calf	Zero N		1	2	3		Year	Treat.	Year x Treat.
1	DM yield (t DM/ha)	6.19	6.41	5.76	4.22	0.159	6.17	5.64	5.12	0.138	<0.001	<0.001	0.613
2	DM yield (t DM/ha)	4.09	4.21	3.72	1.10	0.128	3.35	2.61	3.88	0.111	<0.001	<0.001	<0.001
3	DM yield (t DM/ha)	3.00	3.08	2.98	1.06	0.077	2.58	2.45	2.56	0.067	<0.001	0.328	0.370
4	DM yield (t DM/ha)	0.82	0.90	0.90	0.57	0.040	0.76	0.80	0.83	0.034	<0.001	0.386	0.737
1-4	DM yield (t DM/ha)	14.1	14.60	13.35	6.96	0.249	12.86	11.50	12.40	0.216	<0.001	<0.001	0.031
1	OM yield (t DM/ha)	5.79	5.93	5.33	3.93	0.147	5.71	5.26	4.78	0.128	<0.001	<0.001	0.576
2	OM yield (t DM/ha)	3.79	3.91	3.45	1.03	0.121	3.10	2.40	3.63	0.105	<0.001	<0.001	<0.001
3	OM yield (t DM/ha)	2.75	2.83	2.72	0.96	0.071	2.35	2.22	2.37	0.061	<0.001	0.165	0.408
4	OM yield (t DM/ha)	0.74	0.81	0.81	0.51	0.037	0.68	0.73	0.74	0.032	<0.001	0.313	0.772
1-4	OM yield (t DM/ha)	13.07	13.47	12.31	6.43	0.229	11.83	10.6	11.53	0.199	<0.001	<0.001	0.024
1	N content (g/kg DM)[1]	17.2	14.9	18.1	11.7	0.68	12.8	15.4	18.2	0.59	<0.001	<0.001	0.004
2	N content (g/kg DM)[1]	21.5	20.5	18.7	14.9	1.02	17.1	21.5	18.1	0.88	<0.001	0.003	0.022
3	N content (g/kg DM)[1]	23.3	25.1	27.3	20.3	0.89	22.2	27.1	22.6	0.77	<0.001	<0.001	<0.001
4	N content (g/kg DM)[1]	37.7	36.4	36.3	36.9	0.64	35.9	38.2	36.4	0.55	0.418	0.014	0.824
1-4	N content (g/kg DM)[1]	20.9	19.9	21.4	15.4	0.50	17.1	20.7	20.4	0.44	<0.001	<0.001	<0.001
1	N recovered in crop[2]	0.55	0.48	0.68		0.029	0.54	0.62	0.54	0.029	<0.001	0.081	0.061
2	N recovered in crop[2]	0.64	0.60	0.58		0.030	0.62	0.49	0.69	0.030	0.327	<0.001	0.146
3	N recovered in crop[2]	0.82	0.88	1.06		0.043	0.87	0.96	0.92	0.043	0.002	0.371	0.025
1-4	N recovered in crop[2]	0.70	0.67	0.82		0.020	0.72	0.74	0.73	0.020	<0.001	0.790	0.003
1	P content (g/kg DM)[1]	2.69	2.53	2.65	2.54	0.06	2.83	2.41	2.57	0.052	0.186	<0.001	0.186
2	P content (g/kg DM)[1]	2.88	2.78	2.7	2.86	0.077	2.82	3.08	2.51	0.067	0.357	<0.001	0.223
3	P content (g/kg DM)[1]	3.36	3.58	3.57	3.92	0.131	3.59	3.89	3.34	0.114	0.04	0.007	0.218
4	P content (g/kg DM)[1]	3.93	4.18	4.49	4.17	0.151	4.40	4.18	3.99	0.130	0.094	0.098	0.951
1-4	P content (g/kg DM)[1]	2.95	2.91	2.99	2.94	0.057	3.07	2.98	2.79	0.049	0.818	0.001	0.370
1	P recovered in crop[2]	1.01	0.89	1.39		0.044	1.4	1.07	0.82	0.044	<0.001	<0.001	0.042
2	P recovered in crop[2]	1.3	1.23	1.70		0.075	1.65	1.06	1.52	0.075	<0.001	<0.001	0.229
3	P recovered in crop[2]	1.92	2.11	3.52		0.128	1.93	2.87	2.76	0.128	<0.001	<0.001	0.083
1-4	P recovered in crop[2]	1.32	1.28	1.94		0.050	1.72	1.47	1.35	0.05	<0.001	<0.001	0.063

Confine., Confinement; Treat., Treatment

[1] Of herbage harvested, [2] Nitrogen and phosphorus apparently recovered in harvested crop as a proportion of total nitrogen and total phosphorus applied in fertiliser and slurry

Figure 2.1: Layout of grazing, silage and maize plots within the Blocks A–D within the experimental site

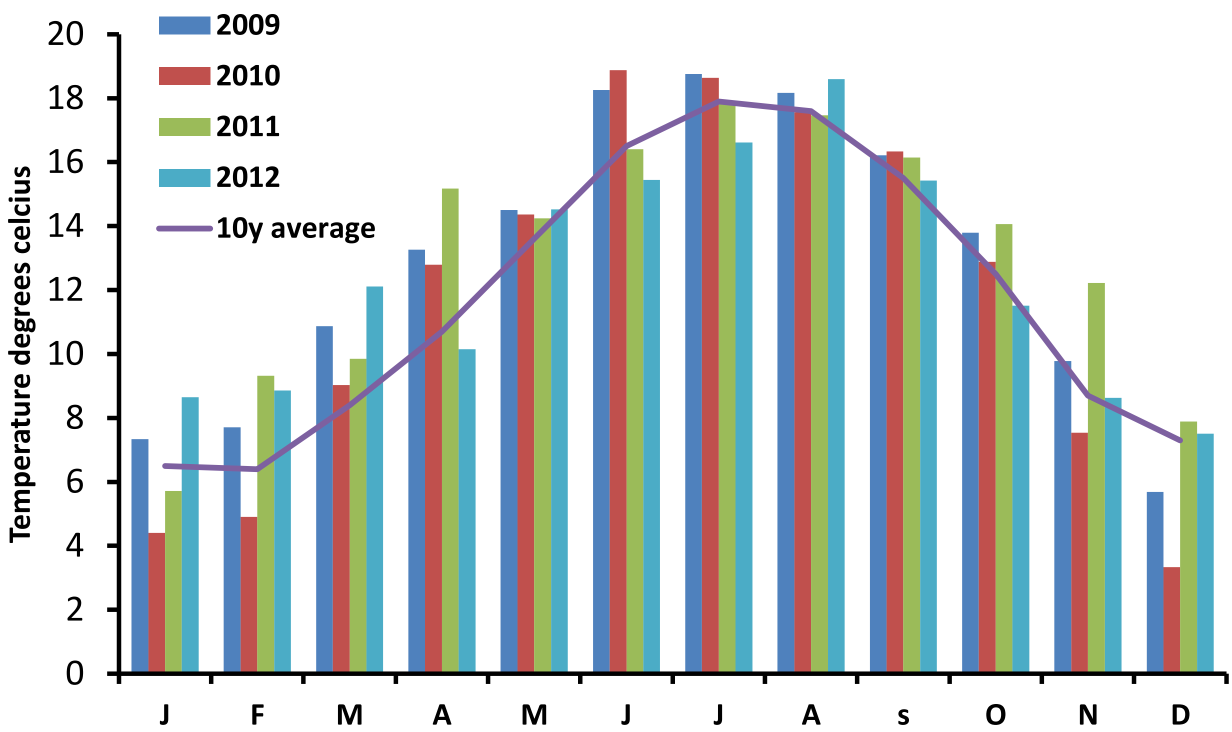

Figure 2.2: Mean monthly temperatures (ºC) during the four-year period when the farmlet site was in operation, together with average monthly temperature trend for the previous 10-year period

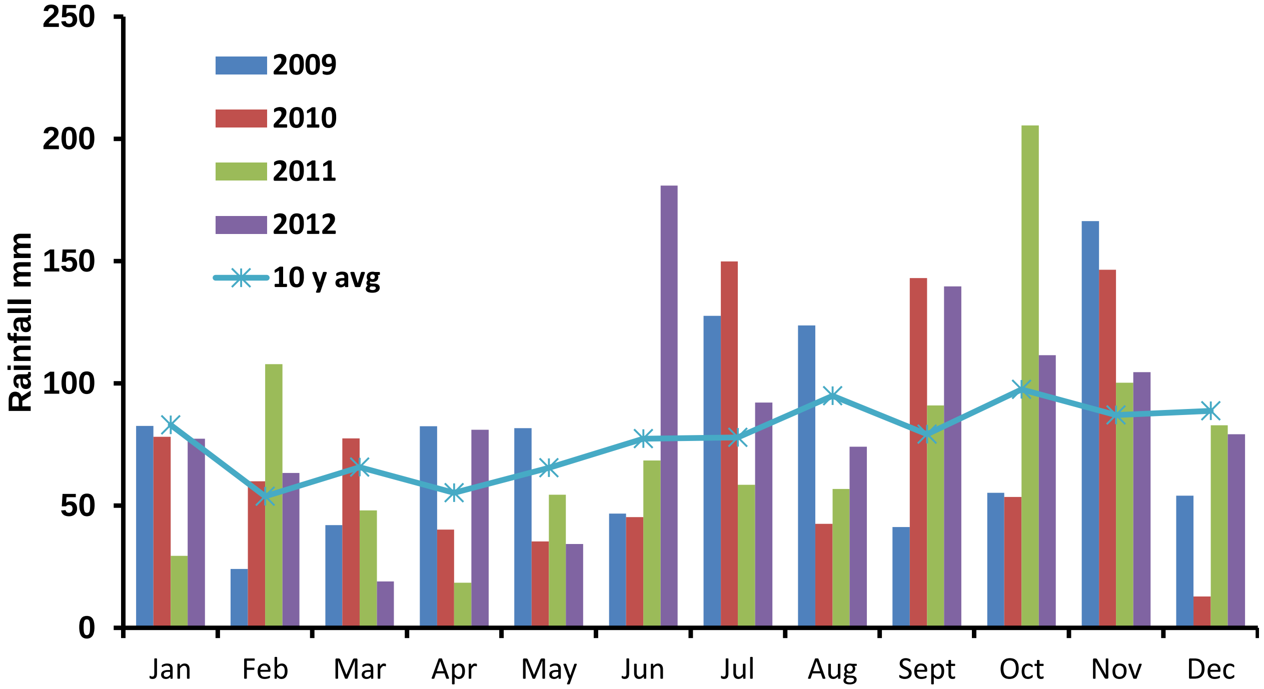

Figure 2.3: Mean monthly rainfalls (mm) during the four-year period when the farmlet site was in operation, together with average monthly temperature trend for the previous 10-year period

SECTION 3

Evaluating the Impact of Grazing on Phosphorus Loss from Grassland Soils

INTRODUCTION

In temperate climates, grasslands make up a significant proportion of available land, with the majority being utilised for grazing of agricultural livestock. In the UK and Northern Ireland (NI) grasslands account for approximately 70% of land use (Bilotta *et al*., 2007), with 94% of the agricultural land area in NI under grassland (DARD, 2011). The link between grazing intensity and the deterioration of water quality has been established, with nutrient export increasing with stocking density (Hubbard *et al*., 2004). Foy and Kirk (1995) reported that on a scale of 1 (good) to 6 (bad), a decrease in water quality of 1 class was associated with a increase in stocking rate of 0.6 dairy cow equivalent/ha. More recently Bilotta *et al.* (2010) concluded that suspended sediment (SS) in streams draining improved temperate grasslands were often in excess of the Freshwater Fish Directive (78/659/EU) water quality guidelines value of 25 mg SS/l and posed a significant threat to aquatic ecosystems.

Due to high rainfall frequency and prevalence of high soil moisture contents in Northern Ireland, dairy cattle are housed for up to six months of the year, during which time they are fed silage and imported concentrate feeds. However, on most dairy farms cows are still given access to grazed pasture, with grazing systems varying greatly from farm to farm. For example, grazing systems differ in terms of cow genotype and stage of lactation of cows during the grazing period, start and end dates of the grazing season, supplementary concentrate feed levels while grazing, and stocking rates. The most ‘extreme’ grass-based systems normally seek to maximise milk output from grazed grass, and do this by using cow genotypes which are suited to grazing (often lighter cows with high grass intake capacities, such as Jersey crossbred cows, e.g. Vance *et al*., 2012), have cows which calve in the spring so that peak yield coincides with periods of maximum grass growth, commence grazing early in the spring and extend the grazing season into the autumn, and feed low levels of supplementary concentrates. Within these low input systems, farmers aim to maximise the inclusion of grazed grass in the diets of cattle as it is the cheapest available feed (Mayne and Laidlaw, 1995). The extension of the grazing season has been shown to improve milk production and reduce the consumption of ensiled forage, resulting in a higher net profit for the farmer (Mayne and Laidlaw, 1995, Sayers and Mayne, 2001).

However, managing systems that maximise the intake of grass through extension of the grazing season can pose significant challenges in terms of the threat posed to water quality. A key concern is the potential impact of grazing on soil structure, soil hydrology and nutrient export, especially during periods of high antecedent soil moisture conditions in spring and autumn. Herbin *et al.* (2011) demonstrated that soil moisture deficit (SMD) was a key factor controlling the impact of grazing animals on soil structural properties. They found that at a SMD of 0 mm there was a 6.1% increase in bulk density compared to 0.5% increase at SMD of 29 mm. Doody *et al*. (2010) reported that soil moisture contents in a surface water gleyed soil in Northern Ireland, were at or above field capacity on average over 75% of days in February, which highlights the risks posed to soil quality from grazing during early spring. Structural changes associated with grazing included; compaction, plugging and poaching, which occur during grazing on low/medium, medium and medium/high soil moisture conditions, respectively (Greenwood and McKenzie, 2001). Under good grazing management it would be expected that plugging and poaching should be minimised, as the current codes of good agricultural practice, prohibit grazing during periods when soil is at or close to saturation. However, soil compaction can occur in unsaturated conditions, when the carrying capacity of the soil is exceeded, compressing soil particles closer together and expelling air and/or water from the soil pore spaces (Greenword and McKenzie, 2001). These impacts arise due to the grazing animal exceeding the load carrying capacity of the soils, with a 530 kg dairy cow exerting as much as 300 kPa of pressure when walking during grazing.

The impact of grazing on soil physical properties varies depending on animal age, species, stocking density, soil moisture and vegetation cover (Bilotta *et al*., 2007; Hubbard *et al*., 2004). Careful management of stocking rates and the timing of grazing events are key to minimising the potential impact of extended grazing periods on water quality. To elucidate the impact of extended grazing regimes on soil structure and nutrient export, the aim of this study was to investigate, under the current codes of good agricultural practice, (1) the cumulative impact of grazing in early spring and late autumn on soil structure and nutrient export from grazed grasslands, (2) the impact of different grazing intensities on nutrient export in overland flow in early spring, directly after a grazing event.

MATERIALS AND METHODS

The impact of grazing on nutrient export in overland flow was examined in two studies at AFBI Hillsborough, which focused on the two objectives previously outlined:

Experiment 1: Evaluation of the cumulative impact of grazing on nutrient export in overland flow at the end of the grazing season

Experiment 2: Evaluation of the impact of grazing intensity during the spring period, on soil associated nutrient losses in overland flow

A description of the study site for Experiment 1 was provided earlier in Section 3. Experiment 2 was carried out at a separate location to the main study, and a description of the study site is given below. Rainfall simulation was the main methodology used in both experiments; a general description of this methodology is given below. For Experiment 1 and 2 daily rainfall, wind speed, air temperature, humidity, sunshine hr/day were recorded at the Meteorological site at Hillsborough, located 1 km from both the experimental sites.

Site Description Experiment 2

The experimental site (35 x 38 m) was located on an undrained drumlin hill slope with a slope of approximately 6.5% and a northerly aspect. The soil type was a Brown Earth clay-loam soil (36% sand, 38% silt, 26% clay) overlying Silurian Shale. The soil had a pH of 5.5 and an olsen P, extractable potassium, magnesium, and sulphur content of 39.2, 284, 233, and 14.9 mg/l, respectively. The study site had a Hydrology of Soil Type (HOST) classification of 24 which is indicative of poorly drained soils with a high capacity for overland flow generation. This is the most common HOST classification in NI accounting for 46% of the land area (Cruickshank, 1997). The field was reseeded in 2002 with perennial ryegrass (*Lolium perenne*) and was grazed by dairy cows during each subsequent year from February - October. Urea fertiliser was applied at a rate of 60 kg N/ha three weeks prior to the start of the experiment in February 2010. Average annual rainfall and duration of the growing season recorded at the site was 890 mm and 254 days for the periods 1971 - 2000 and 1951 - 1990, respectively (Cruickshank, 1997).

Experimental design experiment 1

Experiment 1 was undertaken on the WinterCalf and SpringCalf grazing treatments previously detailed in section 3, with additional 12 m x 1.5 m exclusion plots established in the centre of each plot. The exclusion plots were included in the study to provide an untrampled (UT) treatment for comparative purposes. There were 4 untrampled sub-plots within each of the WinterCalf and SpringCalf grazing treatments. Cattle were prevented from trampling these areas by erecting an electric fence around the plots. Although cattle could graze around the edges of the exclusion plots, these plots were cut as necessary throughout the study period.

The WinterCalf treatment had a mean 'start of turnout' date of 28 March, and were offered 5.0 kg concentrate/cow/day throughout a 214-day grazing season. In contrast system SpringCalf had a 'start of turnout' date of 17 February, and were offered between 1.0 and 2.0 kg concentrate/cow/day throughout a 260-day grazing season. Maximum reliance on grazed grass was a key objective of the latter system, and as such, subject to soil conditions and herbage availability, cows commenced grazing as early as possible in the spring and were allowed to graze as late as possible in autumn. Mean stocking rates during the grazing season were 5.1 and 4.3 cows per/ha with WinterCalf and SpringCalf, respectively.

Rainfall simulations were carried out on the WinterCalf, SpringCalf and UT treatment plots over a two-day period in February 2010 and 2011 prior to grazing commencing at the site. The same grazing plots were used in both years but the location of the rainfall simulation sub-plots within the grazing plots was randomly selected each year. To provide a post-grazing comparison, simulations were also undertaken in late October 2010 and 2011, after the final grazing of the experimental plots had taken place. The aim of conducting rainfall simulations pre- and post-grazing was to determine the accumulative impact of the treatments over a complete grazing season.

Experimental design experiment 2

Sixteen plots, each measuring 3.0 x 7.0 m, were established at the site in a four block (Blocks A, B, C, and D) randomised block design (Figure 3.1). The boundary of each plot was marked by triple strand electrified fencing while a 1.0 m wide grass buffer strip was located upslope of each plot to prevent the contamination of plots from upslope areas. Four treatments were examined in the experiment, with each treatment replicated four times (Table 3.1). Each treatment comprised of two short term ‘grazing events’, which took place on 23 February 2010 (G1) and on 6 April 2010 (G2). Treatments during G1 were: ungrazed (UG-), lightly grazed (LG-) or heavily grazed (HG-), while the fourth treatment also remained ungrazed (UG-). During G2 the first three treatments were grazed to a common grazing intensity (-G), while the fourth treatment again remained ungrazed (UG-UG).

Prior to G1 and G2, herbage within the experimental site (excluding the treatment plots and buffer strips) was cut to ensure that cows focused their grazing activity within the treatment plots during each grazing event. During G1 and G2, ten lactating Holstein Friesian dairy cows (average live weight 650 kg) were given access to the experimental site at approximately 10.00 hr, with cows not having had access to food during the previous two-hr period. Cows were fitted with excreta collection bags using a harness system, thus preventing contamination of the site. During G1 cows had access to treatment plots LG- and HG- (plots in Blocks A and C from the upslope side, and to plots in Blocks B and D from the down slope side). Cows grazed these plots until a residual sward height of approximately 5.0 cm had been achieved, but before sward/soil surface damage became apparent. This grazing period lasted for 90 minutes, with cows then removed from the plots, and placed in a cubicle house without access to food. The ten cows were returned to the experimental site at approximately 13.00 hr on the same day, but were given access to the treatment HG- plots only. Cows were allowed to graze for a further 90 minutes, before they were removed and given access to silage.

During the second grazing event (G2; 6 April) cows had access to the UG-G, LG-G, and HG-G plots for a single period of 120 minutes, with cows removed when all plots had been extensively grazed, but before cows began to lie down.

Overland flow was simulated on all treatments at two days (RD2) and sixteen (RD16) days post-grazing using rainfall simulators as shown in Figure 3.2.

Soil physical and chemical variables

In both Experiments 1 and 2 three replicate soil cores were taken from each treatment plot, pre- and post-grazing, for the determination of bulk density, particle density, field capacity at 5 kPa and total porosity of the soil at 0 - 5 cm using the methods of Hall *et al*. (1977). Resistance to penetration was also measured using a digital penetrometer (Eijkelkamp Agrisearch Equipment), with ten replicates taken within each treatment plot pre- and post-grazing. The presence of stones in the soil matrix at both experimental sites made it difficult to obtain accurate measurements of resistance to penetration below 20 cm in the soil profile.

In Experiment 2 damage to the soil surface was also measured within a 1.0 m^2 quadrant in all grazed plots following both grazing events. The length and breadth of each hoof print or area of disturbed soil within the 1.0 m^2 quadrant was measured. Each imprint or area of exposed soil was also assigned one of four shapes (circular, oval, triangular or rectangular). A crude estimation of the total area of disturbed soil within each plot was then obtained by using the dimensions and shapes of each hoof print to calculate the total area of disturbed soil. This approach to measuring soil damage was chosen as it provided a simple and low-cost method of analysing soil disturbance on each plot.

In both experiments composite soil samples were taken from each treatment at the time of each rainfall simulation to a depth of 7 cm, with each sample being analysed for a range of soil parameters. Soil samples were air dried at 30^{o}C overnight and then analysed for bicarbonate-extractable inorganic P (Olsen-P) using a soil to Olsen reagent ratio of 1:20. Additional chemical analyses carried out on soil samples from Experiment 1 included water extractable P, Oxalate P, pH and organic matter. Water extractable P concentrations were determined spectrophotometrically at 880 nm following extraction at a ratio of 10:1 deionised water-to-soil. Oxalate extractable P, Oxalate extractable Fe, Oxalate extractable Al were determined using ICP-AES following extraction with ammonium oxalate. Soil pH was determined using a pH probe and a 2.5:1 ratio of deionised water-to-soil, while organic content was determined using the loss of ignition test at 850^{o}C.

Soil moisture contents, prior to rainfall simulation were recorded during both experiments. In Experiment 1, this was done using the gravimetric soil moisture method. The sample was weighed within 4 hr of collection from the field, and oven dried at 105°C for 48 hr. The difference between the pre- and post-oven drying sample provide the gravimetric moisture content of each sample which was then converted to volumetric soil moisture using the bulk density measurement taken from each plot on the same date. In Experiment 2 a HH2 Delta-T soil moisture probe was used (Delta-T Devices Ltd, Cambridge, UK).

Rainfall simulation

In both Experiments 1 and 2, 0.5 m^2 subplots within each treatment plot were hydrologically isolated from overland and shallow sub-surface flow using stainless steel surrounds placed 0.05 m into the soil. Stainless steel overland flow collection trays (0.5 x 0.1 x 0.1 mm) were inserted at the down-slope end of each sub-plot. The overland flow collection trays were connected to a two litre high density polyethylene plastic collection bottle by a 0.5 m length of braided PVC pipe buried underground. Collection trays and the stainless steel surrounds were inserted after each grazing event and 24 hr before the rainfall simulations.

In Experiment 1, overland flow collection trays and surrounds were removed following rainfall simulation at the start of the grazing season and re-inserted at the end of the grazing season prior to carrying out the post-grazing rainfall simulations. For Experiment 2, following the rainfall simulation on RD16 of the first grazing (G1), overland flow collection trays and surrounds were removed and the holes which had been dug to accommodate the overland flow collection trays, pipes and collection bottles were in-filled with soil. Exposed soil associated with the in-filled holes was covered with small pebbles to prevent contamination of the plots by soil during trampling at the second grazing event (G2).

Amsterdam rainfall simulators, as described by Bowyer-Bower and Burt (1989) were used in both Experiment 1 and 2 (Figure 3.3). This simulator is designed to form droplets of median diameter 2.3 mm, spaced 30 mm apart over the 0.5 m^2 simulator area. Wire mesh (3 mm spacing) intercepted falling water droplets coagulating some droplets and dispersing others to create a larger variation in drop sizes, similar to that of natural rainfall. During overland flow days on both experiments the two simulators ran simultaneously and were alternated between treatments to ensure there was no bias attributable to either of the simulators. When rainfall simulations were being undertaken, wooden boards were placed along the sides of the rainfall simulators to act as a wind shield and prevent water droplets being blown outside the plot boundaries. Rainfall was simulated at a rate of 40 mm/hr in both Experiments 1 and 2, with this rainfall intensity having a return period of greater than one in 50 years for NI (Cruickshank, 1997). This high rainfall intensity was selected to ensure overland flow was achieved on each of the plots, and is comparable to previous work carried out on these soils by O'Rourke *et al*. (2010). On each occasion time taken to generate overland flow was recorded, with thirty minutes of overland flow being collected thereafter. In Experiment 1, two 15-minute composite samples were collected while in Experiment 2 three 10-minute composite samples were collected during the 30 minutes of overland flow. The volume of overland flow was recorded for each composite sample. Water used in simulations was passed through a DC9 general deionising cylinder (Purite Limited) to reduce its P concentration. The cylinder delivered deionised water with an average dissolved reactive phosphorus (DRP) and nitrate concentration of 19.5 μg/l and 22.5 μg/l, respectively.

Water quality analysis

Water samples collected in both Experiment 1 and 2 were refrigerated at 3°C within 4 hr of sampling and analysed for soluble reactive P (SRP), total soluble P (TSP) and total P (TP) within 24 hr of sampling. Samples for SRP, and TSP were filtered through a 0.45 μm Millipore filter before analysis. Soluble reactive P was determined using the ascorbic acid reduction technique as described by Murphy and Riley (1962). Oxidative digestion with potassium persulphate and sulphuric acid under high pressure saturated steam (1 kPa and 121°C) conditions for 30 minutes using an autoclave (Phoenix Desktop, Rodwell Scientific Instruments, Basildon, Essex) was used to convert TSP and TP to SRP (Eisenreich *et al*., 1975). Particulate P (PP) was calculated as the difference between TP and TSP. Dissolved unreactive P (DUP) was calculated from the difference between TSP and SRP.

Samples for nitrite, total oxidisable nitrogen (TON) and ammonia were frozen at -21°C and analysis completed within two months of sampling. For nitrite, TON and ammonia determination samples were filtered through a 0.45 μm filter (MF-Millipore, Billerica, MA) before colorimetric analysis (Scheiner, 1976) using a Bran and Lubbe continuous flow analyser. Suspended sediment concentrations in the overland flow samples were determined under suction filtration. A known volume of sample was passed through GFC Millipore filters. The filters were oven dried at 105°C and ashed at 550°C, both for 24 hr. The pH and conductivity of the samples were measured using standard electrodes.

Data analysis

In Experiment 1, all data were tested for normality in both its original and transformed states and found not to be normally distributed. As such non-parametric tests were selected for data analysis. Flow-weighted mean nutrient concentrations and total nutrient export quantities were calculated for each rainfall simulation event by combining the nutrient concentrations and quantities of overland flow recorded from each of the 15-minute collection periods. The data from year 1 and year 2 were combined for analysis, with differences in nutrient concentrations, nutrient loads and soil physical properties averaged over two grazing seasons. This was done to increase the replication and confidence in the results obtained. Differences between the treatments for all variables were tested for using the Kruskal-Wallis One Way Analysis of Variance on Ranks and where differences were found all Pairwise Multiple Comparisons were conducted using either Student-Newman-Keuls Method or the Dunn's method depending on whether there were missing values in the dataset. Comparisons were made between WinterCalf and SpringCalf treatment pre-grazing and subsequently between the WinterCalf, SpringCalf and UT treatments post-grazing. No comparison between pre- and post-grazing were included in the analysis as differences observed may be due to seasonal factors.

In Experiment 2, data were analysed using Genstat version 12.1 (VSN International, 2008, UK). Flow-weighted mean nutrient concentrations and total nutrient export quantities were calculated for each rainfall simulation event by combining the nutrient concentrations and quantities from each of the 10-minute fractions collected. Plot nutrient values were taken as the average value from the two rainfall simulation areas in each plot. Initially nutrient data for each overland flow day were analysed individually using a one-way Analysis of Variance (ANOVA). Following this overland flow and nutrient data for each grazing event were analysed separately using a repeated measures analysis (ReML) and fitting an antedependence order 1 correlation model. Cumulative analysis of both grazing events was then undertaken, again using ReML. Soil and grazing parameters at both grazing events were analysed together using a two-way ANOVA.

RESULTS

Experiment 1

Grazing treatment had no impact on the concentrations of nutrients or sediment recorded in overland flow in the WinterCalf, SpringCalf and UT treatment (Table 3.2). In contrast, grazing had a significant impact on soil structure and the generation of overland flow (Table 3.3) which in the case of the SpringCalf treatment resulted in significant differences in the NH_4 NO_2, TSP, and TP loads exported when compared to the UT treatment (Table 3.4). Both the grazed treatments (WinterCalf, SpringCalf) had significantly greater bulk density (P<0.001) and a lower total pore space (P<0.001) than the UT treatment. However, there was no significant difference between resistance to penetration in the two grazing treatments compared to the UT treatment (Table 3.3). When the average resistance to penetration for both grazed treatments was compared to the UT treatment a significant difference was observed (P<0.05). A comparison of resistance to penetration at 1 cm intervals down the soil profile demonstrated that the difference between the grazed and UT plots increased with depth down to 15 cm, from 0.06 mPa at the soil surface to 0.4 mPa at 15 cm depth (Figure 3.4). The biggest difference in resistance to penetration occurred between 8-10 cm (0.42-0.44 mPa) indicating that grazing had the largest impact on compaction at this depth.

The change in soil structure in both the WinterCalf and SpringCalf treatments resulted in an increase in the volume of overland flow generated during rainfall simulation events, with a 54% and 71% difference in overland flow volume from the WinterCalf and SpringCalf treatments, respectively. The change in soil structure due to grazing is also evident in the significant difference (P<0.05) in bulk density and total pore space between the grazed and the UT treatments (Table 3.3) However, the observed increase in air capacity and overland flow volume was only significant for SpringCalf

(*P*<0.001) treatments (Table 3.3). There was also a 55% and 69% increase in the time taken to initiate overland from the WinterCalf and SpringCalf treatments compared to the UT treatment, respectively, but as with overland flow, the difference was only significant in the case of the SpringCalf treatment (*P*<0.05). The impact of the time taken to initiate overland flow on the resulting total P export over 30 minutes is illustrated in Figure 3.5. As a result of an increase in overland flow discharge from both the WinterCalf and SpringCalf treatments post-grazing, these treatments exported greater loads of nutrients and SS than the UT treatment, however these differences were only significant for the SpringCalf treatment (*P*<0.05). There was no significant difference in nutrient or sediment export between the WinterCalf and SpringCalf treatments, although on average the SpringCalf treatment tended to have higher nutrient and sediment loads than the WinterCalf treatment.

Although Tables 3.2-3.4 show results from the analysis of the combined data over two grazing seasons, analysis of the data on an annual basis confirmed that the main differences between the treatments were due to changes in soil structure with significant differences on an annual basis in bulk density, total porosity and air capacity in the grazed plots (*P*<0.05) when compared to the ungrazed plots at the end of both grazing seasons.

Experiment 2

Following G1, hoof-print density and total trampled area was significantly greater (*P*<0.05) on the HG-G treatment plots than the LG-G treatment (Table 3.5). Mean hoof-print density (20.9 hp/m^{-2}, *P*<0.001), hoof-print depth (5.4 cm, *P*<0.001), and total area trampled (37.6%, *P*<0.001) was greater following G1 in February compared with the mean of the grazing treatments following G2 in April (7.8 hp/m^{-2}, 2.3 cm, and 18.1%, respectively). In contrast following G2 there was no significant difference in the level of trampling between the three grazing treatments. In addition there was no significant effect (*P*>0.05) of either grazing event or grazing treatment on either pre- or post-grazing soil bulk density, macroporosity content, or resistance to penetration within the top 20 cm of soil (Table 3.5)

Grazing had a limited impact on the dissolved P factions recorded in Experiment 2 after both G1 and G2 (Table 3.6). In contrast following G1, at both RD2 and RD16 PP concentrations in overland flow from treatment HG-G were significantly higher (*P*<0.05) than from treatment LG-G, while PP concentrations in overland flow from treatment LG-G were significantly higher (*P*<0.05) than from the ungrazed (UG-UG and UG-G) treatments. Particulate P concentrations in overland flow were on average 20% higher at RD16 than at RD2 (*P*<0.01). Total P concentrations in overland flow at RD2 (G1) were significantly greater (*P*<0.05) from the grazed (LG-G and HG-G) treatments than the UG-UG treatment. Total P concentrations in overland flow at RD16 (G1) were significantly different (*P*<0.05) between the three levels of grazing intensity and increased in the order UG-, LG-, and HG- with no significant difference (*P*>0.05) between the UG-G and UG-UG treatments. Total P concentrations in overland flow following G1 did not differ between the two overland flow day events (*P*>0.05) (Table 3.6).

Following G2, PP concentrations in overland flow at RD2 were significantly higher with the HG-G treatment, compared to the ungrazed treatments (*P*<0.05), while PP concentrations with the LG-G treatment did not differ from the ungrazed treatments (*P*<0.05). Particulate P concentrations were 90% lower at RD16 compared to RD2 (*P*<0.001), with treatment having no significant effect (*P*>0.05) on PP concentrations at RD16. Following G2, RD2, TP concentrations in overland flow were significantly higher (*P*<0.05) from the three grazed treatments (UG-G, LG-G, and HG-G) than from treatment UG-UG, with treatment HG-G exhibiting the highest TP concentrations in overland flow. Likewise total P concentrations in overland flow following G2, decreased by 83% between RD2 and RD16 (*P*<0.001) with treatment having no effect on TP concentrations at RD16.

At RD2, following G2, TP exports from the grazed treatments (UG-G, LG-G, and HG-G) were significantly higher (*P*<0.001) than during any of the other three overland flow day events throughout the experiment. In addition, TP export from these three grazed treatments at RD2 (G2), were significantly greater (*P*<0.05) than from the UG-UG treatment (Figure 3.6). Total P exports from the UG-UG treatment were not significantly different across the four overland flow events. There was no significant effect of treatment on TP export at either G1 (RD2 and RD16) or G2 (RD16). Across both grazing events PP accounted for the majority of TP exported, accounting for 62.6 and 82.9% of TP lost at RD2 and RD16, respectively (G1), and 93.2 and 57.1% of TP at RD2 and RD16, respectively (G2). At each RD event throughout the experiment, the HG-G treatment exhibited the highest proportion of PP as a percentage of TP export.

DISCUSSION

Although the two grazing treatments in Experiment 1 had approximately the same total number of grazing hr in each year, the key difference between these treatments was that the SpringCalf treatment started 44 and 20 days before grazing on the WinterCalf plots in Year 1 and 2, respectively. As such grazing in the SpringCalf treatment was being carried out during periods of higher antecedent soil moisture conditions and with a higher risk of grazing coinciding with rainfall. In addition there were differences in the cows used in the two treatments with the average weight of the Holstein (WinterCalf treatment) and Jersey crossbred cows (SpringCalf treatment) 540 and 478 kg, respectively. However, the Jersey crossbred cows also had a smaller hoof area, indicating that the pressure applied to the soil surface by the Jersey crossbred cows was at least equal to, if not greater than, the pressure applied by the Holstein cow. Greenword and McKenzie (2001) report a range of static pressure from cattle depending on weight and hoof area. A 530 kg cow with a hoof area of approximately 400 cm^2, exerted a static pressure of 133 kPa, while a lighter cow of 380 kg with a hoof area of 264 cm^2 exerted a pressure of 144 kPa at the soil surface. The pressure applied at the soil surface significantly increasing when a cow was walking. Difference in the feeding behaviours of Holstein and Jersey crossbred cows may also have impacted on the pressure both breeds applied to the soil surface. Although both breeds have similar intakes at grass, the Jersey crossbred cows have smaller mouths (i.e. smaller bites), resulting in them grazing for longer to achieve the same intake of grass (Vance *et al*., 2012). As a result of the longer grazing time of the Jersey cross in the SpringCalf treatment, in addition to its smaller hoof area and earlier turn-out date, there was a greater risk of soil compaction in the SpringCalf treatment than the WinterCalf treatment.

However, the lack of a significant difference in nutrient and sediment loads or concentrations between the SpringCalf and WinterCalf treatments suggest that if best practice is adhered to, adopting a system with a much greater reliance on grazed grass (longer grazing season, lower concentrate feed levels), does not significantly increase the risk posed to water quality at this site. However, as the difference in nutrient export was significant from the SpringCalf, but not the WinterCalf treatment, in comparison with the UT treatment, this indicates that SpringCalf actually did have a slightly greater impact on soil structure and nutrient export than the WinterCalf treatment.

The lack of a significant difference between the two grazing treatments was confirmed when the data from Experiment 1 were analysed on an annual basis (not presented). As the majority of compaction can occur during one grazing event (Mulholland and Fullen, 1991), any potential difference between the two grazing treatments may have been masked by one mismanaged grazing event (e.g. one 24-hr period during which grazing intensity was too high for the prevailing soil moisture conditions). In addition, this experimental site had been grazed over an extended period of time (over 30 years) prior to the study described in this paper and examination of the soil structural data at the start of the study would suggest that the site was already compacted to some degree. Although this level of compaction is likely to be the norm within intensive dairy grazing systems, caution is required in transferring the findings of this study to a scenario where soils are less compacted at the start of the grazing season.

A number of authors have identified a negative relationship between grazing intensity and soil hydraulic conductivity, which is reflected in an increase in overland flow volume following treading (Drewry, 2003, Drewry and Paton, 2000, Pietola *et al*., 2005). This is also the case in this study, with the impact of grazing on soil structure influencing the magnitude of overland flow and nutrient export from the grazed treatments. There was, on average, a 62% increase in the volume of overland flow generated from the grazed plots compared to the UT plots. This increase in volume resulted in a corresponding average increase of 60% in the quantity of TP, TON and sediment exported from the grazed plots. It is likely that a decrease in the infiltration capacity of the soil caused by compaction was the main factor responsible for the increase in nutrient export observed in this study. Heathwaite *et al*. (1990) observed an 80% reduction in the infiltration capacity of a heavily grazed soil, resulting in 12 times more overland flow from grazed plots compared to ungrazed plots. Doody *et al.* (2010) found that although saturation excess overland flow accounted for a larger volume of overland flow from this soil type in Northern Ireland, infiltration excess overland flow occurred more frequently during the year, with 59% of overland flow events occurring on days when the volumetric soil moisture was at or below field capacity.

Macroporosity was not measured during this study, but as the changes in bulk density, total pore space, air capacity and resistance to penetration, between the grazed and UT plots, were small at 6%, 4%, 2% and 4.5%, respectively, it is likely that changes in macroporosity were also contributing to the observed differences in overland flow initiation and volume. Houlbrooke *et al.* (2009) suggested that macroporosity was the most sensitive soil physical parameter to compaction by grazing, with Drewry and Paton (2000) reporting that macroporosity was 70% higher on plots that had never been grazed, compared to grazed plots at the end of one grazing season. Kurz *et al*. (2006) observed that macroporosity, bulk density and resistance to penetration were the main structural parameters affected in a grazing study in southern Ireland, decreasing by 83% and increasing by 17% and 50%, respectively. Greenwood *et al*. (1998) reported a significant increase in soil unsaturated hydraulic conductivity after 2.5 years of zero grazing. The results indicated an increase in the number of soil pores with equilibrium cylindrical diameters of between 1.2 and 6.0 mm, a size class that is characteristic of macropores.

The cores taken for bulk density and porosity in this study were taken at 0-5 cm depth in the soil profile. However, grazing can also result in more persistent compaction below the soil surface layer, depending on soil type and conditions during grazing. Drewry and Paton (2000) found that the largest decrease in hydraulic conductivity between grazed and ungrazed treatments was at 5-10 cm in the soil profile, with macroporosity and bulk density changing by 56% and 10% at 5-10 cm, compared to 45% and 1.2% at 0-5 cm, respectively. Mulholland and Fullen (1991) reported that grazing resulted in a very compacted zone at 7-10.5 cm in the soil profile which impeded drainage. In Experiment 1 comparison of the resistance to penetration at 1 cm intervals down the soil profile demonstrated that grazing had the largest impact on compaction at a depth of 8-10 cm. This concurs with the findings of Doody *et al*. (2010) who reported a peak in both resistance to penetration and bulk density at a depth of 10 cm in the soil profile following continuous grazing by beef cattle over a 20-

year period, at an experimental site close to the location of the current study and with the same soil type. Compaction below the soil surface is likely to be more persistent than within the top 5 cm of the soil profile as the action of roots, soil biota and the constant wetting and drying cycle will improve the soil structure close to the soil surface following the cessation of grazing (Drewry, 2006). Drewry (2006) suggested that natural recovery of soil from the damage caused by grazing could be quite rapid at 0-5 cm in the soil profile due to the burrowing action of macro-invertebrates during the decomposition of deposited dung.

In Experiment 1, following the winter months between the Year 1 and Year 2 grazing season, when cows were not permitted to graze the plots, the differences in soil structure and hydrology in the grazed and UT plots were no longer evident suggesting that at this site for that year, the accumulative effects of grazing were only evident over a single grazing season. However, data were only available for one winter period in this study and there may be annual variation in the potential for recovery depending of biological activity, freeze-thaw cycles and wetting-drying cycles (Greenwood and McKenzie 2001). Although Drewry *et al*. (2004) observed that in New Zealand natural recovery from compaction was greatest during the summer and autumn period, recovery during winter also occurred with total porosity increasing by on average 2.5%, compared to 4.4% in summer. The soil at the current study site had an organic matter content of 15%, with Drewry (2006) reporting that high organic carbon content of soil was a significant factor in the recovery of soil from compaction. The period required for full recovery of the soil from compaction caused by grazing is unclear and varies with soil type and climate condition (Drewry, 2006). For example, Greenwood *et al*. (1998) observed that following 2.5 years of cattle being excluded from grassland plots, the hydraulic conductivities and bulk density were similar to those measured in grassland plots ungrazed for 27 years, while Drewry (2006) reported on a case where full recovery was incomplete after 16 years of zero grazing. Whether the soil at the current study site had fully recovered from compaction following three months without livestock grazing is unclear and requires future investigation.

When the results of Experiment 1 and Experiment 2 are considered collectively, they support the hypothesis that the impact of grazing on soil structure is cumulative over a grazing season. While there was a significant difference in the time taken to initiate overland flow in the 2nd grazing event in Experiment 2 (data not presented), with an average increase in the time of 181 seconds on the ungrazed plots, there was no corresponding difference in overland flow volume recorded. In contrast the average time taken to initiate overland flow at the end of the grazing season in October (Experiment 1) was 649 seconds in the grazed plots and 1706 seconds in the UT treatment, with significant differences in both the time to initiation of overland flow and volume between the SpringCalf and UT ($P<0.05$). Greenwood *et al*. (1998) concluded that the impacts of grazing were cumulative but that the resulting compaction reached steady-state overtime (Greenwood and McKenzie, 2001). Once compaction has occurred the bearing strength of the soil increases thereby reducing the likelihood of further compaction occurring (Greenwood *et al*., 1998). However, depending on conditions at the time of grazing, the majority of compaction can be caused by the first treading action of the grazing animals (Bilotta *et al*., 2007). Mulholland and Fullen (1991) found that infiltration capacity decreased by 88% following the first treading on a saturated soil by an artificial hoof, while subsequent treading action decreased infiltration capacity by a further 9%.

The importance of grazing intensity in early spring was clear from the difference in concentration of P in overland flow following grazing events in Experiment 2. Increasing grazing intensity was associated with an increase in PP (and total P) concentrations in overland flow at RD2 and RD16 following G1, and this was likely due to the combined effects of soil disturbance and vegetation removal. The hooves of grazing animals cause vertical and horizontal displacement of soil through smearing, skid marks, and hoof print deformation, thus exposing the soil to rainfall. Once exposed, soil slaking and dispersion processes caused by raindrops, facilitates the detachment of soil particles (and its associated PP), for transport via overland flow (McDowell *et al*., 2003). McDowell *et al*. (2003) identified a positive relationship between the density of hoof imprints on a grassland site and PP and TP concentrations in overland flow. However, concentrations of PP and TP in overland flow in the current experiment were approximately double those recorded by McDowell *et al*. (2003), despite a similar number of hoof imprints per m^{-2} in both studies. These differences between experiments may have been due to the lower rainfall intensity (15 mm/hr) and the use of an artificial cow hoof by McDowell *et al*. (2003). While an artificial hoof is able to replicate a similar downward force as a grazing animal, it is unable to replicate the horizontal movement of soil, which would have occurred in the current study.

Vegetation cover can also reduce PP loss in overland flow by intercepting rainfall, thus decreasing its kinetic energy and minimising the detachment of soil particles (Delpla *et al*., 2011). This was demonstrated by Sharpley (1985) who, using mesh screens of sizes 0.5, 1.0, 4.0 and 9.0 mm^2 to simulate different levels of vegetation cover, identified a negative relationship between screen size and the effective depth of interaction (EDI) of rainfall with the soil. The presence of a grass sward can further decrease soil erosion and hence particulate P loss by reducing overland flow velocity and causing the settling out and trapping of previous entrained soil particles (Prosser *et al*., 1995, Carroll *et al*., 2000).

The impact of the grazing treatments imposed at G1 was still evident 16 days later at RD16, and during G2 (RD2), with PP and TP concentrations in overland flow following the same trend as RD2 at G1. As residual herbage heights were similar in each of the three grazing treatments following G2, the trends in PP losses are likely to reflect the residual impact of soil damage, rather than the impact of herbage cover. The rate of soil recovery following grazing depends on factors such as level of soil damage (Drewry *et al*., 2008), frequency of, and time since grazing (Drewry and Paton, 2000), and sward type (Menneer *et al*., 2005). The process of soil recovery following grazing involves the aggregation of soil particles as a result of root development and biological activity, and this can lead to a reduction in nutrient concentrations in overland flow (Angers and Caron, 1998, Drewry, 2006). Following an intensive treading event, Drewry *et al*. (2003) noted that while soil hydraulic conductivity had recovered substantially after two weeks, macroporosity and herbage yield recovery occurred over a much longer timescale. Smith and Monaghan (2003) attributed a reduction in P concentrations in overland flow with increasing time since grazing during the spring, to soil recovery processes. However, the relatively short time period between grazing events in the current study is likely to have reduced the capacity for soil recovery. In addition, as soil recovery involves biological processes, these were likely to have been limited by low soil temperatures during the experimental period.

In other studies, elevated concentrations of dissolved inorganic (reactive) and organic (unreactive) P in overland flow often reflects excretal P returns from grazing animals, with their concentrations normally decreasing with time post grazing due to incorporation of manure deposits into the soil (Mundy *et al*., 2003; McDowell *et al*., 2007; Ebeling *et al*., 2002). However, the excreta collection bags worn by cows during Experiment 2 prevented the plots being contaminated by faeces, so that grazing treatment had no significant effect on either DRP or DUP concentrations at either RD2 or RD16 at either grazing event. Nevertheless, at G1 there was a trend for DUP concentrations from the grazed treatments to be higher than from the ungrazed treatments, with this perhaps attributed to the release of plant-associated organic P following the trampling and destruction of grass stems and roots. A similar effect has been observed by a number of authors (McDowell *et al*., 2003; Mundy *et al*., 2003; McDowell *et al*., 2007). Despite these trends, the results of the current experiment suggest that early spring grazing by dairy cows, even when associated with relatively intensive trampling, has little impact on dissolved phosphorus losses.

CONCLUSIONS

The finding of the current study would suggest that grazing has the potential to increase the frequency and volume of overland flow and nutrient export from similar soil types in Ireland. Where compaction occurs there is a greater risk of critical sources area of nutrient developing, especially in areas that are hydrologically connected to adjacent water bodies. In order to reduce nutrient export from grassland, grazing strategies should aim to minimise the impact on soil structure so as to maximise the time it takes to initiate overland flow. Increasing the risk of overland flow also increases the risk of incidental losses of nutrient following slurry application to grassland soils. In addition to reducing the rainfall and soil moisture thresholds at which overland will occur, compaction will reduce the rate at which slurry infiltrates the soil, resulting in it remaining on the surface of the soil for an extended period of time.

Where conditions are sub-optimal for grazing, it can result in an increase in TP export from grassland, predominately in PP form. While PP is not directly available to aquatic organisms upon entering a river system, PP stores in river sediments can become available over time and consequently can be a key driver of eutrophication.

In addition, the input of large quantities of sediment from grazed grassland can pose a significant threat to freshwater ecosystems.

Effective grazing management is also required to minimise the detrimental impact on soil structure, with compaction having significant consequences for the quantity of nutrients and sediment exported. As such allowing time for soil to recover from compaction could be considered as an effective mitigation strategy for reducing nutrient export from soil, particularly in areas where soil P is significantly above the agronomic optimum for grass production.

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Table 3.1: Summary of management at each of the two grazing events for Experiment 2

Treatment descriptor	Management at grazing 1 (G1)	Management at grazing 2 (G2)
UG-G	Ungrazed	Grazed
LG-G	Light Grazing	Grazed
HG-G	Heavy Grazing	Grazed
UG-UG	Ungrazed	Ungrazed

Table 3.2: Nutrient and sediment concentrations in overland flow from two grazing systems, WinterCalf (WC) and SpringCalf (SC). Results presented are for WC and SC prior to grazing in February (Pre-Grazing) and from these two systems and an untrampled (UT) treatment following the final grazing of the experimental plots in late October (Post-Grazing). All data presented are the average of years 1 and 2 combined

		Overland Flow Water Quality Variables															
Time of Measurement	Treatment	TON (mg/l)		NH_4 (µg/l)		NO_2 (µg/l)		SRP (mg/l)		TSP (mg/l)		TP (mg/l)		PP (mg/l)		SS(mg/l)	
		Mean	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*
Pre-Grazing	WC	0.31	0.06	268.8	33.1	21.6	10.61	95.1	6.5	133.7	8.2	462.9	73.7	329.2	274.4	141.3	19.8
	SC	0.14	0.03	347.3	55.1	11.4	1.4	89.1	11.9	143.9	13.6	695.4	155.1	551.6	556.7	252.8	46.5
Post-Grazing	WC	0.24	0.02	169.1	22.5	11.3	1.2	118.4	20.9	183.6	32.9	778.3	100.8	594.7	369.2	193.3	44.2
	SC	0.18	0.02	113.8	14.6	9.7	0.7	96.5	14.8	151.9	18.7	567.2	56.7	415.3	210.2	138.9	30.5
	UT	0.34	0.1	337.2	223.1	12.6	3.6	122.5	22.5	186.5	32.6	601.7	66.9	460.1	178.4	158.8	35.5

Note: Values in the same columns with different superscripted letters are significantly different. No superscript indicates no significant difference with any other value in a column. When assigning superscripts, valid comparisons were taken only as those within the pre-grazing treatment and those with the post-grazing treatments

Table 3.3: Changes in soil physical properties in three grazing treatments, Wintercalf (WC), Springcalf (SC) and Untrampled (UT). Results presented are for WC and SC prior to grazing in February (Pre-grazing) and from these two systems and UT treatment following the final grazing of the experimental plots in late October (Post-grazing). All data presented are the average of years 1 and 2 combined

		Soil Physical Properties											
Time of Measurement	Treatment	Bulk Density (g cm^{-3})		Total Pore Space (%)		Air Capacity (%)		Resistance to Penetration* (mPa)		Time to Overland flow Initiation (secs)		Overland flow Volume (ml^)	
		Mean	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*
Pre-grazing	WC	0.85	0.01	63.5	0.56	18.3	0.57	0.97	0.05	712	76.7	1425	150.8
	SC	0.85	0.02	63.6	0.92	18.1	0.61	0.92	0.04	600	89.1	1664	283.3
Post-grazing	WC	0.86[a]	0.01	62.9[a]	0.32	22.4	2.37	1.36	0.07	762	94.4	1062	208.7
	SC	0.89[a]	0.01	62.1[a]	0.33	16.63[a]	0.47	1.19	0.04	536[a]	58.5	1661[a]	210.4
	UT	0.82[b]	0.01	65.2[b]	0.23	22.21[b]	0.35	1.22	0.05	1706[b]	222.6	488[b]	118.8

Note: Values in the same columns with different superscripted letters are significantly different at a minimum of $P<0.05$. No superscript indicates no significant difference with any other value in a column. When assigning superscripts, valid comparisons were taken only as those within the pre-grazing treatment and those with the post-grazing treatments

***** Measured to a depth of 15 cm

^ Flow volume for 30 min of overland flow

Table 3.4: Nutrient and sediment loads exported in 30 min of overland flow from three grazing treatments, WinterCalf (WC), SpringCalf (SC) and untrampled (UT). Results presented are for WC and SC prior to grazing in February (Pre-grazing) and from these two systems and an Ungrazed treatment following the final grazing of the experimental plots in late October (Post-grazing). All data presented are the average of years 1 and 2 combined

		Overland flow Water Quality Variables															
Time of Measurement	Treat-ment	TON (g/ha)		NH_4 (g/ha)		NO_2 (g/ha)		SRP (g/ha)		TSP (g/ha)		TP (g/ha)		PP (g/ha)		SS (g/ha)	
		Mean	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*Mean*	*SE*	*SE*	*SE*
Pre-grazing	WC	9.77	3.73	6.7	0.96	0.56	0.31	2.82	0.45	3.95	0.64	15.15	4.47	11.2	3.9	4507.8	1173.9
	SC	5.24	1.95	8.57	1.62	0.32	0.04	2.64	0.53	4.44	0.88	23.17	6.01	18.73	5.35	9088.6	2450.9
Post Grazing	WC	4.78	0.72	3.21	0.58	0.22	0.04	2.37	0.65	3.58	0.85	15.36	3.62	11.78	3.43	3605.8	1172.6
	SC	5.66	1.24	3.55[a]	0.52	0.34[a]	0.07	3.04	0.62	4.87[a]	0.84	18.33[a]	3.67	13.46	3.45	4908.2	2033.3
	UT	2.6	0.56	1.69[b]	0.4	0.11[b]	0.03	1.32	0.43	1.89[b]	0.52	6.67[b]	2.0	5.04	1.44	1575.1	446.7

Note: Values in the same columns with different superscripted letters are significantly different at a minimum of $P<0.05$. No superscript indicates no significant difference with any other value in a column. When assigning superscripts, valid comparisons were taken only as those within the pre-grazing treatment and those with the post-grazing treatments

Table 3.5: The effect of grazing treatment on grass, soil and trampling parameters measured at each grazing event

	Grazing 1				Grazing 2				Grazing x Treatment	
	UG-G	LG-G	HG-G	UG-UG	UG-G	LG-G	HG-G	UG-UG	SE	SIG
Hoof-print density (hoof-prints m^{-2})	-	18.3[b]	23.5[c]	-	6.8[a]	7.5[a]	9.0[a]	-	1.67	***
Average hoof-print depth (cm)	-	5.1[b]	5.6[b]	-	1.9[a]	2.9[a]	2.2[a]	-	0.60	***
Trampled area (%)	-	29.6[b]	45.5[c]	-	14.8[a]	18.4[a]	21.0[a]	-	3.53	***
Pre-grazing bulk density (g cm^{-3})	0.68	0.70	0.72	0.69	0.66	0.73	0.71	0.68	0.037	NS
Post-grazing bulk density (g cm^{-3})	0.69	0.70	0.72	0.67	0.68	0.69	0.68	0.70	0.026	NS
Pre-grazing macroporosity	0.14	0.13	0.11	0.13	0.15	0.12	0.12	0.13	0.012	NS
Post-grazing macroporosity	0.13	0.11	0.11	0.15	0.11	0.12	0.12	0.12	0.013	NS
Pre-grazing RP‡ (kPa)	1.61	1.73	1.80	0.167	1.64	1.78	1.81	1.69	0.124	NS
Post-grazing RP (kPa)	1.46	1.39	1.49	1.56	1.44	1.37	1.55	1.45	0.091	NS

† Values within each row with the same letter are not significantly different (P>0.05),
‡ Resistance to penetration
UG = ungrazed, LG = light grazing, HG = heavy grazing, G = grazed
*** P<0.001

Table 3.6: The effect of treatment and grazing event on flow-weighted mean dissolved reactive phosphorus (DRP), dissolved unreactive phosphorus (DUP), particulate phosphorus (PP), and total phosphorus (TP) concentrations observed in overland flow 2 and 16 days post grazing

		Grazing 1				Grazing 2			
		SRP (mg l^{-1})	DUP (mg l^{-1})	PP (mg l^{-1})	TP (mg l^{-1})	SRP (mg l^{-1})	DUP (mg l^{-1})	PP (mg l^{-1})	TP mg l^{-1})
RD2	UG-G	0.23	0.17	0.39[a]	0.67[ab]	0.12	0.23	1.83[a]	2.03[b]
	LG-G	0.19	0.27	0.53[b]	0.81[b]	0.12	0.17	2.55[ab]	2.78[b]
	HG-G	0.24	0.22	0.72[c]	0.94[b]	0.14	0.17	3.25[b]	3.65[c]
	UG-UG	0.21	0.18	0.39[a]	0.65[a]	0.11	0.17	1.83[a]	1.19[a]
	SED	0.043	0.040	0.067	0.077	0.021	0.046	0.373	0.389
	SIG	NS	NS	**	*	NS	NS	***	***
RD16	UG-G	0.11	0.15	0.44[a]	0.59[a]	0.14	0.13	0.19	0.67
	LG-G	0.10	0.13	0.65[b]	0.79[b]	0.14	0.11	0.17	0.35
	HG-G	0.14	0.15	0.86[c]	1.04[c]	0.13	0.12	0.25	0.42
	UG-UG	0.10	0.11	0.45[a]	0.59[a]	0.19	0.13	0.22	0.35
	SED	0.024	0.024	0.072	0.066	0.026	0.016	0.079	0.088
	SIG	NS	NS	**	***	NS	NS	NS	NS
RD	SIG	***	***	**	NS	NS	**	***	***
Treatment x RD†	SIG	NS	NS	NS	NS	NS	NS	*	*

† Treatment x overland flow day interaction,
UG = ungrazed, LG = light grazing, HG = heavy grazing, G = grazed,
* *P*<0.05, ** *P*<0.01, *** *P*<0.001, NS = *P*>0.05

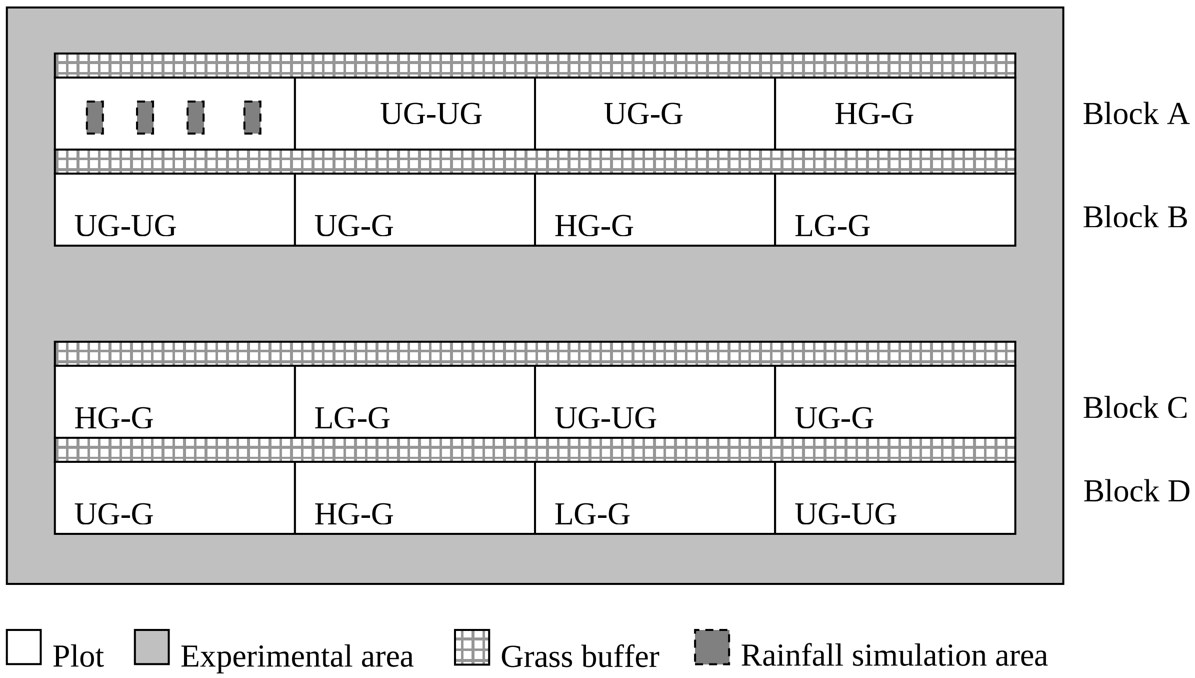

Figure 3.1: Layout of the experimental site and treatment plots for Experiment 2

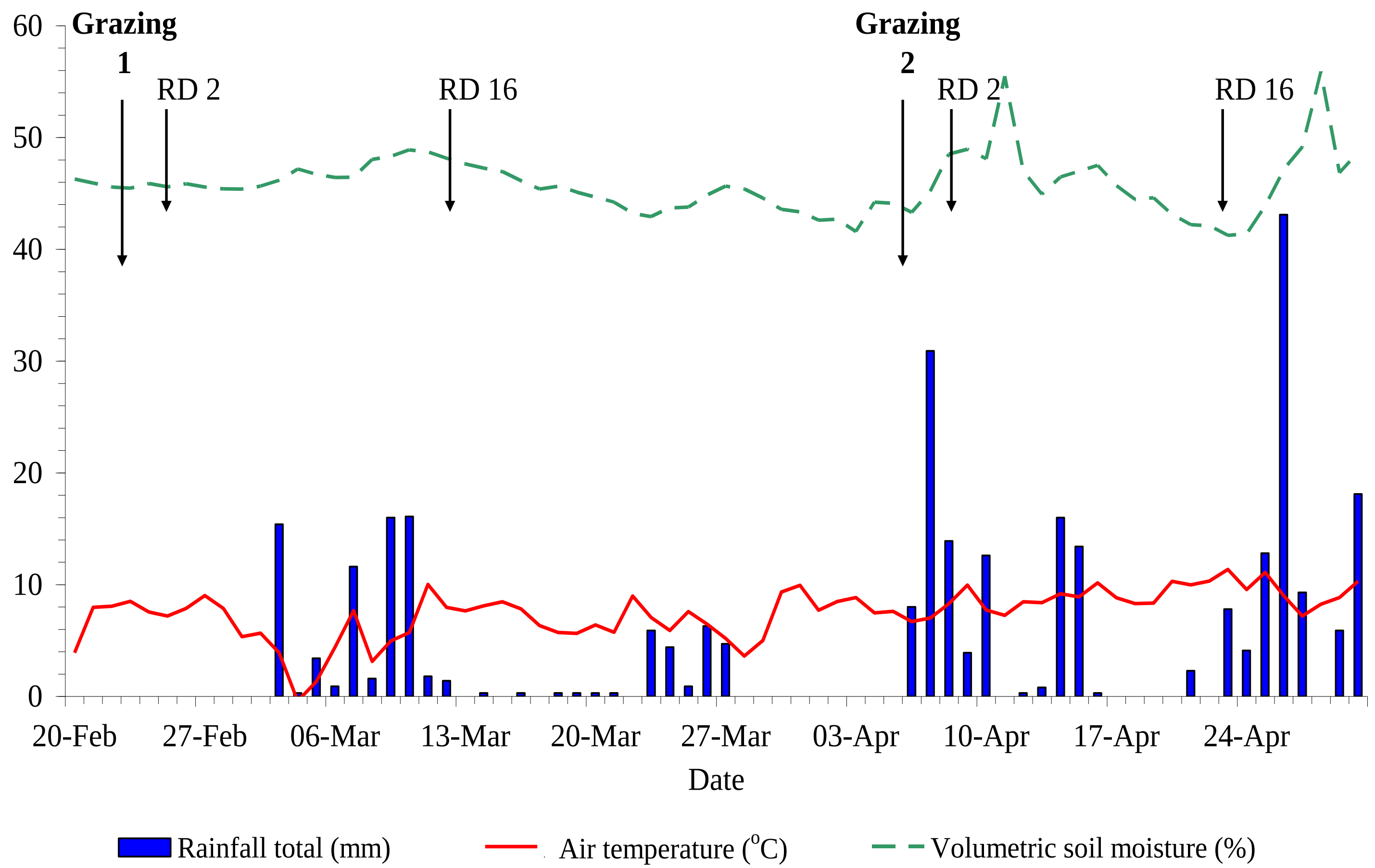

Figure 3.2: Daily rainfall totals, average air temperature, and average volumetric soil moisture content observed throughout the duration of the experiment

Figure 3.3: Rainfall simulation apparatus used to generate overland flow during Experiments 1 and 2

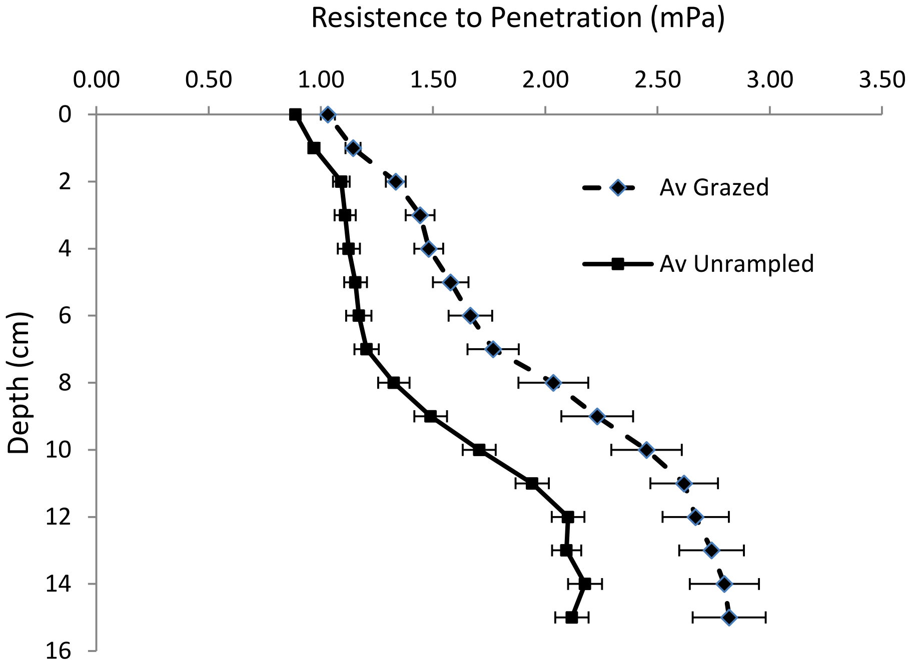

Figure 3.4: Changes in the average resistance to penetration (mPa) with depth following two grazing seasons for grazed and untrampled treatments (*P*<0.01)

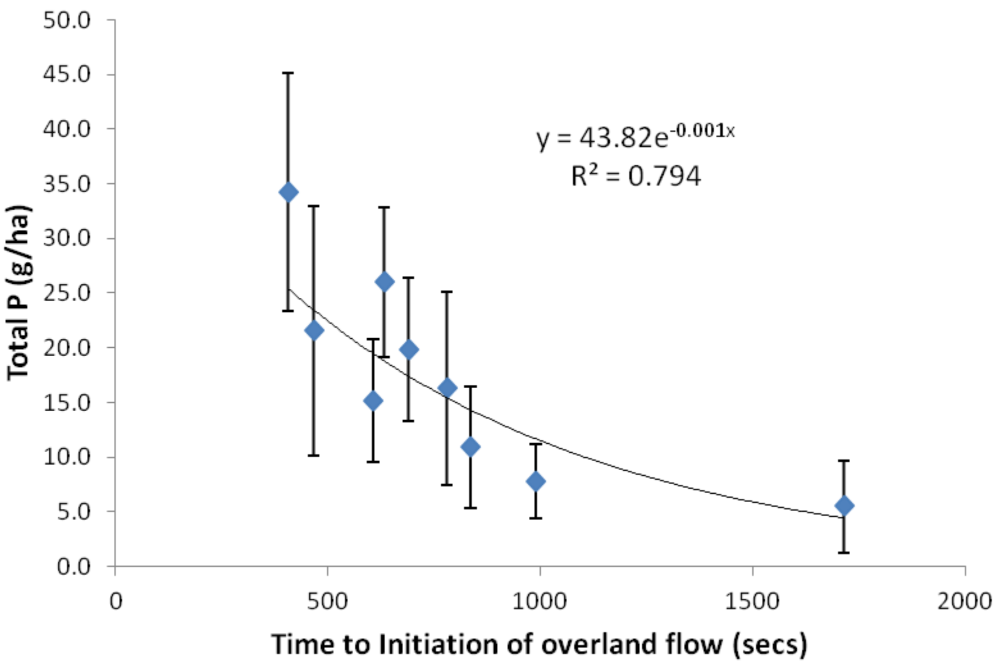

Figure 3.5: Relationship between the time taken to generate overland flow and the resulting load of Total P exported from a grazed grassland soil (*P*<0.01)

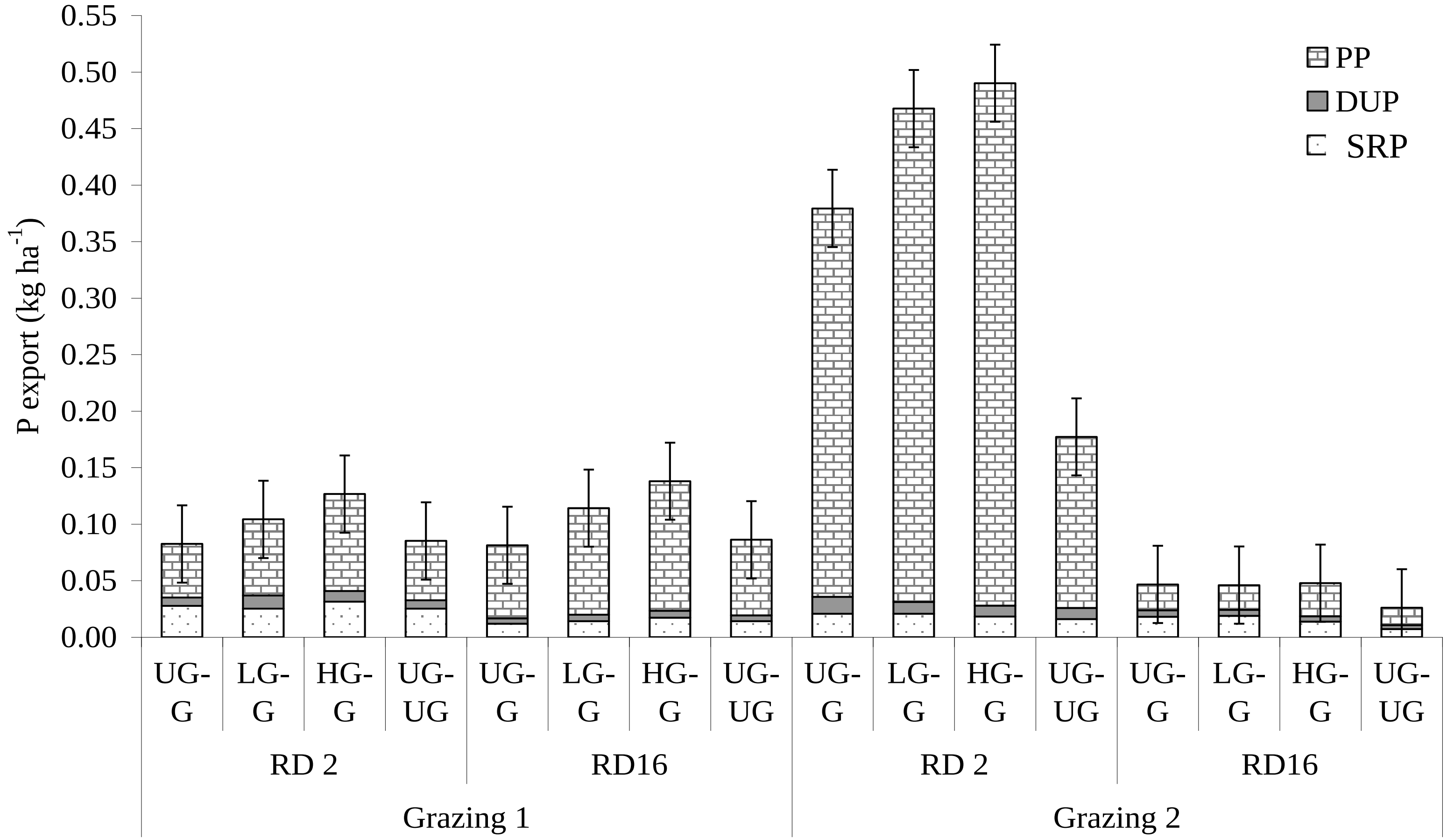

Figure 3.6: The effect of grazing event and treatment on the export of soluble reactive phosphorus (SRP), dissolved unreactive phosphorus (DUP), and particulate phosphorus (PP) at each overland flow event (UG = ungrazed, LG = light grazing, HG = heavy grazing, G = grazed) (RD2 = 2 days post grazing; RD16 = 16 days post grazing)

SECTION 4

Nitrous oxide emissions and nitrogen loss by leaching from three contrasting grassland-based milk production systems over two successive years

INTRODUCTION

Northern Ireland is bound by EU and UK legislation to reduce greenhouse gas (GHG) emissions by 20% by 2020 (taking 1990 as the base) and 80% by 2050. The Northern Ireland Executive has set a programme for government target to work towards a reduction of GHGs by at least 35% of 1990 levels by 2025. Agriculture's contribution to the total GHG budget is high in countries that have a high dependence on pastoral-based systems, such as New Zealand and Ireland, where the contribution of agriculture to total GHG emissions is 48% and 28%, respectively (UNFCCC, 2011). Nitrous oxide (N_2O) is one of the most potent GHGs having a Global Warming Potential approximately 300 times that of CO_2. In grazed grassland soils N_2O is mostly produced by denitrification, the sequential reduction of NO_3^- through NO_2^-, NO and N_2O to N_2 by denitrifying bacteria (Payne, 1973). Although N_2 is the end product of denitrification the process does not always proceed to completion and variable amounts of N_2O are produced. For denitrification to occur denitrifiers must have an electron acceptor (NO_3^-), an electron donor (carbon) and anaerobic conditions. The extent of anaerobiosis is determined either by an increase in soil moisture content (Davidson *et al*., 2000) by rainfall or metabolism of carbon (Parkin, 1987). Other factors such as pH (Simec and Cooper, 2002; Cuhel *et al*., 2010), temperature (Keeney *et al*., 1979), microbial population (Laughlin and Stevens, 2002) and management (Liu *et al*., 2007) can also affect denitrification and the proportion of N_2O and N_2 produced.

Nitrification is also an important environmental process as NH_4^+, which is held on the cation exchange complex, is oxidised to NO_3^-, and when in excess of plant uptake, can either be denitrified or as it is a free ion can be leached down the soil profile to surface and underground water systems.

Pasture-based livestock systems are inherently leaky in terms of N where only 10-20% of N input is utilised in animal product (O'Mara, 2011). Grasslands in Northern Ireland constitute 93% of the agricultural land, of which 83% is classed as permanent grassland (Census, 2010). This inevitably leads to N surpluses within these systems, resulting in N losses to both air and water (Galloway and Cowling, 2002; Galloway *et al*., 2008). Losses of N are derived from urine patches in the field (Di and Cameron, 2002), from application of slurry in a silage-based system or as a result of synthetic N fertiliser.

This part of the study was designed to quantify losses of N by nitrous oxide and nitrate leaching over a period of two years, from 15 February 2010 to 20 February 2012, from three of the dairy production systems (Confinement, WinterCalf and SpringCalf) described in Section 1. A static chamber method was used to measure N_2O emissions, which were verified using the DeNitrification-DeComposition (DNDC) computer simulation model. Residual N in the soil profile to 90 cm was measured in the autumn of 2010 and the DNDC model was used to estimate N leaching losses. The latter approach was adopted as previous research at AFBI, Hillsborough has demonstrated that direct measurement of leaching losses using techniques such as ceramic cups or 'dip wells' can be extremely problematic in Hillsborough soils.

MATERIALS AND METHODS

Nitrous oxide flux measurement

Daily fluxes of N_2O were measured within the silage, grazing and maize components of each of the three systems over a period of two years (not calendar years). Fluxes were measured in Year 2 (2010) of the study (from 15 February 2010 to 11 February 2011) and in Year 3 (2011) of the study (from 7 March 2011 to 20 February 2012). Some details of experimental setup have already been described in Section 1. Briefly, there was 1 chamber in each of the four silage treatments, which were replicated four times (16 chambers), three chambers in each of two grazing plots replicated four times (24 chambers) and one chamber in each maize plot replicated four times (four chambers). The total number of chambers sampled on each occasion was 44 and the N_2O fluxes in the chambers within each replicate were averaged to obtain a mean value for the treatment.

Gaseous N_2O emissions were measured using the static chamber method described by Mosier (1989). Square stainless steel chambers consisted of a lid measuring 0.4 m x 0.4 m wide and 0.1 m high and a base which was inserted into the ground to a depth of ≥5 cm the week before the experiment commenced. The bases were left in position for the two-year duration of the experiment. The base had a trough fitted with neoprene (6 mm depth) into which the chamber lid was placed when sampling and a 10 kg weight was placed on top of the lid to ensure a gas-tight seal. Preliminary studies showed that the neoprene seal provided a gas tight seal for several hr. Gas samples were taken between 10:00 am and 12:00 noon on each sampling occasion, using the mean of 10 ambient air samples as the time zero (T0). Chadwick *et al*. (2014) have shown that the use of ambient air as a surrogate for T0 headspace samples did not result in any consistent bias in calculated fluxes. Sampling occurred three times per week (Monday, Wednesday and Friday) during the spring, summer and autumn periods and was reduced to approximately once a fortnight from November to February (Figure 4.1).

The chambers were sealed using the lids for 60 minutes, and at the end of this time samples of the chamber headspace were taken through a silicone septa positioned in the centre of the chamber lid, using a 20 ml polypropylene syringe equipped with a 25 gauge luer lock needle (0.5 x 16 mm). The syringe was flushed once with headspace air before sampling. A 15 ml sample was withdrawn from the chamber and injected into a 12 ml pre-evacuated glass vial fitted with a double wadded cap (Labco, UK). The lids were removed after gas sampling and placed outside the experimental area until the next sampling occasion.

Linearity checks on N_2O emissions into a static headspace were conducted on three randomly selected chambers, on a monthly basis, during the two-year experimental period. The headspace in these chambers was sampled at 0, 20, 40, and 60 min over the chamber closure period. On all occasions the emission of N_2O was linear. Linearity checks were undertaken on every sampling occasion at another grassland site nearby during 2011 and on approximately 90% of times the production of N_2O was linear (Chadwick *et al*., 2014). Nitrous oxide was determined by gas chromatography (Varian 3800) using a 2.5 m column of Porapak QS 80-100 mesh using an electron capture detector at 300^{o}C with a flow rate of He of 30 ml/min made up to 50 ml/min with N_2. The GC determined N_2O concentrations around ambient (327 ppb) with a precision of determination of ±26 ppb. Gas analysis was undertaken within one month of sampling, which was the maximum storage time for these vials, determined by Laughlin and Stevens (2003). Daily N_2O emissions were expressed as g N/ha and annual fluxes were calculated by linear interpolation between sampling times and the trapezium method was used to calculate the area under the curve. An annual N_2O emission factor was calculated in 2011, by subtracting the emissions from the ungrazed control plots that received no N inputs. Unfortunately N_2O emissions were not measured from the control plots in 2010.

Four composite soil samples were taken (0 to 7.5 cm) from each block at the same time as the headspace sampling, sieved to pass through a 4.5 mm sieve and oven-dried at 105ºC for 24 hr to determine the gravimetric soil water content. Daily rainfall and temperature data were collected from a nearby (<1 km) meteorological station and used to help explain the pattern of daily N_2O fluxes. Soil moisture deficit (SMD) was obtained from the Central Climate Unit, Meteorological office, Exeter, Devon, UK using the Meteorological Office Rainfall and Evaporation Calculation System (MORECS) for a grassland site at Hillsborough.

Residual mineral N in the soil profile to 90 cm was measured during November 2010 from each plot, using a Hydro Care soil sampling system (Model MCL2, Geonor AS) equipped with an auger of 18 mm diameter. Soil cores were taken from each plot and divided into 0-10, 10-20, 20-30, 30-60 and 60-90 cm depths and the sections bulked to form a composite sample at each depth. The composite sample for each depth was broken up by hand and large stones, large roots and plant material removed. The fresh soil was thoroughly mixed before extraction with 2M KCl (1:2 ratio) and concentrations of NH_4^+ and NO_3^- were determined on a Skalar analyser and expressed as mg/kg oven dry soil. Bulk density measurements were taken at each depth and mineral N concentrations were converted to kg N/ha in the total soil profile. For deep coring five cores were taken from each of the four silage treatments, which were replicated four times giving a total of 80 cores. Ten cores were taken from each of the grazed plots, four cores were taken from each maize plot and six cores from each exclusion zone which was also replicated four times (total number of cores 200).

Modelling

The DNDC model (i.e. DeNitrification-DeComposition) is a computer simulation model of carbon and nitrogen biogeochemistry in agro-ecosystems. The model can be used to predict crop growth, soil temperature and moisture regimes, soil carbon dynamics, nitrogen leaching, and emissions of trace gases including nitrous oxide (N_2O), nitric oxide (NO), dinitrogen (N_2), ammonia (NH_3), methane (CH_4), and carbon dioxide (CO_2). Field-DNDC was originally developed for agro-ecosystems in the USA and has been extensively tested at a range of global sites including Ireland (Abdalla *et al*., 2009; Li *et al*., 2011). In this study, the latest version of the field-DNDC model (v. 5:www.dndc.sr.unh.edu) was used to simulate N losses (N_2O and N leaching) from each of the dairy systems. It contains four main sub-models (Li *et al*., 1992; Li *et al*., 2000); the soil climate sub-model calculates hourly and daily soil temperature and moisture fluxes in one dimension, the crop growth sub-model simulates crop biomass accumulation and partitioning, the decomposition sub-model calculates decomposition, nitrification, ammonia (NH_3) volatilisation and carbon dioxide (CO_2) production (heterotrophic and autotrophic respiration) whilst the denitrification sub-model tracks the sequential biochemical reduction from nitrate (NO_3^-) to NO_2, NO, N_2O and N_2 based on the soil redox potential and dissolved organic carbon concentrations.

Essential data to run the model over the 2-year measurement period have been collated and are listed below.

Site and management

- Co-ordinates (longitude, latitude)
- Site history (grassland/arable/forest etc) and dates of land-use change
- Biomass removals of silage and grazing systems and dates of silage cuts
- Grazing: turnout dates, housing dates and stocking rates as livestock units (LU/ha)
- Carbon (C) and nitrogen (N) inputs (urine and faeces). Organic C and N inputs (slurry and/or solid manures) (including dry matter content and Total Available Nitrogen (TAN))
- Mineral fertiliser inputs: type, amount and date of application
- Herbicide/pesticide: date of application (C content)
- Land management i.e. cultivation dates and depth, amount of incorporated biomass, above and below ground (C and N)
- Inclusion of legumes
- Vegetative cover: Shoot/root ratio, C/N ratio contents

Climatic variables

- Precipitation (daily)
- Temperature (daily min and max)
- Solar radiation
- Relative humidity %
- Wind speed
- Potential evapotranspiration
- Actual evapotransformation
- Hydrologically effective rainfall
- Soil moisture deficit
- N deposition wet and dry

Soil parameters

- Soil texture for each layer (default to 15 cm)
- Bulk density (minus stones)
- Stone content
- pH
- Drainage conditions
- Initial soil organic carbon (SOC) contents per layer
- Initial ammonium and nitrate content per layer
- Water-filled pore space (WFPS) at field capacity and wilting point (derived from clay content but can be modified by user)
- Fraction of a) labile carbon, and b) resistant carbon
- Slope

The DNDC model was used to predict the patterns of nitrate loss from each of the dairy production systems described in Section 1. Daily measured values of meteorological parameters and land management records were used as input variables. The DNDC outputs were validated by comparing the modelled data with field measurements of N_2O loss based on static chamber data and leaching of N based on the residual mineral N in the soil profile in the autumn.

Statistical methods

Analyses were carried out using GenStat Version 14 software. The ANOVA model was used to ascertain the significance of silage, grazing and maize systems on the annual cumulative flux of N_2O. A significance level of 0.05 was used, unless otherwise stated. Estimates of means, standard error of means, standard error of differences in means and Fisher's least significant difference (LSD) were calculated. For deep coring a randomised block design ANOVA was conducted based on the blocks (paddocks) and for each depth separately. An ANOVA was conducted on the residual mineral N in the soil profile at each depth and then for the total in the profile to 90 cm.

RESULTS AND DISCUSSION

Meteorological conditions

Total annual rainfall at the site was 884 mm in 2010 and 922 mm in 2011 and the mean annual temperature was 11.7ºC and 13ºC in 2010 and 2011, respectively. In 2010 the average monthly rainfall was lower than the previous 10-year average in April, May, June, October and December and higher in July, September and November (Figure 4.2a). In 2011 rainfall was lower in April, July and August and higher in February and October. In 2010 the average monthly temperature was 19% higher than the 10-year average in June and 56% lower in December, whereas in 2011 temperatures were 50 and 70% higher in April and November, respectively (Figure 4.2b).

The pattern of N_2O flux in 2010 and 2011

Daily N_2O emissions from the silage plots, the grazing paddocks and the maize plots associated with each of the three systems are shown in Figures 4.3 (2010) and 4.4 (2011) (silage), 3.5 (2010) and 3.6 (2011) (grazing), and 3.7 (maize).

In each of 2010 and 2011 there were three applications of slurry and fertiliser (as calcium ammonium nitrate) per year according to RB209 recommendations (as described in Section 2), to the silage plots. At each of the three fertiliser applications slurry was applied 6, 8 and 8 days respectively prior to fertiliser N application and there was little concomitant increase in daily N_2O emission (Figure 4.3, 4.4).

Lampe *et al*. (2006) found that the application of slurry and mineral fertiliser together increased N_2O emissions by between 30 to 150% compared to emissions from CAN alone, following the application in spring to a grassland soil. This effect has been reported in a number of other studies (Stevens and Laughlin, 2001; Stevens and Laughlin, 2002; Velthof *et al*., 1996). A field study conducted by Stevens and Laughlin (2002) showed that if the slurry was applied 3 to 4 days prior to fertiliser application the N_2O flux was greatly reduced as the available C in volatile fatty acids in the slurry had been metabolised before the fertiliser was applied. This therefore reduced the potential interaction between slurry and fertiliser.

In 2010 there were heavy rainfall events in spring which led to small N_2O emissions. The soil was in moisture deficit from April 2010 until heavy rain in July returned the soil to field capacity. August was dry and SMD increased but this was followed by a wet September and dry October. Thereafter the soil remained at field capacity. Peaks of N_2O emission occurred during this autumn and winter period when the grass was slow growing. The largest N_2O peak of 40 g N/ha/day occurred in the Confinement and SpringCalf plots lasting for 21 days from 16 December 2010 to 6 January 2011 with peak emission occurring on 29 December 2010 (Figure 4.3d). December was a very cold month with an average air temperature of 3.5ºC and low rainfall of only 10 mm (Figure 4.3d). Both temperature and rainfall were very low compared with 2011 data and the 10-year average. The peak in N_2O emission at this time was probably due to nitrification producing nitrate during the drier October, which was available for denitrification due to reduced uptake by the grass in December and the wet soil conditions. It is also possible that C may have become available to stimulate denitrification during freezing and thawing processes in the soil during December 2010.

There was substantial mineral N in the top 0-10 cm of soil in November 2010 (Table 4.2). In 2011 there were 3 applications of slurry and fertiliser as shown in Figure 4.4 and on each occasion slurry was applied 7 days prior to fertiliser application. There was a very significant N_2O flux in response to the first N application in all treatments but little response to the other applications later in the year. The N_2O flux peaked in the Confinement, WinterCalf and SpringCalf silage plots at 120, 115 and 80 g

N/ha/day, respectively. The greater N_2O flux in April 2011 compared with 2010, may have been due to a greater amount of rainfall occurring between the first application of slurry and fertiliser in 2011 (7.5 mm) compared with 2010 (2 mm). The amount of rainfall between slurry and fertiliser application in the second and third applications in 2010 was 4.7 and 6.7 mm respectively and in 2011 was 15.4 and 7.5 mm for the second and third applications. Therefore the most likely reason for the absence of a response in these treatments was that there was no excess NO_3^- . A significant flux occurred in all treatments on 30 November 2011 because NO_3^- may have been in excess due to soil mineralisation and nitrification coinciding with 40 mm of rainfall in the previous week (Figure 4.4d).

Grazing paddocks

There were 8 applications of fertilisers to the grazing paddocks (as described in Section 2), with the paddocks fertilised a few days after grazing as indicated by black arrows (Figure 4.5 (2010), 3.6 (2011)). The pattern of N_2O flux is remarkably similar for the WinterCalf and SpringCalf paddocks, in 2010 and 2011 of the study. As with the silage plots, N_2O flux was dependent on the presence of NO_3^-, C and rainfall creating anaerobic conditions in the soil suitable for denitrification to occur. In 2010 the N_2O flux peaked in the WinterCalf and SpringCalf paddocks at 30 g N/ha/day in the autumn when plant uptake of NO_3^- was low and there was a surplus available for denitrification. In 2011 the flux peaked in the WinterCalf and SpringCalf paddocks at 65 and 70 g N/ha/day on 22 June 2012. On 21 and 22 June the daily rainfall was 2 and 13 mm, respectively, hence creating conditions for N_2O emissions. The large response to the grazing and fertilisation in June-July was due to heavy rain in the 7 days after fertiliser application, amounting to 23.6 mm. This coincidental matching of grazing and fertiliser application with heavy rain did not occur to such an intensity in any of the other fertiliser or grazing applications in 2011 or 2010.

Maize system

In 2010 the maize system received an application of cattle slurry (132 kg total N/ha) and synthetic fertiliser (61 kg total N/ha) as calcium ammonium nitrate (CAN) and in 2011 an application of cattle slurry (164 kg total N/ha) and CAN (56 kg total N/ha). The cattle slurry and fertiliser were applied once in each year as indicated by the black arrow (Figure 4.7). The N_2O fluxes were high in both years and continued throughout the growing season, the period with peaks in emissions being associated with rainfall events. This was probably because the establishment of the maize crop was poor, with a dry matter yield in 2010 of only 10.9 tonnes/ha, while the crop failed completely in 2011. Therefore due to limited plant uptake soil NO_3^- concentration must have remained high throughout the growing season and this led to limited uptake of N and therefore a high potential for N loss by denitrification or leaching.

Cumulative N_2O emission

The annual cumulative N_2O-N emission in 2010 for the silage plots associated with the Confinement, WinterCalf and SpringCalf systems were 1.04, 1.03 and 1.30 kg N/ha, respectively, which were not significantly different (Table 4.1). In 2011 the cumulative N_2O-N losses for the Confinement, WinterCalf, and SpringCalf treatments were 1.65, 2.21 and 1.65 kg N/ha, respectively, with these significantly higher than losses from the Control plot (1.20 kg N/ha) which received no N inputs. There was no significant difference between the Confinement and SpringCalf plots, however total N_2O-N emissions were higher from the WinterCalf plots in 2011. Total N_2O emissions were significantly higher in 2011 (1.83 kg N/ha) compared to 2010 (1.14 kg N/ha), when averaged over all silage plots associated with each of the three systems (Table 4.1).

For the Grazed paddocks in 2010 and 2011 there was no significant difference in N_2O-N emissions from paddocks associated with the WinterCalf and SpringCalf systems (Table 4.1). However, over both treatments, the total loss of N_2O-N was significantly higher in 2011 (3.28 kg N/ha) compared to 2010 (1.05 kg N/ha).

For the maize plots the cumulative loss of N_2O-N was 5.21 kg N/ha in 2010 and 7.13 kg N/ha in 2011, with these values not significantly different (Table 4.1).

The loss of N_2O-N expressed as a percentage of N applied either as slurry or fertiliser is also shown in Table 4.1. The average values of N_2O-N loss, as a percentage of N applied in 2010 for the Grazing paddocks, the Silage plots and the maize plots were 0.45, 0.30 and 2.7%, respectively and for 2011 were 1.41, 0.43 and 3.22%, respectively. Grazing can affect the compaction of soil which can create conditions favourable for denitrification. The results presented in Section 3 indicate that there was a difference in the soil properties indicative of soil compaction. The SpringCalf paddocks were significantly more compacted than the WinterCalf paddocks and both treatments were significantly different from the exclusion zone (fertilised and grazed but not compacted). However, there was no significant effect of this compaction on N_2O emission in this study.

If excretal N is taken into consideration in the grazing treatments an additional amount of excretal N equivalent to 91 kg N/ha/yr per dairy cow would be added to the applied N (Nitrate Action Programme and Phosphorous Regulations for Northern Ireland 2011-2014) decreasing the N_2O loss as a percentage of applied N (Table 4.1). Cumulative N_2O-N evolved (excluding excretal N) in all systems was greater in 2011 than in 2010. Total rainfall in 2011 was slightly higher (922 mm) compared to 2010 (885 mm), so it was not the total rainfall that was the driver of N_2O emissions but its distribution at a time when NO_3^- was present in the soil.

Peaks in N_2O emissions occurred when rain fell immediately after calcium ammonium nitrate (CAN) was applied or during the autumn–winter period which coincided with the release of soil NO_3^- from mineralisation and nitrification processes. The high emissions from maize were most likely due to the poor crop in both years of this study.

Residual mineral N in the soil profile in autumn

The mineral N (NH_4^+-N and NO_3^--N) in the soil profile, at different depths, to 90 cm was measured in each plot in November 2010. Mineral N concentrations in the zero N Control silage plot tended to be lower than in the other plots, although this was only significant at the 10 to 20 cm depth (Figure 4.8).

The total mineral N in the soil profile in the maize plots and silage plots (Confinement, WinterCalf and SpringCalf) was 61.4, 67.5, 68.7 and 67.0 mg N/kg, respectively (Table 4.2). There was no significant difference in the residual mineral N content in the soil associated with the silage plots for systems Confinement, WinterCalf and SpringCalf.

The amount of mineral N in the grazing paddocks associated with WinterCalf and SpringCalf was 49.5 and 83.9 mg N/kg, respectively (Table 4.2). However there was no significant difference between the grazing treatments at any depth (Figure 4.9).

The trend with all the plots/paddocks was that most of the mineral N (NO_3^--N + NH_4^+-N) was in the top 30 cm of soil with the proportion of N decreasing with increasing depth from 0-10 to 20-30 cm. There was no significant difference between grazing paddocks and silage plots. There was some mineral N measured below 30 cm indicating that there had been some movement of NO_3^- down the soil profile. Surprisingly the residual mineral N in the maize plots was no greater than in the other treatments despite the poor crop growth (Figure 4.10, Table 4.2).

The total amount of mineral N (NH_4^+ plus NO_3^-) in the soil profile (expressed as kg N/ha) is shown in Figure 4.11 for the Silage and maize plots and in Figure 4.12 for the Grazing paddocks.

The amount of residual mineral N in the silage plots in November 2010 was 79, 78, 77, and 53 kg N/ha for WinterCalf, SpringCalf, Confinement, and Zero N, respectively, the zero N being significantly lower than the other silage treatments. The value for the maize plots was 77 kg N/ha. The amount of residual N in the grazing paddocks was 61.5, 104 and 63.5, kg N/ha respectively, for the WinterCalf, SpringCalf and Exclusion zone (received synthetic fertiliser as CAN but not urine or faeces and was grazed). Although the SpringCalf paddocks had higher residual N in the soil profile there was no significant differences between treatments, due to the large spatial variability. With high residual N levels being measured outside of the growing season there was the potential for leaching to occur during the winter period. However, coring down the soil profile only gives a snapshot of mineral N content at that particular time, which although useful to compare treatments is not an accurate measure of N leaching. The DNDC (i.e. DeNitrification-DeComposition) model was therefore used to predict leaching losses from each of the dairy production systems.

Modelling

The DNDC model was used to validate the measured N_2O emissions and to predict leaching losses. Daily measured values of meteorological parameters and land management records were used as input variables.

A comparison between measured and modelled temporal traces of daily N_2O fluxes for the grazing paddocks associated with the WinterCalf system in 2010 and 2011 is shown in Figure 4.13. There was general agreement between the measured and modelled data, although in some cases, measurements were absent where DNDC generated peaks, and in some cases no measured peaks occurred or were shifted, relative to the model output.

Cumulative modelled emissions for the grazing paddocks associated with the WinterCalf and SpringCalf systems were compared to the measured emissions in 2010 and 2011 (Figure 4.14). The modelled values exhibited the same inter-annual variation as the measured values. For the WinterCalf system the modelled values were a factor of 3.7 greater in 2011 than in 2010 and for the SpringCalf system the modelled values were a factor of 2.7 times greater in 2011. The modelled values for the WinterCalf system were not significantly different from measured values in 2010 or 2011. In the SpringCalf system, the modelled data were higher than the measured values in both 2010 and 2011. However, the measured N_2O emissions generally validated the modelled outputs.

The predicted N losses by leaching generated by the model for the grazing paddocks, maize plots and silage plots are shown in Figure 4.15. Cumulative N leaching losses were observed to range from 7.7 kg N/ha (2% of applied N) for the SpringCalf silage treatment in 2011 to 14.8 kg N/ha (6.7% of applied N) for maize silage cultivation in 2011. Most of the N leaching occurred post September for all systems, with losses higher in the maize plots due to a poor yield of 10.9 tonnes dry matter/ha in 2010 and a crop failure in 2011. The inter-annual variation was generally the opposite to that for N_2O with lower losses predicted in the 2011 measurement period, except for the maize plots.

Leaching losses were low compared to the amount of residual mineral N in the soil profile in the autumn of 2010. However, the predicted leaching losses were comparable to the model MITERRA-EUROPE which was developed to assess N losses from Agriculture in the 27 member states of the European Union (EU-27) (Velthof *et al*., 2009). The model showed that N loss as a percentage of the N applied was 19.1%, 11%, 1.4% and 1.4% for NH_3 volatilisation, N leaching, N_2O emission and NO_x respectively.

The Inter-Governmental Panel on Climate Change (IPCC, 2007) estimate that 1% of applied N whether as slurry or synthetic fertiliser is lost as N_2O-N. In this study only the Maize system in 2010 and 2011 (3.0% average of 2 years) was above this default value when excretal N was accounted for in the N applied. Emission factors (EFs) for N_2O-N could only be determined in 2011 because emissions were measured from the Control (zero N) plots during 2011 only. The EFs for the Confinement, WinterCalf and SpringCalf silage plots in 2011 were 0.09, 0.22 and 0.13, respectively. In 2011 the EFs under grazing were 0.43 and 0.47 for the WinterCalf and SpringCalf paddocks, respectively, which were substantially lower than the IPCC default EF of 1.0%.

In this study, the residual N in the soil profile in autumn was not lost as N_2O, so this suggests that there may be another N loss process that was not measured, for example, the production of benign N_2. However, the production of N_2 gas is very difficult to measure against the large background in the atmosphere, without using expensive ^{15}N stable isotope techniques.

CONCLUSIONS

Each of the three experimental systems in this study had stocking rates close to the 250 kg organic N/ha derogation limit. The results showed that losses of N as either N_2O or N leaching were low, suggesting that with good management, operating under the Nitrates Directive Derogation should have no adverse environmental impact.

Some specific strategies used in this study to reduce GHG emissions included optimal timing of manure within the growing season and the application of slurry prior to the application of CAN which allowed sufficient time for the labile C sources in the slurry to be metabolised before NO_3^- was applied as fertiliser. There were large seasonal variations in N_2O emissions in this study (and others) which points to the uncertainty in estimates of N_2O loss. We must strive to relate variability in N_2O losses between years to differences in fertiliser N input, weather conditions and soil moisture content. Temporal and spatial variability will continue to be a problem, but with more studies of this kind and comparisons between modelled and measured values, the uncertainty in N budgets will be reduced.

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Table 4.1: Total amount of N_2O-N emitted from the grazing paddocks, silage plots and maize plots associated with each of the three systems, for each year of measurement. Letters in parentheses indicate a significant difference at P=0.05 between treatments. Different letters within year or system indicate a significant difference, whereas similar letters show no significant difference. §values in parentheses show additional N applied from excretal N. α values in parentheses indicate N_2O-N loss if excretal N is accounted for in N applied

	Year of study	System	Cumulative N_2O-N evolved (kg N/ha/yr)	Applied N (kg/ha/yr)	N_2O-N loss (% of Applied N)
Grazing paddocks	2010 (Yr 2)	WinterCalf	1.08	235 (234)§	0.46 (0.23)α
		SpringCalf	1.02	234 (199)§	0.44 (0.24)α
	2011 (Yr 3)	WinterCalf	3.34	233 (234)§	1.43 (0.72)α
		SpringCalf	3.21	233 (199)§	1.38 (0.74)α
	2010 (Yr 2)	Average	1.05 (a)		
	2011 (Yr 3)	Average	3.28 (b)		
Silage plots	2010 (Yr 2)	Confinement	1.04	366	0.28
		WinterCalf	1.03	399	0.26
		SpringCalf	1.30	349	0.37
	2011 (Yr 3)	Confinement	1.65 (b)	483	0.34
		WinterCalf	2.21 (c)	456	0.48
		SpringCalf	1.65 (b)	342	0.48
		Control (Zero N)	1.20 (a)		
	2010 (Yr 2)	Average	1.14 (a)		
	2011 (Yr 3)	Average	1.83 (b)		
Maize plots	2010 (Yr 2)		5.21	193	2.70
	2011 (Yr 3)		7.13	221	3.22

Table 4.2: Distribution of total mineral N (NH_4^+ plus NO_3^-, mg N/kg) in the soil profile of the maize plots, the silage plots and the grazing paddocks of all systems in November 2010

Depth	Maize plots	Silage plots				Grazing paddock	
		Confinement	WinterCalf	SpringCalf	Zero N	WinterCalf	SpringCalf
Cm				mg N/kg			
0-10	24.3	34.7	40.7	37.1	22.4	26.2	47.2
0-20	20.0	18.1	16.4	17.4	10.7	13.0	19.5
20-30	9.3	8.2	5.4	6.8	5.1	4.5	7.9
30-60	4.3	3.0	3.8	3.6	2.4	2.5	5.3
60-90	3.5	3.5	2.4	2.1	2.1	3.3	4.0

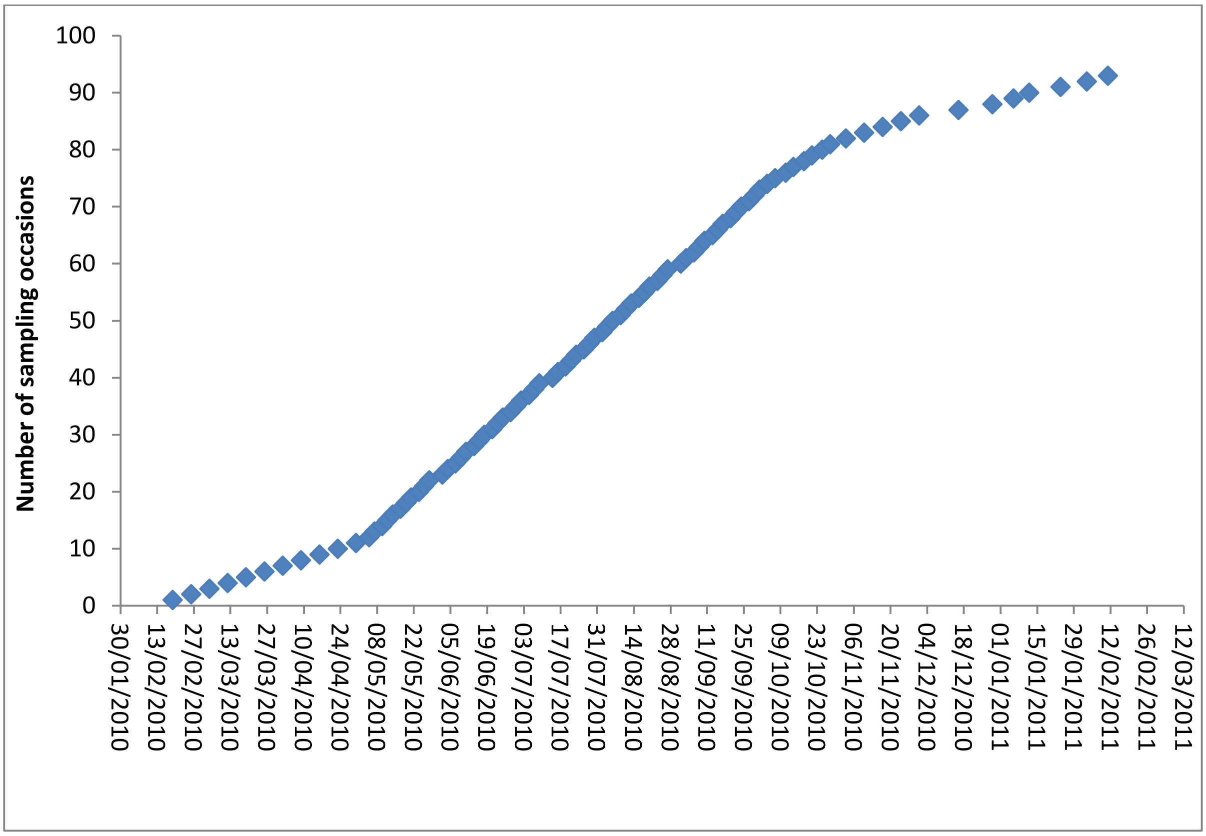


Figure 4.1: Cumulative number of gas sampling events over 12 months.

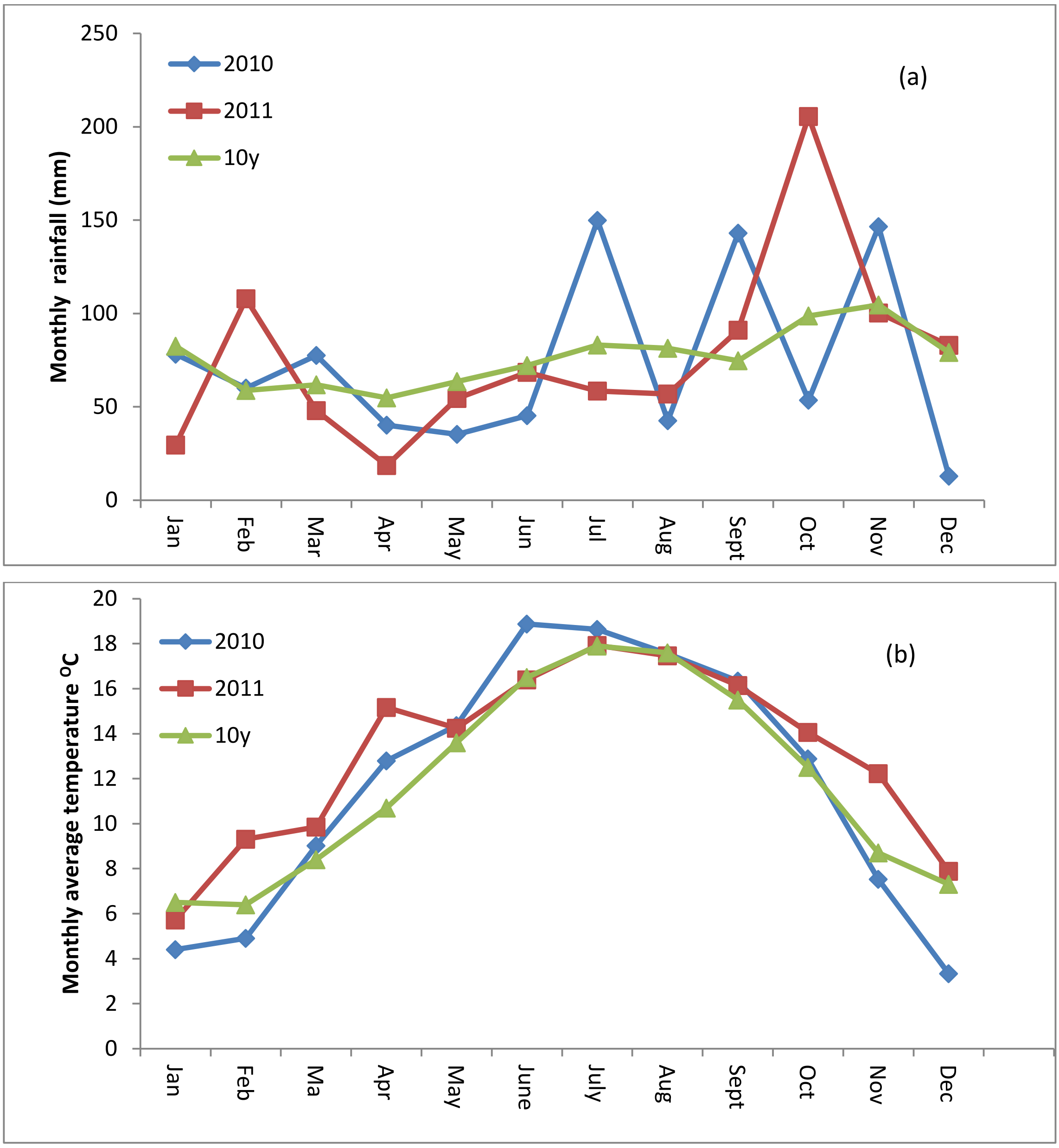


Figure 4.2: Total monthly rainfall (a) and monthly average temperature (b) for 2010 and 2011. The previous 10-year averages are also shown

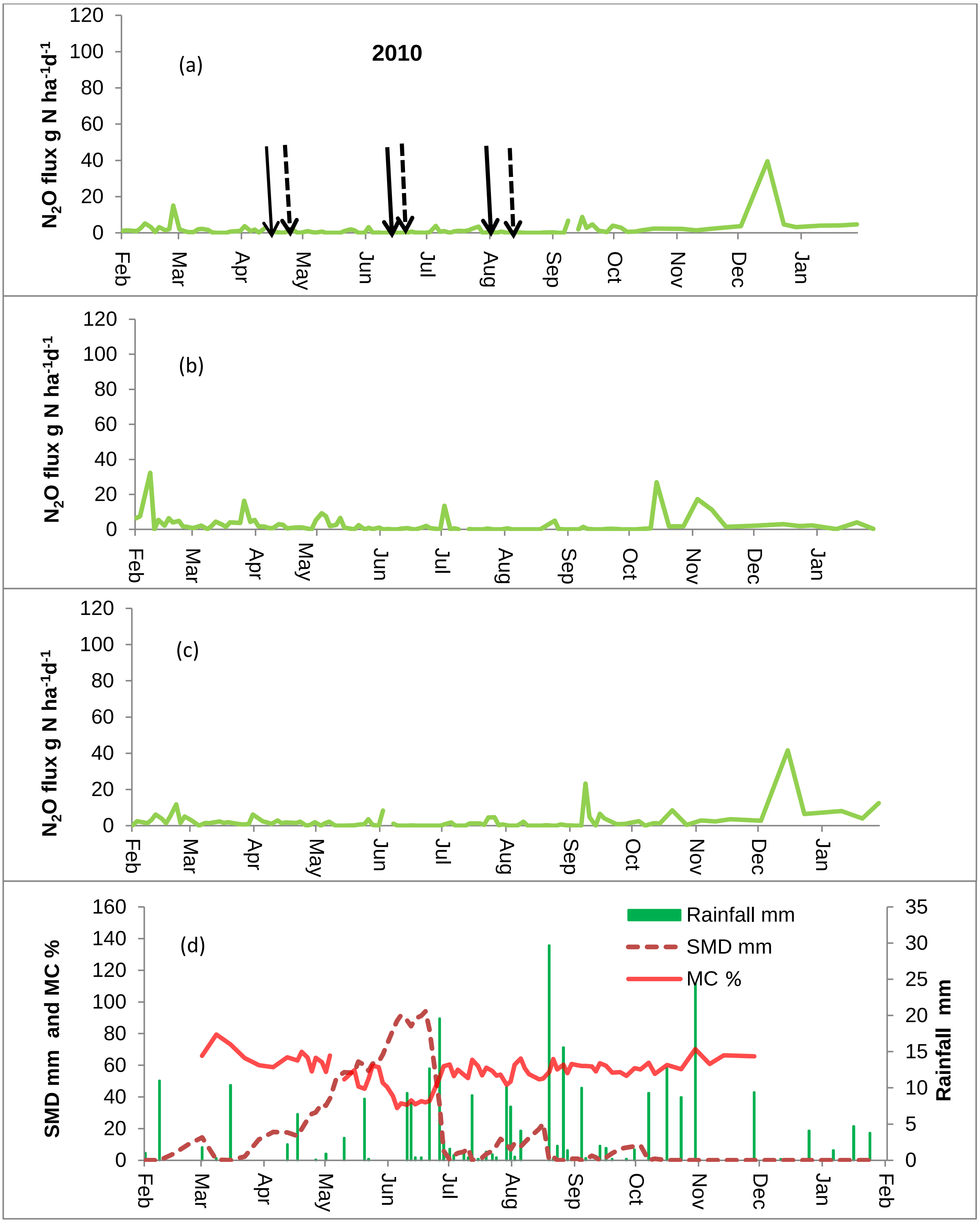


Figure 4.3: Daily nitrous oxide fluxes for silage plots associated with the (a) Confinement, (b) WinterCalf and (c) SpringCalf systems (d) daily soil moisture deficit (SMD), rainfall and gravimetric moisture content (MC %) in 2010. Slurry application dates are shown by a solid black arrow and fertiliser application dates by a dashed black arrow

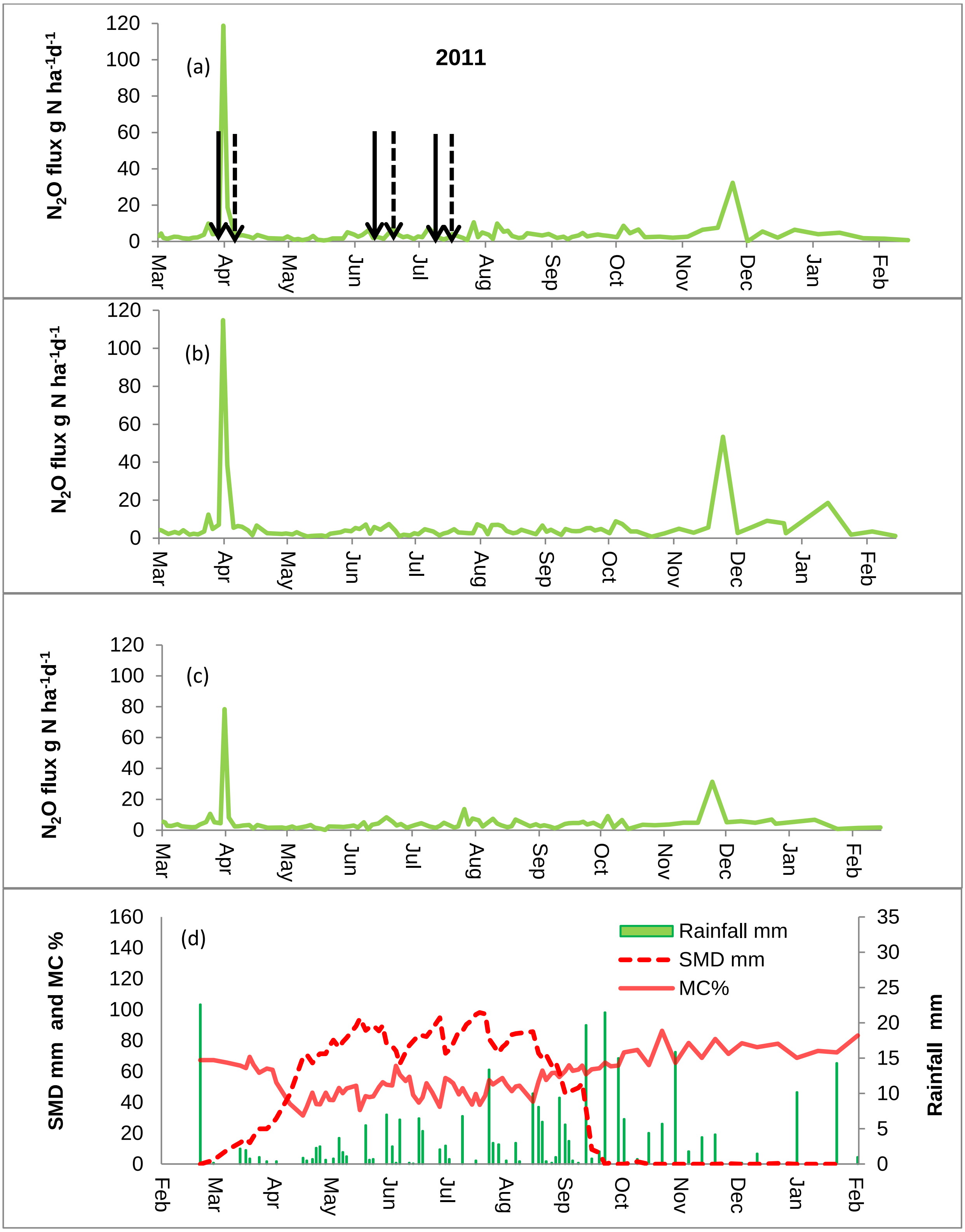


Figure 4.4: Daily nitrous oxide fluxes for silage plots associated with the (a) Confinement, (b) WinterCalf and (c) SpringCalf systems (d) daily soil moisture deficit (SMD), rainfall and gravimetric moisture content (MC %) in 2011. Slurry application dates are shown by a solid black arrow and fertiliser application dates by a dashed black arrow

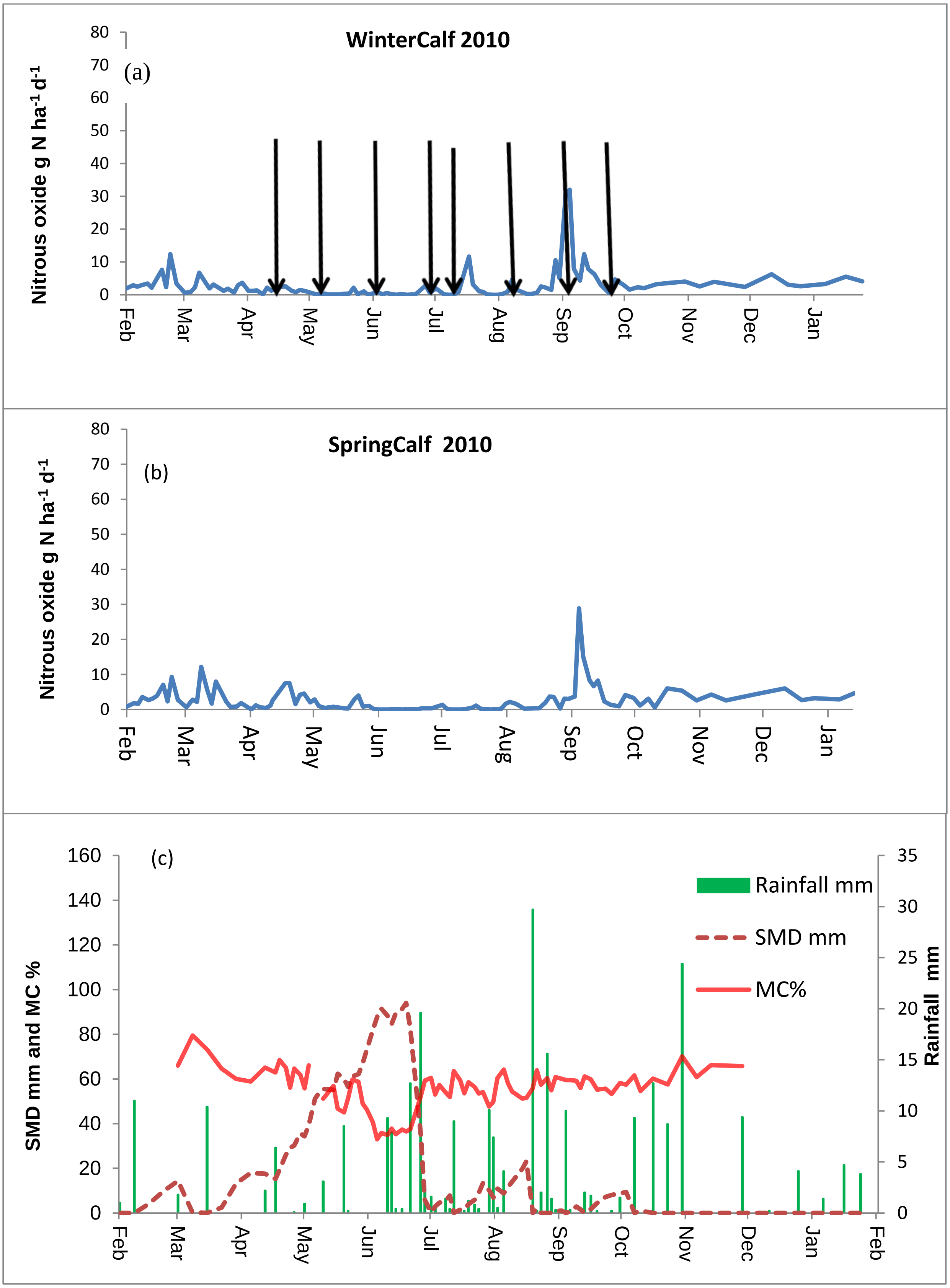


Figure 4.5: Daily nitrous oxide fluxes from the grazing paddocks associated with the (a) WinterCalf and (b) SpringCalf systems together with (c) daily soil moisture deficit (SMD), rainfall and soil moisture content (MC%) in 2010. Fertiliser application dates shown by black arrows

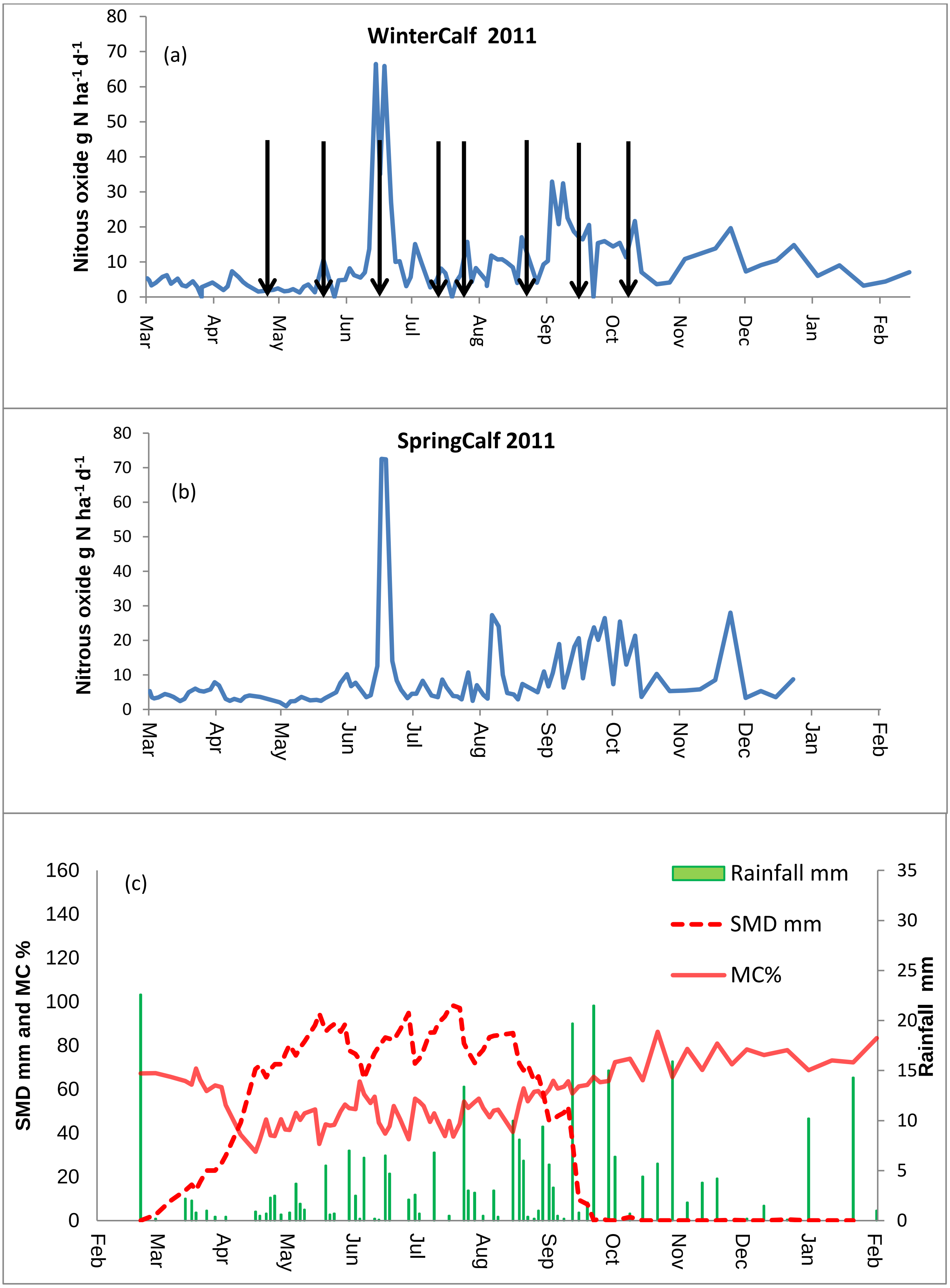


Figure 4.6: Daily nitrous oxide fluxes from the grazing paddocks associated with the (a) WinterCalf and (b) SpringCalf systems together with (c) daily soil moisture deficit (SMD), rainfall and soil moisture content (MC%) in 2011. Fertiliser application dates shown by black arrows

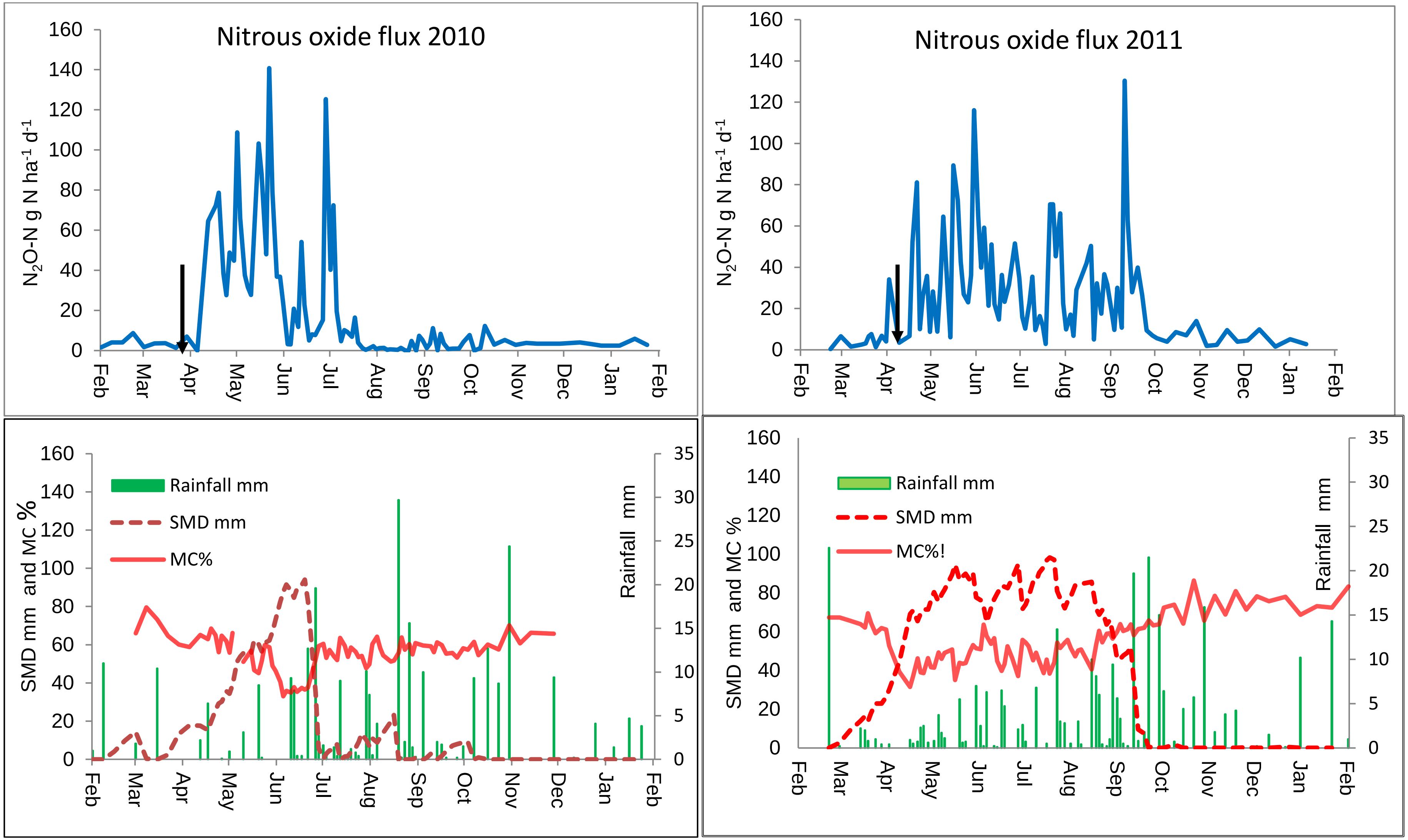


Figure 4.7: Daily nitrous oxide flux from the maize plots associated with the Confinement and WinterCalf system over the two years of the study with daily rainfall, SMD and gravimetric moisture content of soil (on an oven dry basis). The date of slurry application is indicated by a black arrow, while fertiliser as CAN was applied one week later

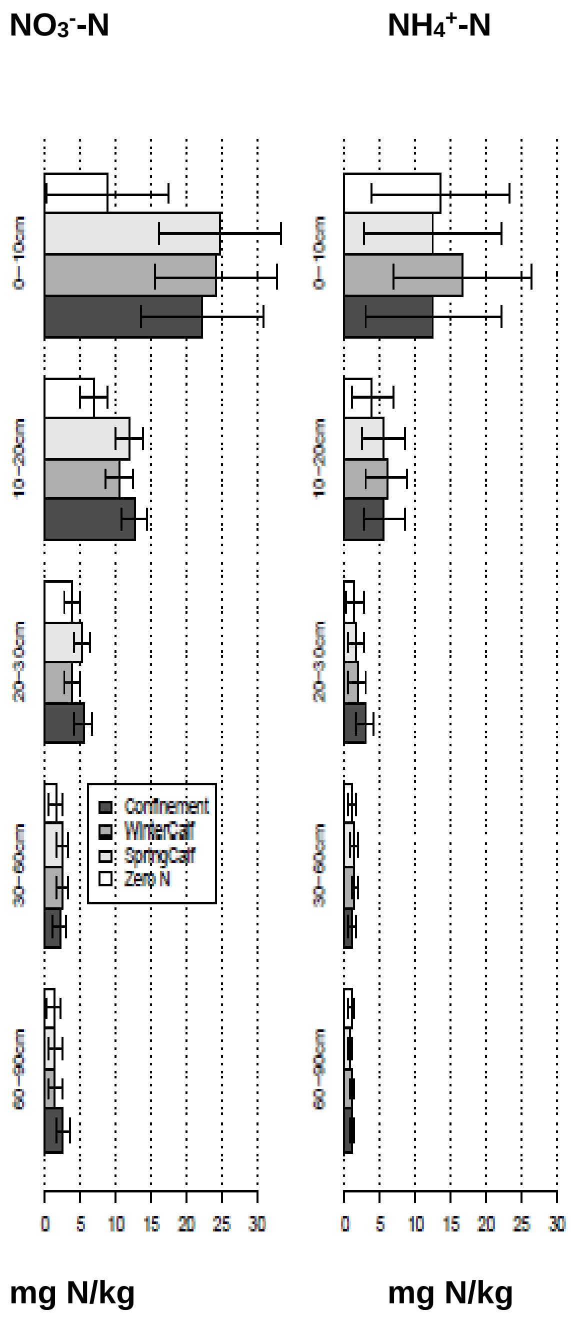


Figure 4.8: Residual mineral N (mg/kg) in the soil profile of the silage plots in November 2010 (mean and confidence interval (ci $_{0.5}$))

Figure 4.9: Residual mineral N (mg/kg) in the soil profile of the grazing paddocks associated with systems WinterCalf and SpringCalf, and in their ‘exclusion zones’ in November 2010 (mean and confidence interval (ci $_{0.5}$))

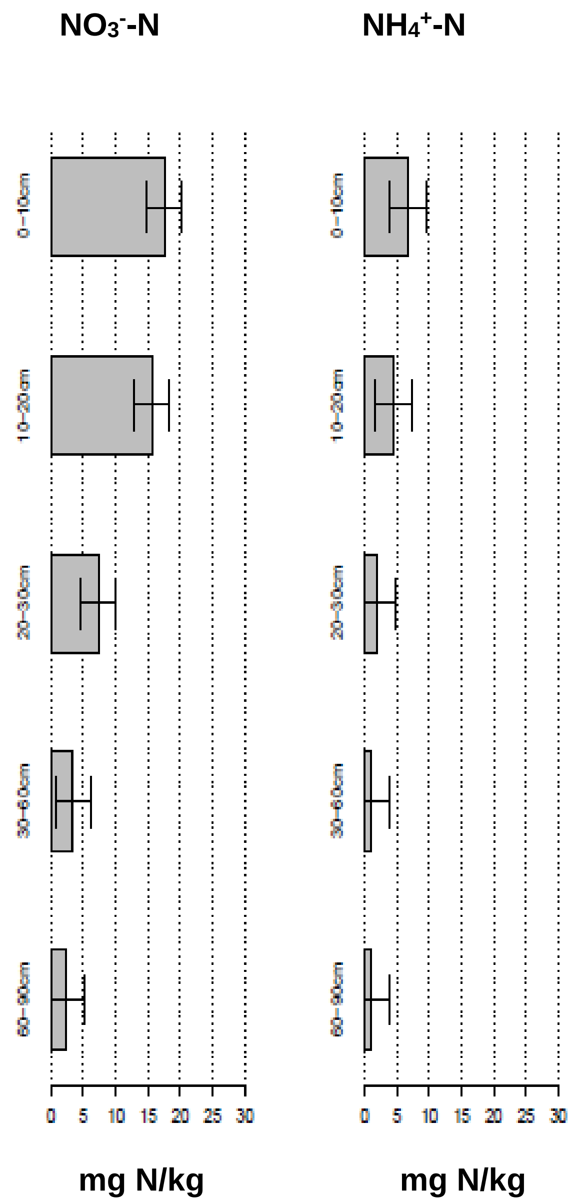


Figure 4.10: Residual mineral N (mg/kg) in the soil profile of the maize plots associated with the Confinement and WinterCalf systems in November 2010 (mean and confidence interval (ci $_{0.5}$))

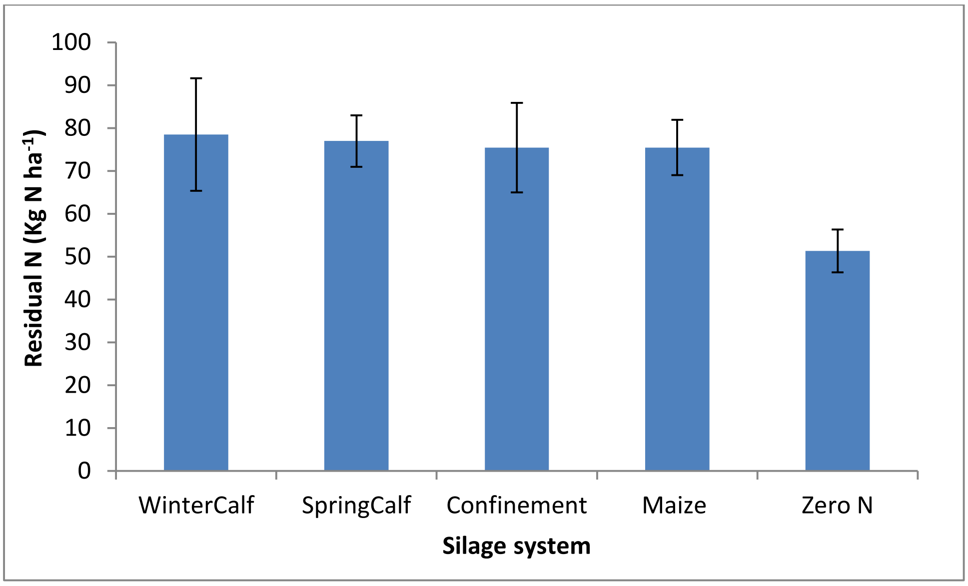


Figure 4.11: Total residual N (kg N/ha) (NH_4^+ plus NO_3^-) in the soil profile of the Silage plots and maize plots. Error bars are standard errors of the mean

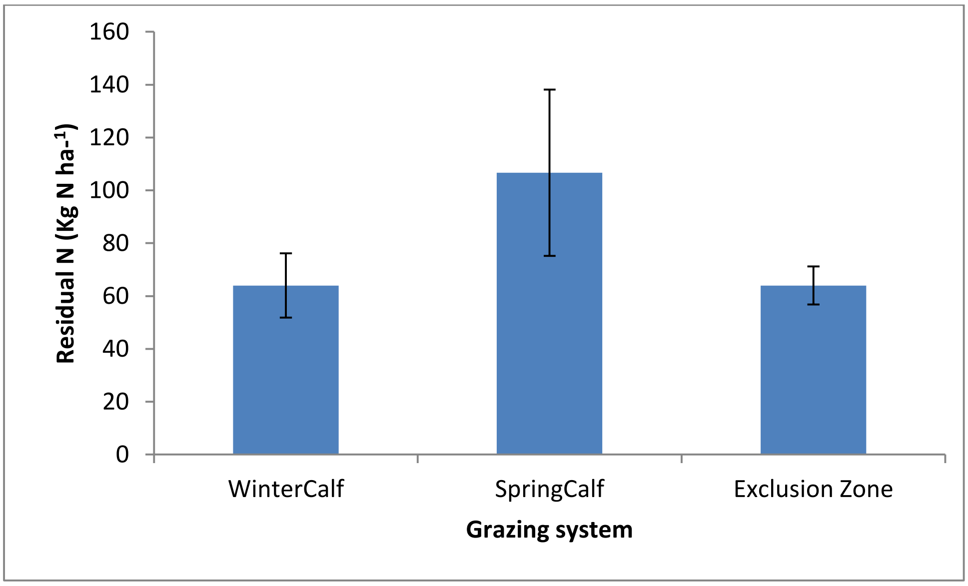
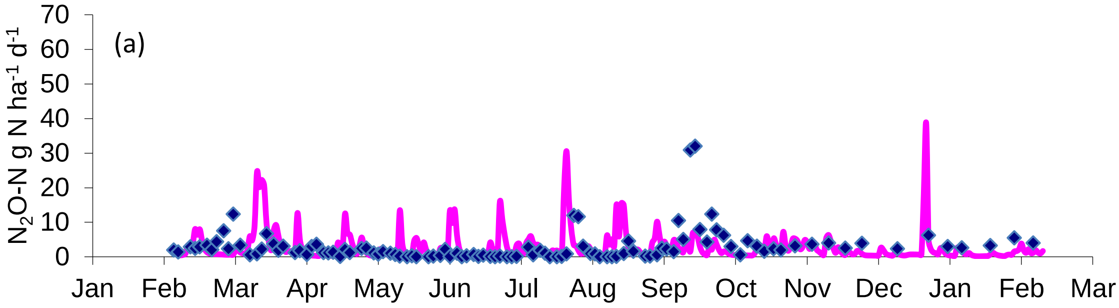


Figure 4.12: Total residual N (kg N/ha) (NH_4^+-N plus NO_3^--N) in the soil profile of the grazing paddocks for WinterCalf and SpringCalf, and in the Exclusion zones. Error bars are standard errors of the mean



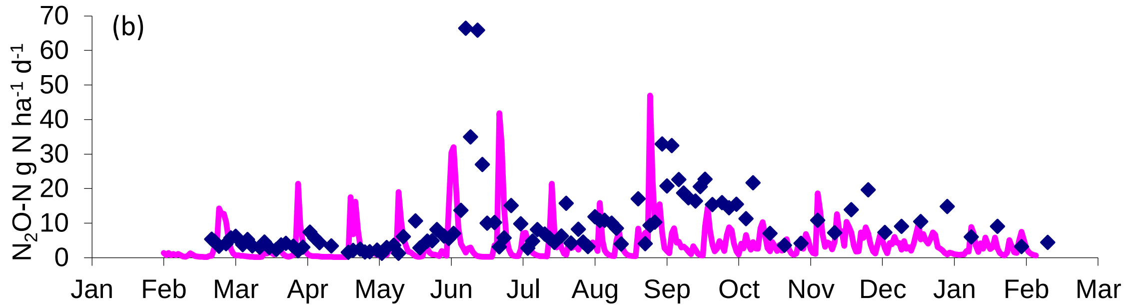


Figure 4.13: Temporal trace of measured (diamonds) and modelled (line) N_2O fluxes from the grazing paddocks associated with the WinterCalf system in (a) 2010 and (b) 2011

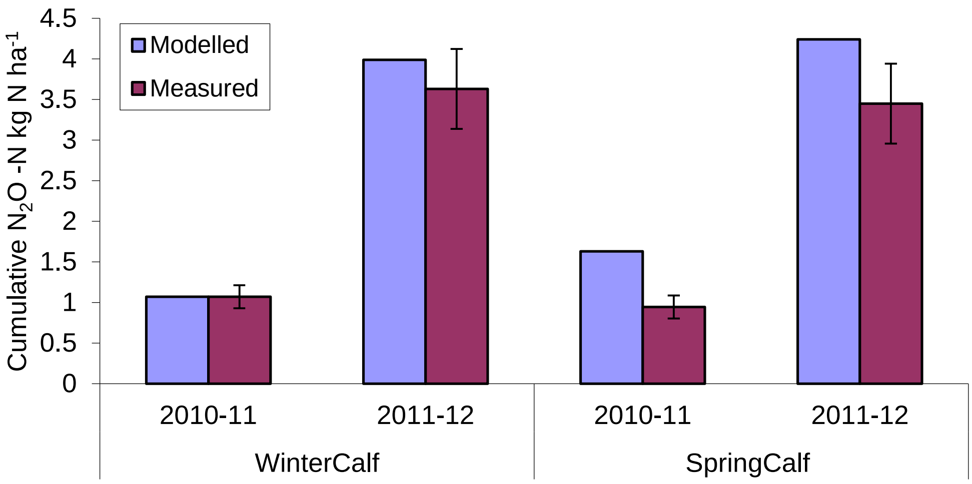


Figure 4.14: Modelled and measured cumulative N_2O losses for the grazing paddocks associated with the WinterCalf and SpringCalf systems. Error bars are standard errors

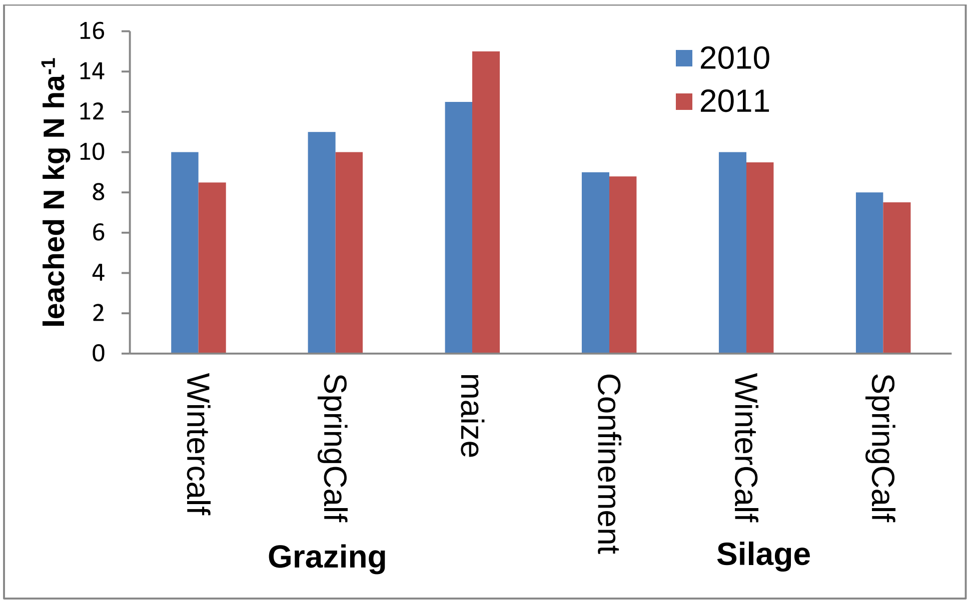


Figure 4.15: Predicted N loss by leaching (kg N/ha) from the grazing paddocks associated with systems WinterCalf and SpringCalf, the Maize plots and Silage plots associated with systems Confinement, WinterCalf and SpringCalf

SECTION 5

Four plot scale experiments examining strategies to reduce phosphorus losses from applied slurry

BACKGROUND

While P losses within the experimental site as a result of grazing have been examined within Section 3, it was realised that large scale plots did not provide the optimum approach by which to examine factors influencing P losses from slurry applied to intensive grassland systems. For this reason, four detailed small experiments were conducted on an adjoining site to examine strategies by which to reduce P losses. These studies were undertaken as part of a PhD linked to the main study, with a brief overview of each of these experiments presented below:

Experiment 1 *(Phosphorus losses from low emission slurry spreading techniques)* was designed to investigate the effect of slurry application technique on slurry-associated phosphorus concentrations in runoff. Dairy cow slurry was applied to freshly harvested grassland stubble by hand to simulate splashplate, trailing shoe, and shallow injection spreading techniques. Both the trailing shoe and shallow injection techniques were applied ‘across’ the slope of the field, or ‘down’ the field slope. Slurry application via the trailing shoe and shallow injection reduced dissolved reactive phosphorus (DRP) concentrations in runoff by 37 and 47%, respectively, relative to traditional splashplate spreading techniques. There was no effect of application direction (across or down slope) on P concentrations in runoff. In addition, slurry was also applied to a four-week regrowth, using the same slurry spreading techniques listed above. In contrast, slurry spreading technique had no effect (P>0.05) on P concentrations in runoff following this application. This was attributed in part to the very dry weather and soil conditions which resulted in problems generating runoff at this time. Nonetheless this experiment clearly demonstrated the potential of the trailing shoe and shallow injection slurry spreading techniques to reduce DRP concentrations in runoff, compared with the traditional splash plate technique. This study has now been published in Journal of Environmental Quality.

McConnell, D.A., Ferris, C.P., Doody, D.G., Elliott, C.T. and Matthews, D.I. (2013). Phosphorus losses from low-emission slurry spreading techniques. *Journal of Environmental Quality*, **42:** 446-454.

The second experiment (**Experiment 2:** *The impact of herbage regrowth interval on phosphorus losses in runoff post slurry application)* was designed to investigate the effect of herbage mass on P concentrations in runoff, following slurry application with the trailing shoe technique. Slurry was applied by hand to plots with three levels of herbage cover: a 0-day regrowth, a 10-day regrowth, and a 20-day regrowth. Dissolved reactive P concentrations in runoff were significantly reduced (P<0.05) following slurry application to a 10-day or 20-day herbage regrowth, relative to the 0-day regrowth treatment. In contrast, herbage regrowth had no significant effect on PP concentrations in runoff. Thus this experiment demonstrated that allowing a grass sward to recover for between 10 to 20 days following harvest before applying slurry, can be highly effective in reducing P losses in runoff. This study has now been published in Agriculture, Ecosystems and the Environment.

McConnell, D.A., Doody, D.G., Elliott, C.T., Matthews, D.I. and Ferris, C.P. (2013). The impact of herbage regrowth interval on phosphorus losses in runoff post slurry application. *Agriculture, Ecosystem and the Environment*, **178:** 100-108.

Experiment 3 *(The impact of slurry application method on phosphorus loss in runoff from grassland soils during winter and early spring)* examined the effect of slurry application technique (Splashplate/Trailing shoe) and timing of slurry application (winter/early spring) on P concentrations in runoff. Slurry was applied by hand on four occasions during the winter/spring period (7 December, 18 January, 1 March, and 12 April) simulating either the splashplate or trailing shoe technique. Following each application, DRP, PP and total P concentrations in runoff were significantly greater (P<0.05) from the Splashplate treatment than from the Trailing shoe treatment. In addition, DRP concentrations in runoff from the Splashplate treatment were greater following the December and March slurry applications, than following the January and April applications, with the former application dates coinciding with periods of higher volumetric soil moisture content. In contrast, P concentrations in runoff from the Trailing shoe treatment did not differ between the four slurry application dates. While again highlighting the potential of the trailing shoe system to mitigate against P losses from applied slurry, this experiment also demonstrated that soil moisture content, and not season *per se*, was a significant driver of P losses. This paper has been submitted to the Irish Journal of Agriculture and Food Research.

The fourth experiment (**Experiment 4**: *Phosphorus loss in runoff following the application of anaerobically digested slurry to grassland)* was designed to investigate the effect of anaerobic digestion of slurry on P losses in runoff following slurry application to grassland. Both anaerobically digested (AD) slurry and undigested (UD) slurry were applied to grassland via a simulated splashplate spreading technique. Despite AD slurry having a higher (P<0.001) water extractable P content than UD slurry, DRP concentrations in runoff were unaffected (P>0.05). In contrast, both dissolved unreactive P and PP concentrations in runoff from the AD slurry treatment were lower (P<0.05) than from the UD slurry treatment. The results of this experiment highlight that the anaerobic digestion of slurry does not increase the risk of P being lost in runoff following slurry application.

Full details of the methodologies, results, and implications of the findings of each of these four experiments are presented in Appendix 1.

APPENDIX 1

Small scale Experiment 1

Phosphorus losses from low emission slurry spreading techniques

ABSTRACT

While low emission slurry spreading techniques are known to improve nitrogen use efficiency, their impact on phosphorus (P) losses in surface runoff has received little attention. The current study was designed to examine the effect of slurry spreading technique on P losses in runoff. Twelve treatments were examined on 0.5 x 1.0 m plots in a nominal 2 x 6 factorial design experiment. Treatments comprised grass swards at two different stages of growth: a stubble and a four-week regrowth and six different slurry application treatments: Control (no slurry), and slurry applied to simulate Splashplate, Injection (ACROSS and DOWN slope) and Trailing shoe (ACROSS and DOWN slope) spreading. Slurry was applied by hand (40 m^3/ha). Rainfall simulations (40 mm/hr) were conducted at 2, 9 and 28 days post slurry application. When slurry was applied to the Stubble, dissolved reactive P (DRP) concentrations in runoff at day 2 were 47 and 37% lower ($P<0.05$) from the Injection and Trailing shoe treatments compared with the Splashplate treatment. Similarly, at day 2 TP concentrations in runoff from the Injection treatments were 27% lower ($P<0.05$) than the Splashplate treatment. In contrast, application technique had no effect ($P>0.05$) on P concentrations in runoff following slurry application to regrowth treatment. Phosphorus concentrations in runoff were unaffected by direction of slurry spreading (ACROSS or DOWN) at both applications. Consequently, trailing shoe and injection techniques offer the potential to reduce DRP concentrations in runoff during the period immediately after slurry application.

INTRODUCTION

Agriculture is a key contributor to the decline in surface water quality witnessed in many countries which have intensive agricultural sectors (Carpenter *et al.*, 1998; Smith *et al.*, 2005). In Northern Ireland (NI) agriculture has been identified as the primary driver of phosphorus (P) induced eutrophication of surface waters, accounting for almost 60% of terrestrial P inputs to inland waterways (Smith *et al.*, 2005). These inputs can be attributed in part to the large agricultural P surplus (16.5 kg/ha/yr) which existed until recently (Foy *et al.*, 2002). To address this problem, and comply with European Union legislation (for example, the Nitrates Directive (1991) and Water Framework Directive, (2000)), P inputs to agricultural systems have been reduced by restricting inorganic P use to meet crop requirements (Maguire *et al.*, 2009), and through a reduction of the P content of animal feedstuffs (Ferris, 2010; McCann *et al.*, 2007). In addition, on-farm manure management practices are increasingly being advocated to minimise direct transfers of manure P (incidental losses) to surface runoff (Haygarth and Jarvis, 1999). Within NI these include: the introduction of a ‘closed period’ (15 October–31 January) during which land application of slurry is prohibited, a maximum slurry application rate for grassland of 50 m^3/ha, and restrictions based on ground and weather conditions during which slurry application is not permissible (EHS, 2006).

In addition, the use of alternative ‘low emission’ slurry application techniques have also been suggested as a potential strategy to minimise incidental P losses post slurry application (Maguire *et al.*, 2011). These techniques, which include trailing shoe and shallow injection spreading apparatus, are becoming increasingly common within grassland-based systems as they allow slurry to be placed at the base of the sward thus reducing the risk of contaminating the sward with slurry (which can hinder growth) while affording a greater window for slurry application (Webb *et al.*, 2010). In addition, there is now a considerable body of evidence which suggests that nitrogen (N) use efficiency following slurry application can be greatly improved using ‘low emission’ spreading techniques. For example, in a recent review Webb *et al.* (2010) reported average reductions in gaseous ammonia emissions of 65 and 86% following slurry application to grassland via trailing shoe and open slot injection techniques, respectively, in comparison to broadcast spreading. Similarly, Frost *et al.* (2007) recorded a 25% increase in crop N recovery when slurry was applied using a ‘trailing shoe’ system, compared to a traditional ‘splashplate’ spreading system. However, the impact of these techniques on P losses from grasslands has not been examined. The incorporation of manure into soil by ploughing, disking and cultivating have all been shown to reduce dissolved P loss in both runoff (Allen and Mallarino, 2008; Little *et al.*, 2005; Sharpley *et al.*, 2004) and leachate (Geohring *et al.*, 2001, Kleinman *et al.*, 2009). However, these techniques impose a trade-off between slurry-associated P loss and soil-associated P loss, whilst the levels of soil disturbance involved makes these techniques unsuitable within grassland systems (Daverede *et al.*, 2004; Little *et al.*, 2005; Maguire *et al.*, 2011). There is however evidence that soil aeration prior to slurry application can reduce P losses in runoff from grassland (Butler *et al.*, 2008; Shah *et al.*, 2004). Similarly, Van Vliet *et al.* (2006) found that slit aeration (depth = 15 cm) prior to broadcast slurry application on grasslands reduced dissolved reactive P loss in natural runoff events by 74%, whilst Johnston *et al.* (2011) found that applying slurry in bands over aeration slots was highly effective in reducing P losses in runoff. In addition, Uusi-Kamppa and Heinonen-Tanski (2008) observed lower P concentrations in surface and near-surface runoff following the injection of slurry to a depth of 0.1 m, relative to broadcast application. However, to date it would appear that research has not been undertaken into incidental P losses in runoff from low emission slurry spreading techniques such as 'shallow injection' and 'trailing shoe'. As these techniques have become increasingly common within intensive grassland systems in many parts of Europe, it is important that their impact on P losses is quantified.

Thus the aim of the current experiment was to compare P losses in runoff following slurry application to grassland using either traditional broadcast (splashplate), shallow injection, or trailing shoe slurry spreading techniques.

MATERIALS AND METHODS

Site description

This experiment was undertaken at the Agri-Food and Biosciences Institute, Hillsborough, NI (54˚27'N; 06˚04'W). The experimental site (approximately 400 m^2) was located on the northerly aspect of a drumlin hill slope with a uniform slope of 7.2%. The soil was a Soil Water Gley Class 1 soil overlying Silurian Shale (FAO classification: Dystric Gleysol). Immediately prior to the start of the experiment the soil had an Olsen P content of 39 mg/kg, twice the agronomic optimum. The site had a bulk density of 0.7 g/cm^3 at 0-5 cm depth. and a Hydrology of Soil Type (HOST) classification of 17, representing a drained mineral soil overlying an impermeable substrate and corresponding to approximately 2.95% of the land area of NI (Higgins, 1997). The field where the site was located was reseeded in 2002 with perennial ryegrass (*Lolium perenne*) and was grazed by dairy cows during each subsequent year. Prior to the beginning of the experiment in February 2009, the site was last grazed on the 21 September 2008. The experimental site received urea fertiliser at a rate of 28 kg N/ha on 28 February 2009. Average annual rainfall and duration of the growing season recorded at the site was 890 mm and 254 days for the periods 1971–2000 and 1951–1990, respectively (Betts, 1997).

Treatments

Two growth stages: a stubble (Stubble) and a four-week regrowth (Regrowth) and six slurry application treatments (control, splashplate, shallow injection across the slope, shallow injection down the slope, trailing shoe across the slope, and trailing shoe down the slope) were combined in a full factorial treatment structure to generate 12 treatments that were arranged in a split plot design and replicated four times. No slurry was applied to the Control treatment. Slurry applied to the 'ACROSS' slope treatments followed the contour of the field slope whilst slurry applied to the 'DOWN' treatments followed the line of the gradient of the slope. Forty-eight plots, each measuring 1.0 x 0.5 m, were laid out in four blocks (A, B, C and D). Each block was divided into two (representing the two grass growth stages), with each of the six slurry treatments represented within each half of each block. Plots within each block were sited 1.0 m apart, while the distance between each block was 3.0 m.

Herbage within the blocks was cut on 25 (Blocks A and B) and 26 (Blocks C and D) May using a side mounted tractor mower (Amazone, Hasbergen, Germany) so as to avoid damage to blocks by tractor tracks, and cut herbage removed by hand. Herbage from the areas between blocks was cut using an Agria mower (5400, Agria, Möckmühl, Germany) and removed by hand.

Slurry application

Dairy cow slurry was used in this experiment. This was collected from an underground slurry store two weeks before the experiment commenced and subsequently stored in a high density polyethylene (HDPE) container at <4ºC until applied. Slurry was applied to the Stubble treatments on 25 and 26 May, and to the Regrowth treatments on 22-23 June. On each occasion slurry applications were split over two days (Blocks A and B on day 1, and Blocks C and D on day 2) as time constraints meant it was not possible to conduct all runoff measurements on a single day. Within each of the slurry treatments, slurry was applied at a rate of 40 m^3/ha (2 litres/plot). The slurry used had a mean dry matter (DM), P, N and ammonia N content of 107 g/kg, 9.96 g/kg DM, 41.5 g/kg fresh, and 17.0 g/kg fresh, respectively, and a mean pH of 7.32. Prior to slurry being applied, mean grass heights were measured across the Stubble and Regrowth treatment blocks using a rising plate meter (Filips folding plate pasture meter, Jenquip, New Zealand) (30 'drops' per block), with mean heights being 5.9 and 15.7 cm, respectively.

Slurry was applied by hand within each of the slurry treatments, with no slurry being applied within 0.05 m of the top and bottom of the plot edges, and within 0.025 m of the plot sides, so as to minimise edge effects following the installation of stainless steel sheets which were used to isolate the plots. The Splashplate treatment was simulated using a pouring jug and a wooden board, with the slurry evenly applied across the plot to an area measuring 0.9 x 0.45 m. With the Trailing shoe treatments, grass was parted by hand and held in place with wooden boards while slurry was applied to the base of the sward using a thin spouted plastic jug. Likewise, to simulate the Injection treatments, grass was parted by hand and held in place using wooden boards, while a metal 'V' shaped cutting blade (0.03 m deep, maximum breadth 0.03 m) was used to create a slit into which slurry was placed. This blade was attached to a flat metal 'foot plate' (0.05 m above the top of the cutting V), and downward pressure applied to this plate to force the blade into the soil surface. Slurry was placed directly into the soil using a thin spouted plastic jug. Following slurry application the slit remained open and slurry exposed to the air. With the Injection DOWN and Trailing Shoe DOWN treatments, slurry was placed in three tramlines each 0.9 m long and spaced 0.225 m apart, running parallel to the direction of the slope. Slurry in the ACROSS treatments was applied along six tramlines running across the slope, each tramline 0.45 m long, with tramlines spaced 0.18 m apart. Both the ACROSS and DOWN treatments had a total tramline length of 2.7 m, with slurry divided equally between each tramline. Slurry occupied 81% of the plot area under the Splashplate treatment while slurry on the Injection and Trailing Shoe plots accounted for approximately 10 and 20% of the plot area, respectively.

Immediately after slurry was applied, each plot was isolated from the surrounding area using stainless steel surrounds which were placed vertically into the soil to a depth of approximately 0.05 m along the sides and across the up-slope end of each plot. On the down-slope plot edge a shallow trench was excavated and a stainless steel V-shaped collection tray placed in the trench to act as a runoff collector. The upslope edge of each tray was fitted with a 0.07 m horizontal lip, and this was driven horizontally (approximately 0.05 m) into the soil directly underneath each plot, at a depth of approximately 0.03 m below the soil surface. The collection trays were placed in position two weeks prior to herbage being removed so as to minimise soil disturbance during the experiment. Care was taken not to disturb these when harvesting grass from each block. A 0.02 m diameter outlet at the base of the collection tray allowed runoff to drain into a two litre HDPE collection container via an underground pipe. Plots were covered with translucent plastic sheeting for the 48-hr period between slurry application and the first rainfall simulation event, after which they remained uncovered.

Rainfall simulation

Rainfall simulations were performed at three time intervals: 2 (Runoff day 2; RD2), 9 (Runoff day 9; RD9), and 28 (Runoff day 28; RD28) days post slurry application. Two Amsterdam drip-type rainfall simulators, as described by Bowyer-Bower and Burt (1989), were employed to supply rainfall at a constant rate. During simulations, wooden boards (1.2 m^2) were placed along two sides of the rainfall simulators to act as a wind shield, thus preventing water droplets being blown outside the plot boundaries. Rainfall was delivered at a rate of 40 mm/hr. Following the initiation of runoff, thirty minutes of runoff was collected as 3 x 10 minute fractions at each rainfall simulation. Runoff volume and time taken to generate runoff were recorded. Water used in the rainfall simulations was passed through a deionising cylinder (DC9, Purite Ltd, Thames Oxon, UK) to reduce its P concentration. The cylinder delivered deionised water with an average dissolved reactive P (DRP) and nitrate concentration of 20.8 μg/l and 416 μg/l respectively. Three volumetric soil moisture (VSM) readings were taken on each plot to a depth of 6.0 cm at each rainfall simulation event using a soil moisture probe (HH2, Delta-T Devices Ltd., Cambridge, UK).

Water quality analysis

Runoff samples were placed in a fridge (3°C) within 4 hr of being collected. Samples were analysed for DRP, total dissolved P (TDP), and total P (TP) within 48 hr of being collected. Samples for DRP and TDP analysis were filtered through 0.45 μm filters (MF-Millipore, Billerica, MA) before analysis. Dissolved reactive P was determined by the acidic molybdate-ascorbic acid method of Murphy and Riley (1962). Total dissolved P and TP were determined by digestion with potassium persulphate and sulphuric acid, followed by analysis of the digest as outlined above for DRP (Eisenreich *et al.*, 1975). Particulate P (PP) was calculated as the difference between TP and TDP. Dissolved unreactive P (DUP) was calculated as the difference between TDP and DRP.

Statistical analysis

Data were analysed using Genstat (Version 12.1, VSN International Ltd, 2009, UK) according to the split plot design. Flow-weighted mean P concentrations were calculated from the three ten-minute fractions of runoff collected. Phosphorus export rates were calculated as a product of runoff volume and P concentration for each 10 minute fraction and totalled for each 30 minute rainfall simulation event. As weather and soil conditions at the Stubble and Regrowth applications were very different, direct comparisons of slurry treatments at different growth stages would have been inappropriate, thus data from each application were analysed separately. Nutrient concentrations from RD2 were analysed independently using a one-way analysis of variance (ANOVA). The effect of spreading method at the second and third runoff events (RD9 and RD28) at each application could not be analysed similarly, as these measurements were not independent from previous runoff day events. Consequently, the runoff generation data and P export from RD2, RD9, and RD28, were analysed using a repeated measures analysis (REML) to take account of three rainfall events. An antedependence order 1 correlation model was fitted to the REML analysis to account for differences between the three runoff day events. Cumulative P export following each application was calculated as the sum of P export from all three runoff day events (RD2, RD9, RD28). This was also analysed using a one-way ANOVA.

RESULTS

Throughout the experimental period (25 May - 21 July) average daily air temperature was 14.3^{o}C. Rainfall totals during each 28-day period following the first and second slurry application (25 May - 23 June and 22 June - 21 July) were 50.8 and 246.3 mm, respectively (Figure 1). During the same periods, average VSM contents were 31.6 and 37.9% for slurry application 1 and 2 (Stubble and Regrowth), respectively. On the day of slurry application VSM content was 46.4% and 34.1% for the Stubble (25-26 May) and Regrowth treatments (22-23 June), respectively.

Runoff generation

Throughout the experiment there was no significant effect of spreading method on either the time taken to generate runoff or runoff volume (Table 1). Following the Stubble slurry application, the VSM content with the Splashplate spreading method was significantly lower (P<0.05) than for the Control or Trailing shoe DOWN treatments. There was no effect of spreading method on VSM content with the Regrowth application. Following the Stubble slurry application, VSM and runoff volumes were greatest (P<0.001) at runoff day (RD) 2 with the time required to generate runoff at RD2 (298 seconds) significantly shorter (P<0.05) than for either RD9 (540 seconds) or RD28 (680 seconds). Runoff volumes were on average 5.6 times lower at RD9 and RD28, compared to RD2. Following the Regrowth slurry application, runoff generation time also increased with time since slurry application with RD9 and RD28 exhibiting significantly higher runoff generation times (717 and 813 seconds, respectively) than RD2 (478 seconds; P=0.007). There was no significant Spreading method x Runoff day interaction for either VSM content, runoff generation time, or runoff volume at either slurry application.

Runoff P concentrations at RD2

Spreading method had a significant effect (P<0.001) on flow-weighted mean concentrations (FWMC) of DRP in runoff at RD2 following the Stubble slurry application (Table 2). The Splashplate spreading method exhibited significantly higher (P<0.05) DRP concentrations in runoff (1.75 mg/l) than both the Injection (0.93 mg/l) and Trailing shoe (1.10 mg/l) spreading methods. In addition, all slurry treatments, with the exception of the Injection DOWN treatment, exhibited significantly higher (*P*<0.005) DRP concentrations than the Control (0.53 mg/l). Spreading method also had a significant effect (*P*<0.001) on PP and TP concentrations in runoff at RD2. The five treatments that received slurry (Injection ACROSS, Injection DOWN, Splashplate, Trailing shoe ACROSS, Trailing shoe DOWN) exhibited significantly higher PP (average 3.99 mg/l) and TP (average 5.62 mg/l) concentrations in runoff than the Control, which exhibited PP and TP concentrations in runoff of 0.67 mg/l and 1.12 mg/l, respectively. Of the five treatments that received slurry, numerically runoff PP concentrations were highest from the Splashplate treatment (4.28 mg/l) however this was only significantly different from the Injection DOWN treatment (3.49 mg/l). In contrast, runoff TP concentrations were significantly higher from the Splashplate treatment (6.66 mg/l) than the Injection ACROSS (5.07 mg/l) or DOWN (4.66 mg/l) treatments. Spreading method had no significant effect (*P*>0.05) on DUP concentrations in runoff at RD2. Likewise, there was no significant effect (*P*>0.05) of spreading method on DRP, DUP, PP or TP concentrations in runoff at RD2 following the Regrowth slurry application.

Phosphorus exports over time

Over the course of the three RD events following Application 1 (Stubble) there was a significant Spreading method x Runoff day interaction for both TP (*P*=0.033) and PP (*P*=0.026) export (Repeated measures analysis; Figure 2). At RD2, TP export was greatest from the Splashplate (0.514 kg/ha) treatment followed by the Trailing shoe (ACROSS = 0.463 kg/ha; DOWN = 0.467 kg/ha) and the Injection (ACROSS = 0.384 kg/ha; DOWN = 0.293 kg/ha) treatments. Runoff TP exports were significantly greater (*P*<0.05) from the Splashplate and Trailing shoe (ACROSS and DOWN) treatments than the Injection (ACROSS and DOWN) treatments. In addition, all five treatments that received slurry (Injection ACROSS, Injection DOWN, Splashplate, Trailing shoe ACROSS, Trailing shoe DOWN) exhibited significantly higher TP exports than the Control (0.068 kg/ha^{-}treatment at RD2. At RD9, the Injection DOWN treatment exhibited significantly higher TP export than all other treatments (with the exception of the Splashplate treatment) however by RD28 there was no significant difference in TP exports between treatments.

Runoff PP exports were significantly greater (*P*<0.05) from all five treatments that received slurry (average = 0.298 kg/ha) than the Control (0.040 kg/ha) treatment at RD2 following the Stubble application. However, at RD9 and RD28 there was no significant effect of treatment on PP export. In addition, there was no significant (*P*>0.05) Runoff day x Spreading method interaction for DRP or DUP export following the Stubble application.

Irrespective of spreading method, exports of DRP, DUP, PP and TP at RD2 were significantly greater (*P*<0.001) than those measured at either RD9 or RD28 following the Stubble application. In contrast, at the Regrowth application there was no significant (*P*>0.05) Runoff day x Spreading method interaction for DRP, DUP, PP or TP export (Figure 3). Phosphorus exports did not differ significantly over the three runoff days following this application.

When P exports from all three RS events following the Stubble slurry application were totalled, the five treatments that received slurry (Injection ACROSS, Injection DOWN, Splashplate, Trailing shoe ACROSS, Trailing shoe DOWN) exhibited significantly higher (*P*<0.05) cumulative exports of PP and TP in runoff than the Control treatment (Table 3). There was no significant effect of Spreading method on the cumulative exports of DRP and DUP following the Stubble application. Likewise, following the Regrowth application there was no significant effect of Spreading method on cumulative exports of DRP, DUP, PP or TP.

Stubble vs. Regrowth

While not compared statistically, across all spreading methods and runoff events DRP, DUP, PP and TP concentrations in runoff were numerically 58, 35, 57 and 53% lower with the Regrowth application in comparison to the Stubble application. Consequently export rates of DRP, DUP, PP, and TP, were also 92, 87, 96 and 95% lower respectively at the Regrowth application in comparison with those recorded from the Stubble slurry application.

DISCUSSION

Factors affecting runoff generation

Evidence regarding the impact of slurry application on runoff is conflicting. In long term studies the addition of manure to soils has been found to increase soil organic matter content, facilitating an increase in soil aggregate stability and leading to reduced runoff volumes (Kleinman and Sharpley, 2003). At a shorter timescale, Smith *et al.* (2001) suggested that broadcast cattle slurry application may actually increase the propensity for runoff generation by slurry sealing the soil surface however, this was not the case with the Splashplate treatment in the present study. Indeed the absence of a significant effect of spreading method on runoff generation in the current experiment is in agreement with findings from other researchers investigating spreading techniques under both natural and artificial rainfall e.g. Johnson *et al.* (2011), Srinivasan *et al.* (2007), and Uusi-Kamppa and Heinonen-Tanski (2008).

In the current experiment weather and sward conditions at each runoff date were most likely the predominant factors influencing runoff generation. For example, runoff volume was highest at RD2 (Stubble application), corresponding to the highest VSM content during the experiment. The rapid fall in VSM content observed between RD2 and RD9 (Stubble), most likely resulted in the lower runoff volume seen at RD9. In addition, visible cracking of the soil surface was evident at RD9, and the rapid downward movement of water through these cracks most likely facilitated infiltration and discouraged runoff generation at this time (Haygarth and Jarvis, 1999; Wilcock, 1997). With the Regrowth application, interception of rainfall by the grass sward is likely to be the main reason for the lower runoff volumes observed relative to the Stubble application, however, drier conditions also prevailed at application facilitating greater infiltration of slurry. Interception of rainfall by vegetation cover has been found to reduce the kinetic energy of rainfall and increase the level of resistance presented to surface water movements, thus encouraging infiltration of rainwater and reducing the volume of runoff (Loch, 2000; Pan and Shangguan, 2006; Turnbull *et al.*, 2010).

Effect of slurry spreading method on P losses in runoff

The higher DRP concentrations observed in runoff from the Splashplate treatment following the Stubble application (RD2) are most likely a result of the larger slurry-rainfall contact area associated with this technique. While slurry occupied more than 80% of the surface area on the Splashplate plots, slurry covered less than 20% and 10% of the surface area on the Trailing shoe and Injection treatment plots, respectively. This most likely resulted in a greater rate of slurry-water mixing and an increased likelihood of the dissolution and solubilisation of slurry P occurring on the Splashplate plots, resulting in elevated DRP concentrations in runoff. In contrast, with the Trailing shoe and Injection techniques a reduction in contact area, cumulative raindrop impact, and interaction time, is likely to have limited the potential for dissolution of P from slurry to runoff. While no previous studies appear to have examined the effect of trailing shoe slurry applications on P loss from grassland, work done by van Vliet *et al.* (2006) and Johnston *et al.* (2011) examined P losses associated with band spreading over aeration slots. Both of these studies observed much higher reductions in DRP export (83% and 95%, respectively) relative to splashplate spreading, compared to that observed in the current experiment. This was most likely due to the displacement of slurry into the soil via the aeration slots (8-16 cm deep).

Similar reductions in DRP concentrations in runoff were observed with the Injection and Trailing shoe treatments compared to the Splashplate treatment at RD2 (proportionally 0.53 and 0.63, respectively) suggesting that the cutting of soil in the Injection treatments had no significant influence on initial DRP concentrations. Other authors e.g. Johnson *et al.* (2011) and Uusi-Kamppa and Heinonen-Tanski (2008) have noted decreases in DRP concentrations in runoff (74 and 86%, respectively) with Injection relative to Splashplate spreading techniques. These reductions are greater than that observed in current study due to the placement of slurry deeper into the soil (>0.1 m) in their experiments, reducing the exposure of slurry to rainfall more so than in this experiment (depth = 0.03 m).

While elevated levels of PP might have been expected as a result of increased soil disturbance with the Injection treatment compared to the other application methods, no such effect was observed. This may be due to slurry covering the disturbed soil at

RD2 thus preventing the entrainment of soil P particles (McDowell and Sharpley, 2002). In fact, Kleinman *et al.* (2002) observed a significant reduction in suspended sediment concentrations in runoff from plots of bare soil following the application of liquid manures. In addition, the absence of a significant difference in PP export between the Splashplate, Trailing shoe and Injection treatment was unexpected. Slurry spread using a splashplate will be subject to a greater cumulative raindrop impact than slurry spread using either injection or trailing shoe techniques. This, due to the physical breakdown of slurry aggregates by raindrop impact, and the exposure of previously occluded P (Leinweber *et al.*, 2002; Nash and Butler, 2011), might be expected to result in elevated losses of slurry derived PP from the Splashplate treatment. However, it may be the case that the terminal velocity of raindrops, supplied by the drip-type rainfall simulator used in this experiment, was not sufficient to fully facilitate the detachment and transport of slurry particulates (Kinnell, 2005). This highlights the need to evaluate P losses from these spreading methods at a larger scale under natural rainfall conditions.

The significantly lower TP export at RD2 from the Injection technique relative to the Splashplate and Trailing shoe techniques witnessed is most likely due to the smaller amount of exposed slurry present on the soil surface in comparison to the Splashplate or even Trailing shoe techniques, causing a reduction in slurry-rainfall contact, and resulting in a lower rate of transfer of slurry P to runoff.

In the current study, the contribution of PP to the total amount of exported P decreased from 66% at the Stubble slurry application to 40% at the Regrowth application. This reduction is likely due to interception of rainfall by the grass sward reducing the terminal velocity and erosive power of the rainfall (Self-Davis *et al.*, 2003), and may have disguised some of the effects of spreading method on P losses in runoff following the Regrowth application. Sward cover can be effective at reducing particulate P losses in runoff, whose entrainment is dependent on the available energy in runoff, which in turn is dominated by raindrop energy and surface roughness (Heathwaite, 1997; Prosser *et al.*, 1995). Likewise, the numerically lower DRP concentrations between the Stubble and Regrowth treatments can be attributed to a reduction in rainfall-slurry interactions arising from a decrease in the effective depth of interaction of rainfall (often associated with increasing sward cover)

(Sharpley, 1985). A similar effect was demonstrated by Ahuja *et al*. (1982), who found that the relative kinetic energy of raindrops diminished with an increase in sward cover, resulting in lower DRP concentrations in runoff. Consequently, the effect of vegetation cover at the time of slurry application should be investigated as a potential management option for reducing P concentrations in runoff.

Effect of ACROSS slope vs. DOWN slope slurry applications

The ACROSS and DOWN slope treatments were included to examine if direction of slurry application had an effect on nutrient losses. For example, with slurry application techniques that involve soil disturbance, such as the shallow injection, applying slurry along the direction of the slope might be expected to facilitate nutrient movement through the creation of preferential flow channels at the soil surface (Maguire *et al.*, 2011). However, in the current grassland study there was no evidence that either P concentration or export were significantly affected by direction of slurry application for either the Injection or Trailing shoe treatment. Nevertheless, at RD9 (stubble application) DRP and TP exports from the Injection DOWN treatment were numerically 3.55 and 3.89 times higher than for the Injection ACROSS treatment. Visual observations of runoff generation on the Injection DOWN plots showed that the Injection slits did appear to facilitate water movement in contrast to the ACROSS injection slots, and in essence became preferential flow pathways for runoff. Consequently, further work is needed to evaluate this at a larger scale.

Phosphorus losses over time

Across all the Stubble treatments, DRP and TP exports were 81.2 and 93.2% lower at RD9 and 87.1 and 86.8% lower at RD28, respectively, compared to losses at RD2. These reductions in P export rates may be attributed to a number of factors, including the decrease in runoff volume associated with changes in soil conditions, and increasing herbage covers. In addition, the availability of manure P will have declined over time due to the dissolution of P into rainfall and its removal in runoff at previous runoff events, and the subsequent infiltration of slurry into the soil over time. For example, Kleinman and Sharpley (2003) observed a negative correlation between runoff DRP and TP concentrations, and time since slurry application over three successive rainfall events at 3, 10 and 24 days post application. Similarly, infiltration of slurry during the first four days after application has been shown to account for up to 60% of total slurry P (Vadas, 2006; Vadas *et al.*, 2007), and once infiltrated, soil sorption processes rapidly to reduce the amount of runoff-associated P transfer. In addition, visible changes in the consistency of the applied slurry were evident over time, with the slurry adopting elastic and hydrophobic tendencies as conditions became drier. Other authors (McDowell and Stewart, 2005) have found that the air-drying of slurry can result in the conversion of readily available labile pools of P in slurry into residual pools of P, which are less prone to removal by water. These changes have been found to increase the number of hydrophobic surfaces within slurry, leading to low amounts of water extractable P and visible surface water repellency. This may also account for a decrease in DRP loss over the three successive rainfall events.

With the Regrowth application, P losses were much lower at RD2, and as such the decrease in losses with subsequent events was much less than with the Stubble application. While all of the factors mentioned above will have contributed to this decline, the predominant factor is likely to be the increased interception of rainfall as a result of the much greater sward cover.

The absence of a significant effect of treatment on cumulative (RD2 + RD9 + RD28) P export following the Stubble application suggests that although the Splashplate treatment results in a greater magnitude of P loss in runoff immediately after the event, over time P export is unaffected by spreading method. Consequently, the duration of the P signal in runoff from each of these treatments should be investigated.

CONCLUSIONS

The results of the current experiment demonstrate that both Shallow injection and Trailing shoe techniques reduce DRP concentrations in runoff relative to traditional Splashplate spreading techniques immediately after slurry application. While these benefits were not observed when slurry was applied to herbage following a four-week regrowth period, runoff volumes were low following dry weather conditions at the time of application, while the higher sward covers may also have impacted on P concentrations in runoff. Nonetheless, the results of this experiment suggest that the application of slurry with the trailing shoe or injection spreading technique can reduce the magnitude of P loss in runoff following slurry application, and in combination with the other proven advantages of these techniques on grassland they should be considered for future slurry management strategies. They have a particular role to play within intensive grassland systems, such as in NI, where agriculture presents a threat to water quality through the transfer of P in surface runoff.

ACKNOWLEDGEMENTS

This work was co-funded by the Department of Agriculture and Rural Development for Northern Ireland and AgriSearch, Northern Ireland. The authors thank the staff of the dairy unit for technical assistance and members of the AFBI agriculture and freshwater laboratories for undertaking chemical analysis.

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Table 1: The effect of spreading method and runoff day on volumetric soil moisture content, runoff generation time, and the volume of runoff following slurry application to a grass Stubble and a four week Re-growth

		Stubble			Re-growth		
		Volumetric soil moisture content (%)	Runoff generation time (sec)	Runoff volume[†] (litres)	Volumetric soil moisture content (%)	Runoff generation time (sec)	Runoff volume[†] (litres)
Spreading method	Control	39.3[b§]	352	1.56	37.2	693	0.13
	Injection ACROSS	38.8[ab]	499	1.68	39.9	607	0.35
	Injection DOWN	37.0[ab]	394	2.05	39.1	693	0.31
	Splashplate	36.3[a]	485	1.89	38.5	812	0.40
	Trailing shoe ACROSS	36.9[ab]	780	1.74	35.6	461	0.45
	Trailing shoe DOWN	38.9[b]	525	1.61	36.6	751	0.28
	SED[#]	1.26	160.2	0.435	1.72	168.0	0.166
	P	0.016	NS	NS	NS	NS	NS
Runoff day	2	45.1[c]	298[a]	3.89[b]	35.9[a]	478[a]	0.30[ab]
	9	25.0[a]	540[b]	0.60[a]	35.7[a]	717[b]	0.18[a]
	28	34.1[b]	680[b]	0.78[a]	41.8[b]	813[b]	0.48[b]
	SED	0.81	113.2	0.307	1.22	119.4	0.118
	P	< 0.001	0.004	<0.001	<0.001	0.007	0.046
Method X Runoff[‡]	*SED*	2.19	276.3	0.751	2.98	289.0	0.288
	P	NS	NS	NS	NS	NS	NS

[†] Total volume produced per 0.5 m^2 plot within the first 30 minutes from commencement of runoff
[‡] Spreading method x runoff day interaction
[§] Within each factor, values with the same letter within each column, are not significantly different (P>0.05)
[#] Standard error of difference

Table 2: Effect of spreading method on flow-weighted mean concentrations of dissolved reactive phosphorus (DRP), dissolved unreactive phosphorus (DUP), particulate phosphorus (PP), and total phosphorus (TP) in runoff at runoff day-2, following slurry application to a grass Stubble and a four week Re-growth

Spreading method	Stubble (mg/l)				Re-growth (mg/l)			
	DRP	DUP	PP	TP	DRP	DUP	PP	TP
Control	0.53[a†]	0.15	0.67[a]	1.12[a]	0.15	0.08	0.97	1.21
Injection ACROSS	1.10[b]	0.42	3.68[bc]	5.07[b]	0.37	0.26	0.93	1.56
Injection DOWN	0.75[ab]	0.44	3.49[b]	4.66[b]	0.48	0.56	2.26	3.31
Splashplate	1.75[c]	0.60	4.28[c]	6.66[c]	0.23	0.21	0.32	0.77
Trailing shoe ACROSS	1.06[b]	0.51	3.76[bc]	5.34[bc]	0.67	0.37	1.76	2.84
Trailing shoe DOWN	1.14[b]	0.48	4.76[c]	6.39[c]	0.33	0.36	1.47	2.17
SED[‡]	0.207	0.240	0.386	0.534	0.1446	0.139	0.919	1.152
P	<0.001	NS	<0.001	<0.001	NS	NS	NS	NS

[†] Within each factor, values with the same letter within each column, are not significantly different (P>0.05)
[‡] Standard error of difference

Table 3: Effect of spreading method on the cumulative (RD 2 + RD 9 + RD 28) exports of dissolved reactive phosphorus (DRP), dissolved unreactive phosphorus (DUP), particulate phosphorus (PP), and total phosphorus (TP) in runoff following slurry application to a grass Stubble and a four week Re-growth

Spreading method	Stubble (kg)				Re-growth (kg)			
	DRP	DUP	PP	TP	DRP	DUP	PP	TP
Control	0.040	0.004	0.108[a†]	0.188[a]	0.001	0.001	0.053	0.080
Injection ACROSS	0.105	0.028	0.352[b]	0.502[b]	0.007	0.003	0.077	0.156
Injection DOWN	0.106	0.069	0.312[b]	0.540[b]	0.009	0.008	0.166	0.294
Splashplate	0.174	0.060	0.410[b]	0.696[b]	0.013	0.005	0.048	0.134
Trailing shoe ACROSS	0.112	0.046	0.402[b]	0.608[b]	0.010	0.014	0.128	0.236
Trailing shoe DOWN	0.115	0.035	0.458[b]	0.660[b]	0.008	0.002	0.084	0.150
SED[‡]	0.0360	0.0228	0.0746	0.1106	0.0047	0.0049	0.0527	0.0812
P	NS	NS	0.004	0.005	NS	NS	NS	NS

[†] Values with the same letter within each column, are not significantly different (P>0.05)
[‡] Standard error of difference

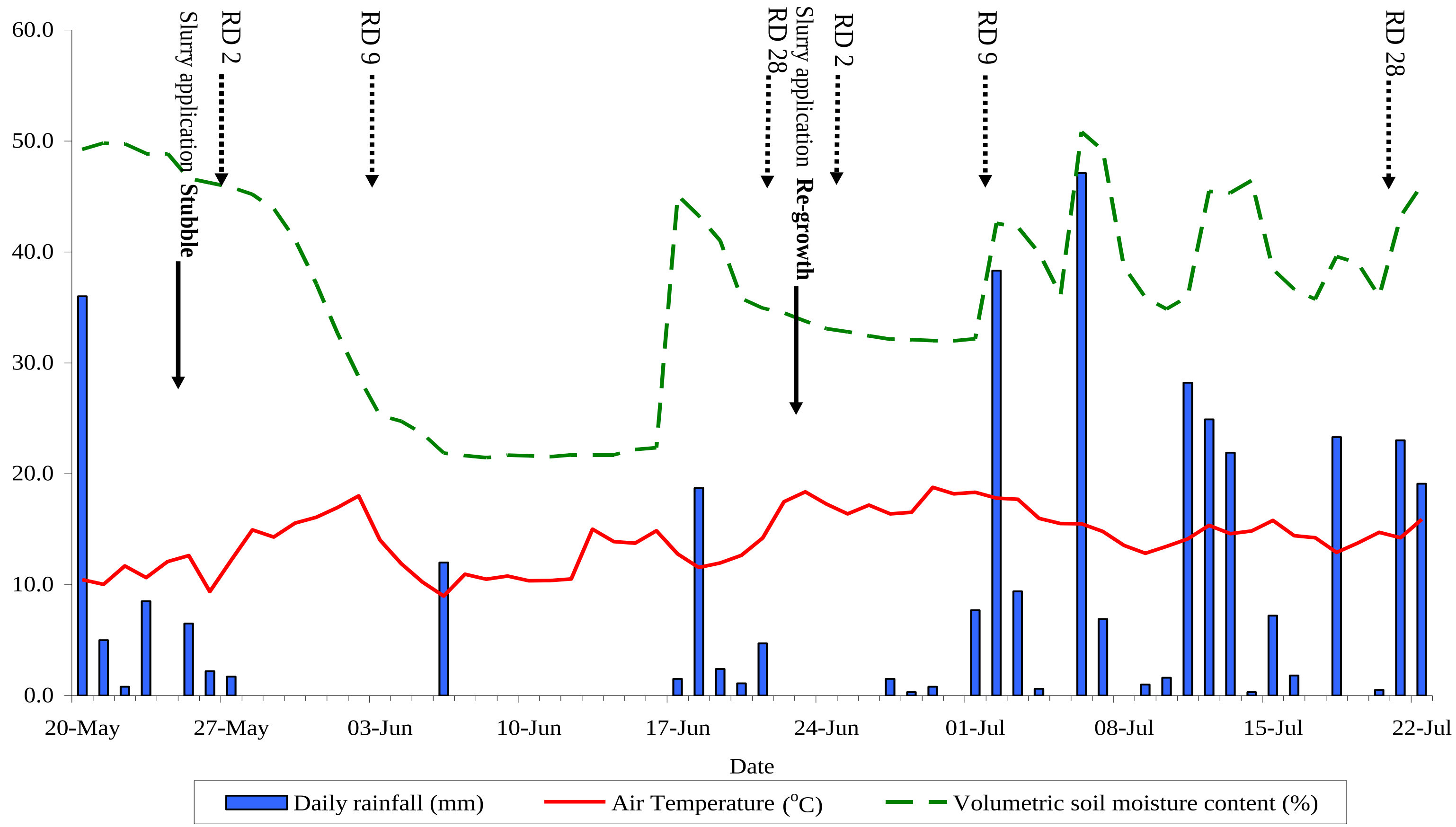

Figure 1: Daily rainfall, average daily air temperature and volumetric soil moisture content during the experimental period (20 May - 2 July 2009), with details of slurry application and rainfall simulation (RD) events superimposed

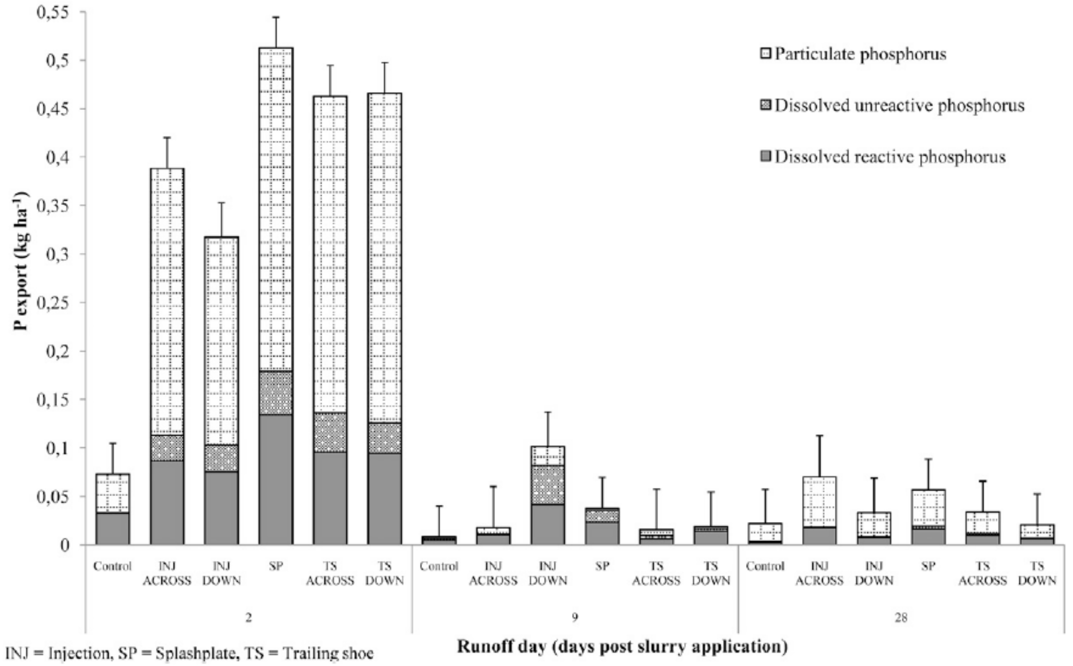

Figure 2: Effect of spreading method and day of rainfall simulation (RD) on P export rates following slurry application to a grass stubble

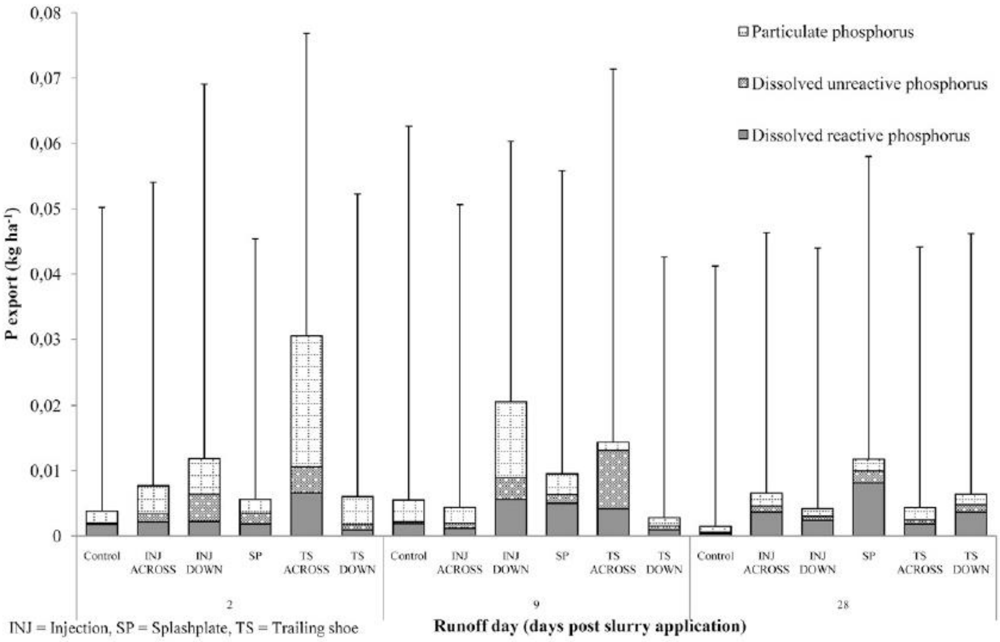

Figure 3: Effect of spreading method and day of rainfall simulation (RD) on P export rates following slurry application to a four week grass regrowth

Small Scale Experiment 2

The impact of herbage regrowth interval on phosphorus losses in runoff post slurry application

ABSTRACT

Low emission slurry spreading techniques such as the trailing shoe are known to increase nitrogen use efficiency and to increase the available window for slurry application, whilst minimising sward contamination. However, little is known about the ability of the trailing shoe to mitigate against incidental phosphorus (P) losses, especially following slurry application to mature swards. This 3 x 2 randomised block design experiment investigates the impact of sward cover on P losses in runoff following application of manure via simulated trailing shoe apparatus. Treatments comprised of three stages of grass growth (0, 10 and 20-day regrowths; equivalent to 1550, 2050 and 2900 kg DM/ha, respectively), and two slurry application rates: 25 m^3/ha (Slurry) and 0 m^3/ha (No slurry). Runoff was generated at day 2, 9 and 16 post slurry application using a drip type rainfall simulator. Throughout the experiment P concentrations in runoff were significantly greater from the Slurry treatments than from the No Slurry treatments ($P<0.05$). When runoff was generated two days post slurry application, dissolved reactive P concentrations in runoff were significantly lower ($P<0.05$) with the 10 and 20-day regrowth treatments, compared to the 0-day regrowth treatment. In contrast, herbage regrowth had no significant effect ($P>0.05$) on particulate P concentrations in runoff. This experiment demonstrated that allowing a grass sward to recover for at least 10 days post harvest provides a potential mitigation strategy to reduce P losses in runoff following slurry application.

INTRODUCTION

Over the past half century the intensification of agricultural systems has posed a significant threat to freshwater ecosystems across the world. Diffuse phosphorus (P) losses from agricultural land have contributed to the eutrophication of freshwater systems, with a decline in surface water quality evident in many areas of intensive agricultural production (Carpenter *et al.*, 1998). Within Northern Ireland (NI) agriculture has been identified as the primary driver of P-induced eutrophication in surface waters, accounting for almost 60% of terrestrial P inputs to inland water systems (Smith *et al.*, 2005). To address this issue a number of management strategies have been implemented in NI including: the adoption of a crop requirement approach to P fertiliser usage (Maguire *et al.*, 2009), a reduction of the P content of animal diets (McCann *et al.*, 2007, Ferris *et al.*, 2010), and controls on the application of organic manures to agricultural land (EHS, 2006). In view of the recent rapid decline in the use of inorganic P fertilisers (DARD, 2011), the application of organic manures by both grazing animals and slurry spreading now represents the primary input of P to most agricultural soils. Hence the identification of strategies which minimise the magnitude of P loss following the spreading of animal slurry is a crucial component in tackling P-driven eutrophication in waterways in NI.

Manipulation of vegetation cover, whether through the use of edge of field buffer strips or the introduction of cover crops to cultivated arable soils, is a popular management strategy for the reduction of nutrient loads in surface runoff (Dorioz *et al.*, 2006, Stevens and Quinton, 2009). The dense vegetation cover and high roughness coefficient in buffer strips often results in a reduction in runoff velocity which promotes the infiltration of runoff water, the settling of particulate matter, and the potential sorption of dissolved nutrients to the upper soil horizons (Deletic, 2001, Dorioz *et al.*, 2006). In addition, sward interception of rainfall results in a decrease in both rain-splash erosion and the physical detachment of nutrient rich soil particulates (Prosser *et al.*, 1995). Whilst the effects of both buffer strips and cover crops have been extensively investigated, the effect of level of sward cover on nutrient losses from grassland systems has received little attention despite the identification of grasslands as a significant source of both dissolved and particulate P (Bilotta *et al.*, 2008, Heathwaite *et al.*, 1990). Moreover, the interactions between sward cover, organic manure applications to grassland systems and nutrient losses, have been poorly investigated. Although Butler *et al.* (2006) identified significant reductions in P export in runoff from grassland in comparison with bare ground following slurry application, the interaction between different levels of sward cover and slurry application remains unclear.

This interaction is increasingly relevant as low emission slurry spreading techniques, which facilitate slurry application to a wide range of vegetation covers has become commonplace on grassland systems. Slurry spreading via the traditional broadcast (splashplate) technique normally takes place immediately after a harvest or grazing events so as to minimise contamination of the grass sward (Carter *et al.*, 2010). In contrast, the trailing shoe technique places slurry on the soil surface below the grass canopy, thus allowing the application of slurry to fields with greater levels of sward cover with minimal sward disturbance. Although low emission slurry spreading techniques such as the trailing shoe are known to improve nitrogen (N) use efficiency through the reduction of gaseous emissions (Frost *et al.*, 2007, Webb *et al.*, 2010), it has been suggested that they may also reduce P losses in runoff following slurry spreading (Maguire *et al.*, 2011, Sharpley *et al.*, 2004). A recent study by McConnell *et al.* (2012) comparing the post-harvest application of slurry using the trailing shoe and splashplate techniques demonstrated a significant reduction in dissolved reactive phosphorus (DRP) losses in runoff with the trailing shoe system. However, it remains unclear if these reductions can be further increased by applying slurry with the trailing shoe technique to swards with greater levels of herbage cover. Thus the aim of this paper is to evaluate the effect of herbage cover on P losses in runoff following slurry application via the trailing shoe slurry spreading technique.

MATERIALS AND METHODS

Site description

This experiment was undertaken (8 August - 22 October 2010) at the Agri-Food and Biosciences Institute, Hillsborough, NI (54^{o}27'N; 06^{o}04'W). The 192 m^2 experiment area was located on a drumlin hill slope with an average slope of 4.5% and a north-easterly aspect. The soil type was classified as a Soil Water Gley Class 1 soil overlying Silurian Shale (DANI Soil Survey of Northern Ireland) (FAO classification: Dystric Gleysol). The area has a Hydrology of Soil Types (HOST) classification of 24 which is indicative of poorly drained soils and high runoff rates. This HOST classification accounts for 46% of the land area in NI (Higgins, 1997). The soil has an average Olsen P content of 57.7 mg/l, twice the agronomic optimum, and an average bulk density of 0.83 g/cm^3 in the 0-5 cm horizon. Field capacity for the field site was 40.1%. Average annual rainfall and average annual evapotranspiration recorded at the site equated to 890 mm and 524 mm, respectively, for the period 1971-2000 (Betts, 1997). The average annual duration of the growing season was 254 days for the period 1951-1990 (Betts, 1997). Daily meteorological data for the site are supplied by a United Kingdom Meteorological Office weather station located 300 m from the field site.

Site management

From 1985 onwards, 2-4 harvests of grass silage were removed from the field site each year with livestock grazing removing surplus grass during the winter months. The field was last reseeded in 1997 with perennial ryegrass (*Lolium perenne*) and has a history of high nutrient inputs in the form of chemical and organic fertilisers. From September 2009, livestock and machinery were excluded from the site to minimise the potential for 'hot spots' of compaction and/or high soil nutrient concentrations. From September 2009 until September 2010, nutrients were applied by hand as chemical fertilisers. Calcium Ammonium Nitrate was applied to the whole area twice during this period, on 12 April 2010 and on 5 July 2010, at a rate equivalent to 100 kg N/ha each time. Grass from the area was also harvested four times (22 September 2009, 29 October 2009, 12 April 2010 and 5 July 2010) during the period of animal exclusion. Grass was cut using a hand operated, self-propelled mower (3600BM, Agria, Möckmühl, Germany) and removed by hand.

Treatments

This 3 x 2 factorial design experiment examined three different stages of sward development: a zero-day regrowth (0-day RG), a ten-day regrowth (10-day RG), and a twenty-day regrowth (20-day RG), at two levels of slurry application: Slurry *vs*. No slurry (Control). Twenty-four 0.5 m^2 plots (0.5 x 1.0 m) were established in a randomised block design consisting of four 20.0 m^2 (10.0 x 2.0 m) blocks. Blocks consisted of six plots with plots spaced 2.0 m apart within each block. Blocks were separated by a three metre wide buffer zone located upslope of each block, with each block containing one replicate of each treatment.

Swards were managed so as to achieve the three regrowth intervals simultaneously on 30 August 2010. Initially, the plots associated with the 20, 10 and 0-day regrowths were cut by hand at 55, 45 and 35 days, respectively, prior to the planned slurry application date (30 August 2010) using battery-operated Gardina hand shears (Accu 6; Kress and Kastner, Weiterstadt, Germany). These plots were then cut 20, 10 and 0 days prior to slurry application to achieve the three levels of sward regrowth required. This double cutting cycle was necessary to ensure that the base of the sward had a similar sward structure within each of the three regrowth treatments.

Slurry collection and application

The macerated dairy cow slurry used in this experiment was collected from an above ground slurry store at the Agri-Food and Biosciences Institute (Hillsborough) four months prior to the start of the experiment. The slurry had an average dry matter (DM) content, pH, total N, and total P content of 81.2 g/kg, 7.43, 2.27 g/kg and 0.58 g/kg, respectively. Once the slurry was collected, it was stored in high density polyethylene (HDPE) containers at 4^{o}C.

With the slurry treatments, slurry was applied by hand to each of the three regrowth interval treatments on the 30 August 2010, at a rate of 25 m^3/ha using a simulated trailing shoe spreading technique. The slurry was placed in three 0.9 m long tramlines, spaced 0.2 m apart and parallel to the slope. No tramlines were situated within 0.05 m of the plot sides thus minimising any possible edge effects. To simulate the trailing shoe spreading technique, the grass sward was parted and held in place using two 1.0 m long wooden boards. Slurry was placed at the base of the sward using a long spouted pouring jug, and the wooden boards then removed allowing the sward to return to its natural position.

Plot description

Each plot was hydrologically isolated from surrounding runoff using three stainless steel surrounds which were inserted 0.05 m into the soil. A V-shaped stainless steel runoff collection tray (0.5 x 0.1 x 0.1 m) was inserted 0.03 m below the soil surface at the down-slope end of each sub-plot, perpendicular to the slope. Each collection tray had a 0.01 m shelf which was pushed into the soil beneath each plot (approximately 0.05 m), parallel to the soil surface. The runoff collection trays were connected to a two litre HDPE plastic collection bottle by a 0.5 m length of braided PVC pipe buried underground. Collection trays were inserted two weeks before slurry application to minimise disturbance to the soil at time of application. The stainless steel surrounds were inserted immediately following slurry application. Plots were lightly covered with translucent plastic sheeting for 48 hr following slurry application to prevent any rainfall reaching the plots.

Rainfall simulation

Rainfall simulations were performed on day 2 (RD2), 9 (RD9) and 16 (RD16) post slurry application. Two Amsterdam drip-type rainfall simulators, as described by Bowyer-Bower and Burt (1989), were employed to supply rainfall at a constant rate of 40 mm/hr to each plot. This rainfall intensity has a return period equivalent to a one in fifty year event for NI (Cruickshank *et al.,* 1997). During simulations, wooden boards (1.2 m^2) were placed along two sides of the rainfall simulators to act as a wind shield, thus preventing water droplets being blown outside the plot boundaries. Twenty minutes of runoff was collected at each rainfall simulation in 2 x 10 minute fractions. Runoff volume and time taken to generate runoff was recorded. Water used in the rainfall simulations was passed through a DC9 general deionising cylinder (DC9, Purite Ltd, Thames Oxon, UK) to reduce its P concentration. The cylinder delivered deionised water with an average dissolved reactive P (DRP) and nitrate-N concentration of 8.7 μg/l and 212 μg/l, respectively.

Water quality analysis

Runoff water samples were refrigerated at 3°C within 4 hr of sampling and analysed for DRP, total dissolved P (TDP) and total P (TP) within 24 hr of sampling. Samples for DRP and TDP were filtered through 0.45 μm filters (MF-Millipore, Billerica, MA) before analysis. Dissolved reactive P was determined by the acidic molybdate-ascorbic acid method of Murphy and Riley (1962). Total dissolved P and TP were determined by digestion with potassium persulphate and sulphuric acid, followed by analysis of the digest as outlined above for DRP (Eisenreich *et al.*, 1975). Particulate P (PP) was calculated as the difference between TP and TDP. Dissolved unreactive P (DUP) was calculated from the difference between TDP and DRP.

Sward measurements

On the day of slurry application, ten sward height measurements were taken randomly across each plot, using a rising plate meter (Folding pasture meter, Jenquip, New Zealand). Compressed sward height measurements were subsequently converted to kg herbage DM/ha using the equation Herbage DM = grass height (cm) * 316 + 330. Ten extended tiller height readings were also taken at random within each plot on the day of slurry application, and at each subsequent rainfall simulation event using a hand ruler.

Statistical analysis

Data in this experiment were analysed using Genstat v 12.1 (VSN International Ltd, 2009, UK). Flow-weighted mean nutrient concentrations and total nutrient export quantities were calculated using the two ten-minute fractions of runoff collected. Runoff generation and nutrient concentration data were analysed as a factorial design experiment using repeated measures analysis to account for the three RD events. An autoregressive order 1 correlation, which assumes an uneven effect between plots of previous RD events, was used. Pairwise comparisons of nutrient concentrations in runoff were also undertaken using Genstat v 12.1 to examine individual differences between treatments at each RD event.

RESULTS

Throughout the experiment (10 August 2010 - 25 October 2010) the total rainfall, average daily air temperature, and average volumetric soil moisture (VSM) content were 191.2 mm, 11.7^{o}C and 31.7%, respectively (Figure 1). On the day of slurry application (30 August 2010) 0.9 mm of rainfall and a VSM content of 28.6% were recorded. During the 16-day period following slurry application, which included the three runoff day (RD) events, the total rainfall, average daily temperature, and average VSM content were 70.5 mm, 14.1^{o}C and 31.7%, respectively. At RD2, 9 and 16, the total daily rainfall was 0.0, 0.4 and 4.2 mm, respectively, while average VSM content was 28.4, 32.7 and 36.6%, respectively.

At the time of slurry application the herbage mass on the 0-day, 10-day and 20-day RG treatment plots, was 1626, 2058 and 2860 kg DM/ha (above ground level), respectively (Table 1). Similarly, extended tiller lengths were significantly different (P<0.001) between the three regrowth treatments, averaging 5.1, 15.5 and 23.9 cm for the 0, 10 and 20-day regrowths, respectively. The effect of treatment (0-day, 10-day, and 20-day RG) on extended tiller length remained significant at RD16 (21.2, 24.6 and 29.7 cm, respectively; SED 0.76; P<0.05).

Throughout the experiment the Slurry treatments exhibited significantly lower runoff volumes (P<0.001) and longer runoff generation times (P=0.011) than the No Slurry treatments (Table 2). There was no significant effect of regrowth interval on either the time taken to generate runoff or runoff volume throughout the experiment. In contrast, runoff day had a highly significant effect (P<0.001) on both the time taken to generate runoff and the total volume of runoff produced during the 20-minute collection period. The smallest runoff volume (average = 0.27 litres) was recorded at RD2 when the average time taken to generate runoff was 1193 seconds. Runoff volume at RD9 was significantly greater (P<0.05) than at RD2, although runoff generation times were not statistically different. In contrast, a significantly (P<0.05) greater runoff volume and shorter runoff generation time was recorded at RD16 in comparison to RD2 and RD9. Runoff:rainfall ratios also differed numerically between RD2 (0.02), RD9 (0.05) and RD16 (0.08) and reflect the changes in runoff generation time and runoff volumes outlined above.

Flow-weighted mean concentrations (FWMC) of DRP (P<0.001), PP (P = 0.012) and TP in runoff were significantly greater (P<0.001) from the Slurry treatment than from the No Slurry treatments throughout the experiment (Table 3). Regrowth interval also had a significant (P<0.001) effect on the concentration of DRP in runoff, with DRP concentrations in runoff from the 0-day RG treatment greater (P<0.05) than from either the 10-day or 20-day RG treatments. In contrast, regrowth interval had no significant effect (P>0.05) on PP or TP concentrations in runoff. Runoff day had a significant effect on DRP (P<0.001) and TP concentrations in runoff, with concentrations decreasing between RD2 and RD16. A significant Slurry x Regrowth interaction was observed for both DRP (P=0.011) and PP (P=0.015) concentrations in runoff throughout the duration of the experiment. With the No Slurry treatments, neither DRP or PP concentrations differed with regrowth interval. With the Slurry treatment DRP concentrations decreased with regrowth interval, while PP concentrations followed a similar, although non-significant trend.

The relationship between Slurry treatment, regrowth interval and runoff day are presented in Figure 2. No significant Slurry x Regrowth x RD interaction was observed for DRP, PP or TP, despite the appearance of significant differences between treatments as indicated by the individual standard error bars in Figure 2. This difference was particularly evident within the Slurry treatment plots at RD2. To further examine this trend, a pairwise comparison analysis was used to evaluate the significance of differences between individual treatment means within each RD event (Table 4). Significant differences in DRP and TP concentrations in runoff occurred between regrowths from the Slurry treatments at RD2 (Table 4). Dissolved reactive P concentrations from the 20-day and 10-day RG treatments were 49% (P=0.049) and 75% (P=0.034) lower than from the 0-day RG treatment. Likewise, TP concentrations in runoff from the 20-day (P=0.007) and 10-day RG (P=0.015) Slurry treatments were lower than for the 0-day RG treatment at RD2. The difference between PP treatment means at RD2 was only weakly significant (P=0.092) for the 0-day – 20-day RG comparison. No significant difference (P>0.05) was evident between the Slurry or No Slurry 10-day and 20-day RG treatments for DRP, PP or TP concentrations in runoff at RD2. No significant differences (P>0.05) were evident within the other RD events between the three regrowth treatments for DRP, PP or TP concentrations in runoff with the exception of the Slurry plots at RD16. At RD16 the 0-day RG treatment exhibited significantly higher levels of DRP in runoff than the 10-day (P=0.006) and 20-day (P=0.003) RG treatments.

There was no significant effect (P>0.05) of slurry treatment on DRP, PP and TP exports in runoff, while DRP and TP exports were significantly higher (P<0.05) from the 0-day RG treatment in comparison to the 20-day treatment RG (Table 5). There was also a significant effect of runoff day on DRP (P=0.008), PP (P<0.001), and TP (P<0.001) export, with exports at RD9 and RD16 significantly higher than at RD2. A significant Slurry treatment x Regrowth interval interaction was observed for PP and TP export rates, with these interactions largely driven by a greater decrease in export rates between the 0-day RG and the 20-day RG treatments with the Slurry compared to the No slurry treatments.

DISCUSSION

Effect of slurry and regrowth interval on runoff generation

While there was a trend towards increased runoff generation time and reduced runoff volume with increasing regrowth interval, these effects were non-significant. In contrast, a number of authors have recorded lower runoff volumes from plots with dense vegetation covers in comparison to bare, or sparsely covered ground (Kleinman *et al.*, 2005; Borin *et al.*, 2005). Similarly, the progressive colonisation of slopes of bare soil by Buffel grass over a four-year period, reduced annual runoff volumes relative to plots of bare soil (Carroll *et al.*, 2000). However, in a study comparing four levels of Fescue - Dallisgrass vegetation cover (0, 45, 70 and 95% ground cover), while higher runoff volumes were observed from the bare ground treatment (0% cover) there was no significant difference in runoff volumes between the 45, 70 and 95% cover plots (Butler *et al.*, 2006).

Why the Slurry treatment exhibited lower runoff volumes relative to the No slurry treatment is unclear. However, over a longer time scale slurry application has been found to decrease runoff volumes, with the addition of organic matter to soil, increasing soil aggregation, and pore space and thus promotes the infiltration of rainfall (Haynes and Naidu, 1998, Gilley and Risse, 2000). However, the results of these processes are unlikely to have been evident within the timescale of the current experiment.

Effect of slurry and regrowth interval on runoff DRP

The results of studies examining the role of herbage cover on DRP losses in runoff from unmanured areas have been reported. For example, Sharpley (1985) observed that as the degree of vegetation cover increased, the effective depth of soil water interaction decreased. Through a reduction in the dissolution of soil P, this results in lower DRP concentrations in runoff (Sharpley, 1985; Ahuja *et al.*, 1982). In contrast, a number of field trials investigating the presence/absence of cover crops have found higher DRP concentrations in runoff from plots with cover crops in comparison to plots with no cover crops, with this increase having been attributed to an increased contribution of biomass P to runoff (Sharpley and Smith, 1991, Stevens and Quinton, 2009).

It is hypothesised that the elevated DRP concentrations in runoff from the No Slurry, 0-day RG treatment at RD2, relative to the 10-day RG or 20-day RG treatments, is due to the loss of plant soluble P stores being released following cutting of the grass sward two days prior to the RD event. Indeed, a number of authors have observed that extreme changes to the plants physiology, such as those which occur during freezing (Bechmann *et al.*, 2005) or grazing events (Miller *et al.*, 1994), have resulted in the release of biomass P to runoff. In an evaluation of the sources of P in runoff from cattle grazing, McDowell *et al.* (2007) attributed 20% of total P loss from grazed, manured pasture to plant associated P. Breakage of plant stems during the grazing process, and by trampling, can expose xylem and phloem cells, and release internal stores of P, much of which is stored as orthophosphate (McDowell *et al.*, 2007). In the study of McDowell *et al.* (2007) over half of all measured DRP losses from the plant-soil system were attributed to the mobilisation of plant P stores, following sward damage. In the current study, no plant associated-P is likely to have been released by the sward from the 10-day and 20-day RG No Slurry treatments at RD2 due to the time which had elapsed since cutting. This may explain the lower DRP concentrations observed in runoff from the 10-day and 20-day RG No Slurry treatments.

The current study has clearly demonstrated that applying slurry to higher herbage mass swards will reduce DRP concentrations in runoff. At RD2, DRP concentrations in runoff from the Slurry 10-day RG and 20-day RG treatments were reduced by 75 and 49%, respectively, in comparison to the 0-day RG treatment. Other authors have also observed that vegetation cover can impact on DRP concentrations in runoff. For example, Edwards *et al.* (2000) using simulated dung pats, observed lower orthophosphate concentrations in runoff from plots with a herbage height of 20 cm compared to plots with a herbage height of 2.5 cm. Likewise, in another study involving rainfall simulations on dung pats, Butler *et al.* (2006) recorded significantly lower DRP losses in runoff from plots with 70 and 95% grass cover in comparison to bare ground plots treated with a dung pat. Sward interception of rainfall can play a crucial role in reducing runoff DRP concentrations, with the sward reducing the kinetic energy of rainfall, thus lessening the impact of raindrops on the receiving soil and slurry surfaces (Butler *et al.*, 2006; Sharpley, 1985). Whilst soil P stores were likely protected from raindrop impact by a dense root structure in the current experiment, the exposed slurry was susceptible to aggregate breakdown and consequently enhanced slurry-water interaction. It is hypothesised that a greater level of slurry-water interaction ensures higher levels of slurry P dissolution, and the breakdown of complexes allows the release of previously occluded P for transport in runoff (Leinweber *et al.*, 2002). Therefore the higher levels of sward interception afforded by the 10-day RG and 20-day RG treatments reduced this slurry-water interaction and consequently reduced the concentrations of DRP in runoff. This is similar to the study of Ahuja *et al.* (1982), who observed reductions in runoff DRP concentrations with increasing levels of simulated vegetation cover.

Effect of slurry and regrowth interval on runoff PP

The extent of sward regrowth did not have a dramatic effect on PP losses in runoff in the current experiment, with only a weak difference ($P<0.1$) observed between the 0-day RG and 20-day RG treatments at RD2. Conflicting reports exist in the literature as to the effect of vegetation cover on PP concentrations in runoff. Edwards *et al.* (2000) examined the impact of repeated manure applications to swards with three different grass heights (2.5, 10 and 20 cm), and found no significant effect of grass height on PP concentrations in runoff. Likewise, differences in total Kjeldahl P concentrations in runoff in their experiment were limited to differences between bare soil plots and plots with vegetation, regardless of percentage vegetation cover. However, when considering the effectiveness of grass buffer strips over a range of studies, Dorioz *et al.* (2006) observed reductions of between 50-97% in PP losses in runoff, relative to plots without buffer strips. Similarly, at field and catchment scales, the introduction of cover crops within arable systems has been observed to consistently reduce PP losses in runoff by over 50% (Sharpley and Smith, 1991), and indeed this is becoming a popular mitigation strategy for reducing PP losses (Dorioz *et al.*, 2006).

The absence of a significant effect of herbage regrowth interval on PP concentrations in runoff in the current experiment is most likely a result of two factors, namely the issue of scale and the high percentage of soil surface covered by grass with each of the three treatments. The issue of scale can be divided into two components: the plot size, and the intensity of rainfall. McDowell and Sharpley (2002) noted that PP concentrations in runoff increased with an increase in plot length whilst, Sharpley and Kleinman (2003) consistently recorded higher concentrations of PP from 10.0 m long runoff plots compared to 1.0 m long plots. The higher runoff volumes and the greater runoff energy associated with larger plots facilitates greater entrainment of particulate matter, thus enhancing the transport of PP in runoff. Consequently, in small scale plot studies such as the current experiment, PP concentrations in runoff can often be under-represented, and as a result the potential effects of regrowth interval on PP concentrations in runoff may not have been fully expressed. In addition, drip-type rainfall simulators, as employed in the current experiment, often supply rainfall with terminal velocities considerably less than that of natural rainfall. This impacts both on the level of physical detachment of slurry particulates by rain-splash, and the amount of energy available for particulate transport in runoff (Kinnell, 2005).

A number of authors have identified threshold values of vegetation cover in relation to sediment and the associated PP loss in runoff. For example, Rogers and Schumm (1991) observed that a threshold sediment load is reached at a vegetation cover of 60-70%, above which sediment loss in runoff is no longer correlated with vegetation cover. Likewise, Carroll *et al.* (2000) reported that once buffel grass reached 80% soil coverage, total sediment loss in runoff became negligible. In the current experiment all three regrowth treatments were situated on a well-established ryegrass sward with a nominal cover of 100%, thus minimising the potential for particulate transport.

Effect of slurry and regrowth interval on runoff TP

Throughout the course of the experiment TP concentrations in runoff were dominated by the dissolved P fraction, with dissolved P accounting for 65% of TP concentrations in runoff across all treatments. Approximately 77% of dissolved P was present as DRP, and consequently the effects of slurry and regrowth interval on TP losses in runoff to a large extent reflects the effect of these factors on DRP concentrations in runoff, as described earlier. The decline in TP (and DRP) concentrations in runoff over the three rainfall events with the Slurry treatments is indicative of P having been removed during previous rainfall simulation events (Vadas *et al.*, 2011), and of P assimilation into the soil over time (Vadas, 2006). In addition, it may also be a function of the increasing grass cover between RD2 and RD16, particularly with the 0-day RG treatments.

Effect of slurry and regrowth interval on P export

Phosphorus export rate is the product of P concentrations in runoff and runoff volume. While there was a significant increase in DRP, PP, and TP concentrations in runoff from the Slurry treatments relative to the No slurry treatments in the current experiment, this was not reflected in a significant difference in their export rates. The absence of a significant difference can be explained by the lower runoff volumes observed from the Slurry treatment plots. The trend for decreasing DRP and TP export rates with increasing regrowth interval can also be attributed to both the decrease in plant associated P contribution and the high level of sward interception associated with the greater herbage masses. In addition, the numerically smaller (35%) runoff volumes from the 20-day RG may also have contributed to this effect.

While a significant Slurry x Regrowth interval interaction was present for exports of PP and TP in runoff throughout the experiment, the differences between treatments for these variables were not as transparent. While the numerically higher DRP, PP, and TP export from the 0-day RG Slurry treatment highlights the potential for higher herbage covers to reduce runoff P losses at the time of slurry application, the large variation in runoff volumes achieved from the treatment plots throughout the study may have prevented this from reaching significance.

CONCLUSIONS

Applying slurry to grassland, even with a trailing shoe system, results in increased DRP, PP and TP concentrations in runoff compared to unmanured grassland. However, when slurry was applied to either a 10 or 20-day herbage regrowth, DRP concentrations in runoff were reduced by 62% compared to a freshly cut sward. Particulate P loss was unaffected by herbage regrowth. The results from this study demonstrate that slurry application to higher sward covers can be considered as both a practical mitigation strategy for reducing P losses in runoff from agricultural grasslands, and good nutrient management practice for conserving nutrients in agricultural soils. However, further investigation of the benefits of using trailing shoe technology to reduce the risk of P loss following slurry application, is required at larger scales.

ACKNOWLEDGEMENTS

This work was co-funded by the Department of Agriculture and Rural Development for Northern Ireland and AgriSearch, Northern Ireland. The authors thank the staff of the Dairy Unit for technical assistance and members of the AFBI Agriculture and Freshwater laboratories for undertaking chemical analysis.

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Table 1: Average herbage mass and extended tiller heights recorded for the three regrowth intervals treatments (0, 10, and 20 days), at the time of slurry application

Regrowth interval (days)	Herbage mass* (kg DM/ha)	Extended tiller height (cm)
0	1626[a†]	5.1[a]
10	2058[b]	15.5[b]
20	2860[c†]	23.9[c]
SED	105.9	0.56
P	<0.001	<0.001

* Above ground level
† Values within each column with the same letter are not significantly different ($P>0.05$).

Table 2: Effect of slurry treatment, regrowth interval, and runoff day (RD) on runoff volume and runoff generation time

Runoff day (days post application)	Runoff volume[†] (litres)	Runoff generation time (seconds)	Runoff: Rainfall ratio[¶]
Slurry			
No slurry	0.70	1137	0.054
Slurry	0.58	1216	0.043
SED	0.051	115.5	
P	<0.001	0.011	
Regrowth interval			
0	0.75	1080	0.059
10	0.70	1138	0.054
20	0.47	1305	0.034
SED	0.099	141.0	
P	NS	NS	
Runoff day			
2	0.27[a§]	1193[b]	0.020
9	0.68[b]	1445[b]	0.046
16	0.98[c]	892[a]	0.084
SED	0.085	132.4	
P	<0.001	<0.001	
Slurry x Regrowth Interval			
P	NS	0.016	
Slurry x RD			
P	NS	NS	
Regrowth Interval x RD			
P	0.032	0.016	
Slurry x Regrowth x RD[‡]			
P	NS	NS	

† Runoff collected in the first 20 minutes following initiation of runoff
‡ Slurry treatment x Regrowth interval x Runoff day interaction
§ Values within factors in each vertical column with the same letter are not significantly different (P>0.05)
¶ Runoff:rainfall ratio calculated as the ratio of runoff collected during the 20 minute collection period to the total amount of simulated rainfall applied from the start of rainfall until the end of this 20-minute period. REML analysis could not be completed on non-parametric runoff:rainfall ratio
NS = not significant (P>0.05)

Table 3: Effect of slurry treatment and regrowth interval on the average flow-weighted mean concentrations of dissolved reactive phosphorus (DRP), particulate phosphorus (PP) and total phosphorus (TP) observed in runoff throughout the duration of the experiment

	DRP (mg/l)	PP (mg/l)	TP (mg/l)
Slurry treatment			
No slurry	0.25[a†]	0.16[a]	0.48
Slurry	0.51[b]	0.39[b]	1.05
SED	0.051	0.079	0.152
P	<0.001	0.012	<0.001
Regrowth interval			
0	0.52[b]	0.32	1.00
10	0.32[a]	0.23	0.63
20	0.30[a]	0.27	0.67
SED	0.062	0.096	0.186
P	<0.001	NS	NS
Runoff day			
2	0.52[b]	0.29	0.96[b]
9	0.35[a]	0.28	0.75[ab]
16	0.27[a]	0.25	0.59[a]
SED	0.043	0.068	0.110
P	<0.001	NS	0.021
Slurry x Regrowth[‡]			
No slurry	0.35[ab]	0.16[a]	0.59
0	0.21[a]	0.19[a]	0.46
10	0.20[a]	0.11[a]	0.38
20			
Slurry			
0	0.69[c]	0.48[b]	1.40
10	0.44[b]	0.27[ab]	0.79
20	0.40[b]	0.32[ab]	0.96
SED	0.087	0.136	0.262
P	0.011	0.015	NS
Slurry x Runoff day			
P	NS	NS	NS
Regrowth x Runoff day			
P	0.034	NS	0.009
Slurry x Regrowth x RD[§]			
P	NS	NS	NS

† Values within factors in each column with the same letter are not significantly different (*P*>0.05)
‡ Slurry treatment x Regrowth interval interaction
§ Slurry treatment x Regrowth interval x Runoff day interaction
NS = not significant (*P*>0.05)

Table 4: Pairwise comparisons for flow-weighted mean concentrations of dissolved reactive phosphorus (DRP), particulate phosphorus (PP), and total phosphorus (TP), measured in runoff from the Slurry treatment plots at RD2

	Regrowth interval comparison	Difference between treatment means	L.S.D.[†]	t	*P*
DRP	0 day – 10 days	0.717	0.608	2.620	0.034
	0 day – 20 days	0.462	0.434	2.367	0.049
PP	0 day – 10 days	0.663	0.863	1.738	NS
	0 day – 20 days	0.476	0.552	1.949	0.092
TP	0 day – 10 days	1.609	0.504	3.192	0.015
	0 day – 20 days	1.254	0.329	3.814	0.007

[†] Least significant difference
NS = Not significant (P>0.05)

Table 5: Effect of slurry treatment, regrowth interval, and runoff day on dissolved reactive phosphorus (DRP), particulate phosphorus (PP), and total phosphorus (TP) exports in runoff throughout the experiment

	DRP (g/ha)	PP (g/ha)	TP (g/ha)
Slurry treatment			
No slurry	2.86	2.16	5.9
Slurry	6.37	4.11	12.1
SED	1.284	0.871	2.49
P	NS	NS	NS
Regrowth interval			
0	6.57^{b†}	3.87	12.0^{b}
10	4.78^{ab}	3.42	9.7^{ab}
20	2.51^{a}	2.12	5.4^{a}
SED	1.573	1.067	3.06
P	0.001	NS	0.003
Runoff day			
2	2.44^{a}	1.17^{a}	4.3^{a}
9	5.51^{b}	3.78^{b}	11.1^{b}
16	5.90^{b}	4.47^{b}	11.6^{b}
SED	0.924	0.709	1.75
P	0.008	<0.001	<0.001
Slurry x Regrowth‡			
No slurry	3.92	2.18^{a}	7.1^{ab}
0	2.94	3.24^{ab}	7.1^{ab}
10	1.72	1.07^{a}	3.5^{a}
20			
Slurry	9.21	5.56^{b}	17.0^{b}
0	6.62	3.60^{ab}	12.2^{b}
10	3.29	3.16^{ab}	7.2^{ab}
20	2.224	1.509	4.32
SED	NS	0.043	0.012
P			

† Values within factors in each vertical column with the same letter are not significantly different (*P*>0.05)

‡ Slurry treatment x Regrowth interval interaction

NS = not significant (*P*>0.05)

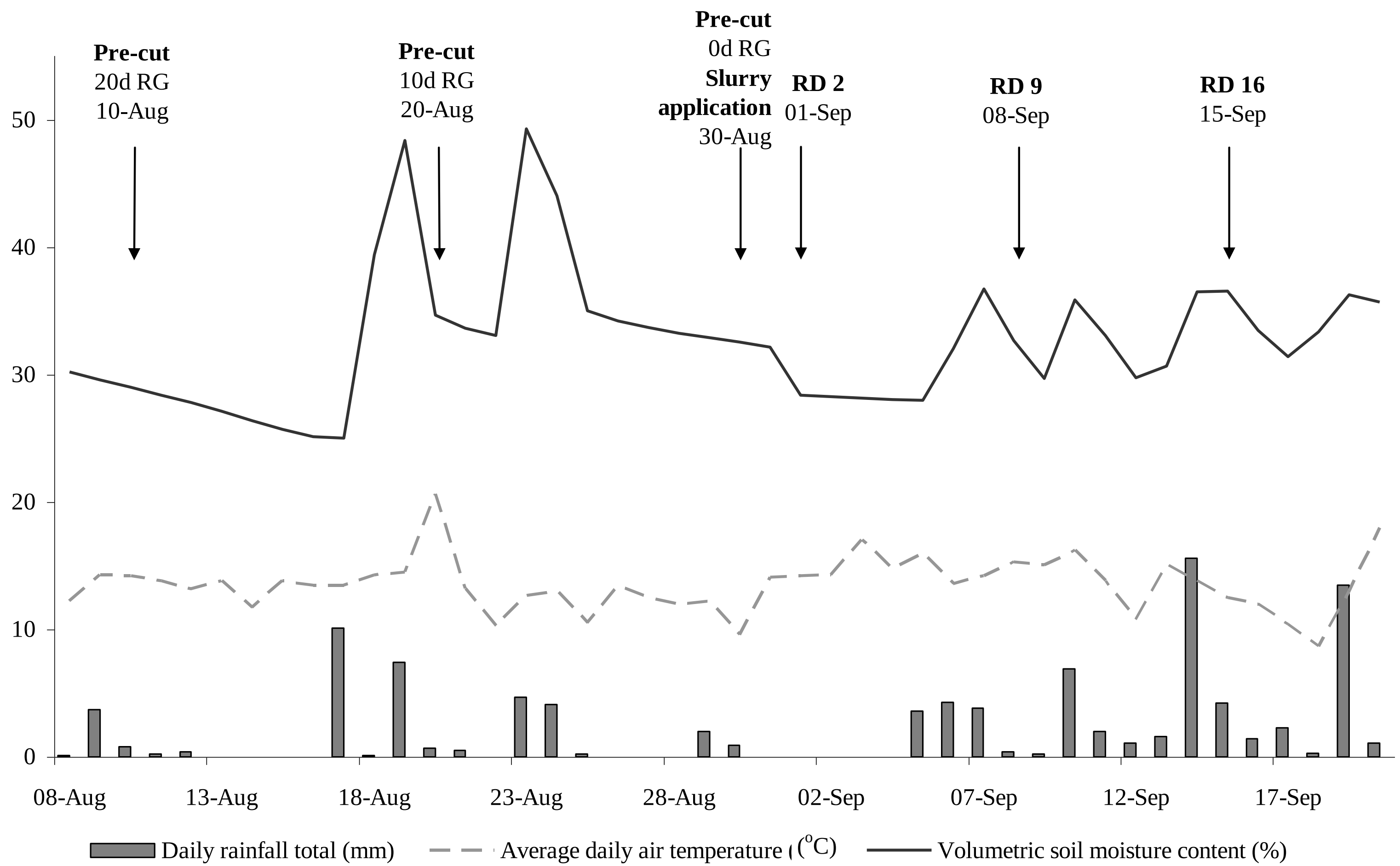

Figure 1: Daily average air temperature, total rainfall and volumetric soil moisture content recorded throughout the duration of the experiment, including the pre-cutting period and subsequent herbage harvest, with dates of cutting (RG), slurry application and runoff days (RD) superimposed

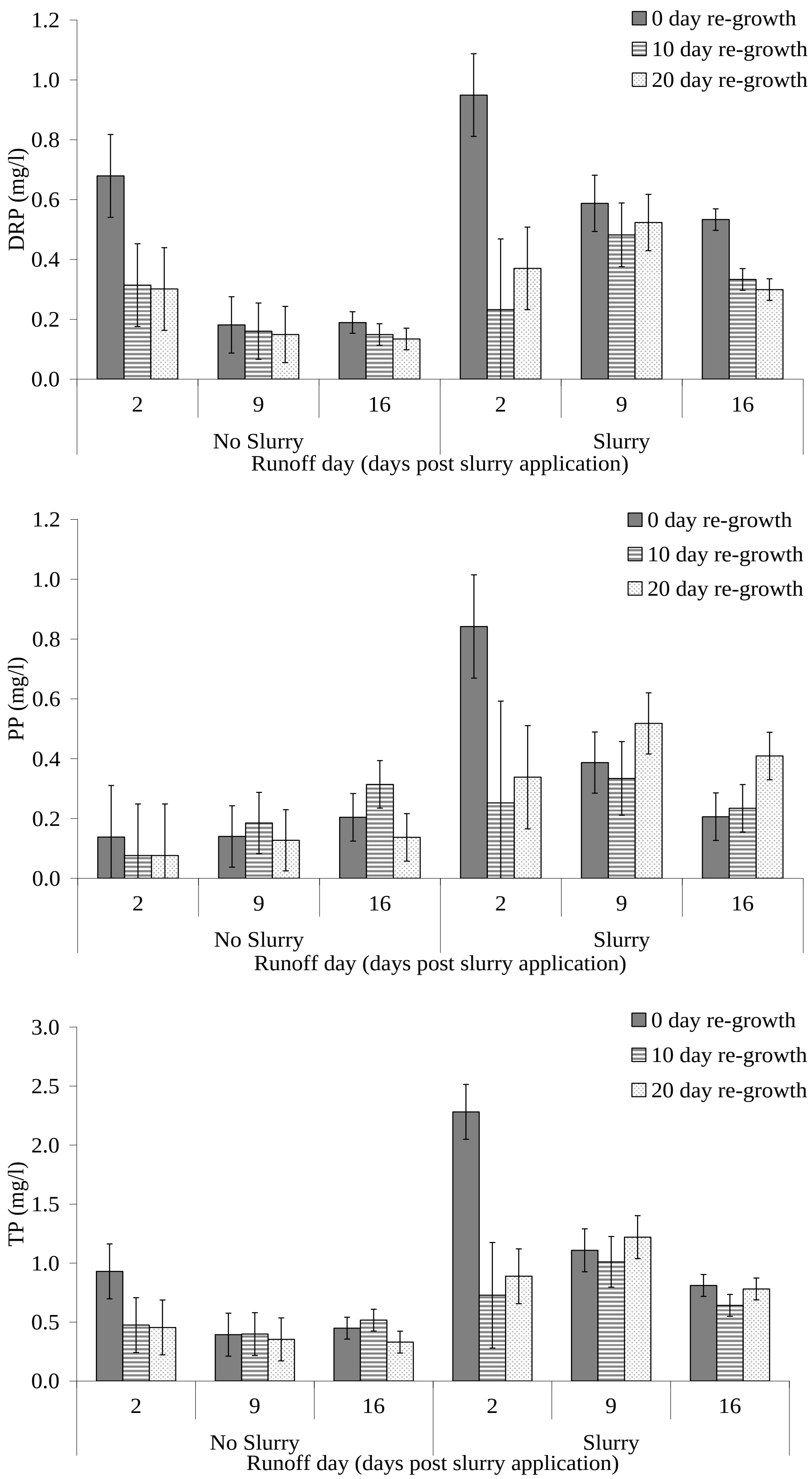

Figure 2: Flow-weighted mean concentrations of dissolved reactive phosphorus (DRP), particulate phosphorus (PP), and total phosphorus (TP), recorded at each runoff day (RD) throughout the duration of the experiment

Small Scale Experiment 3

Impact of slurry application method on phosphorus loss in runoff from grassland soils during winter and early spring

ABSTRACT

While there is a risk of phosphorus (P) being lost in overland flows when livestock manures are applied to grassland, this risk can be reduced when slurry is applied using the trailing-shoe, rather than the traditional splash-plate application technique. However, the effectiveness of the trailing-shoe technique as a means of reducing P losses has not been evaluated when slurry is applied during the winter and early spring periods, a time when soil moisture levels tend to be higher and herbage covers lower. To address this issue three treatments were examined in a 3 x 4 factorial design split-plot experiment, with treatments comprising three slurry treatments: Control (no slurry), Splash-plate and Trailing-shoe, and four slurry application dates: 7 December, 18 January, 1 March and 10 April. Dairy cow slurry was applied at a rate of 25 m^3/ha, while runoff was generated two, nine and sixteen days later and analysed for P content. Dissolved reactive P concentrations in runoff at day two was 41% lower when slurry was applied using the trailing-shoe technique, compared to the splash-plate technique (P<0.05). In addition, P concentrations in runoff were higher (P<0.05) from slurry applied in December and March compared to slurry applied in January or April, coinciding with periods of higher soil moisture contents. While the latter highlights that ‘calendar’ based non-spreading periods might not always achieve the desired consequences, applying slurry using a trailing-shoe system was an effective strategy for reducing incidental P losses.

INTRODUCTION

Excessive nutrient loss from livestock systems is an issue of increasing concern within many countries. Within the European Union (EU), legislation, primarily in the form of the Nitrates Directive, has been implemented to reduce losses of both nitrates and phosphorus (P) to water courses. Within regions where grassland-based systems dominate, such as western areas of the United Kingdom (UK), P losses in overland flows from animal manures applied to grassland pose a very significant risk. For example, within Northern Ireland (NI) grassland occupies approximately 80% of the total land area, and has been estimated to account for almost 60% of terrestrial P inputs to inland waterways (Smith *et al*., 2005: Jordan *et al*., 2007).

While reducing phosphorus levels in livestock diets can improve P use efficiency, and reduce P excretion in faeces (Ferris *et al*., 2010), livestock manures will inevitably contain P. Thus management practices need to be adopted to minimise the risk of P being lost once manure is applied. System of applying slurry can influence P losses. For example, in a recent study (McConnell *et al*., 2013) dissolved reactive P (DRP) concentrations in runoff were reduced by 37%, following slurry application using the trailing shoe technique, in comparison to traditional 'splash-plate' spreading. While this study was undertaken during the 'summer' months (May/June), the effectiveness of this technique as a means of reducing P losses does not appear to have been tested during the winter period. This is important in view of the fact that soil moisture levels tend to be higher during the winter months, while herbage covers tend to be lower. With regards the latter, McConnell *et al*. (unpublished) demonstrated increased P losses with decreasing herbage cover. Thus the primary objective of this study was to examine the effectiveness of the trailing shoe technique as a means of reducing P losses in surface runoff from slurry applied during the winter/early spring period.

A secondary objective of this experiment was to examine the impact of delaying applying slurry from the 'winter' to the 'early spring' period on P losses in surface runoff. While slurry spreading during the winter is still permissible in many regions of the EU, in regions where water quality is poor or likely to become poor as a consequence of either nitrates or phosphorus losses from agriculture, the Nitrates Directive stipulates that 'closed periods' should be adopted during which slurry spreading is not permitted. For example, within NI the Nitrates Action Programme legislates that slurry should not be spread between 16 October and 31 January (the 'closed period'). It is of course true that this closed period for manure application coincides with the months where crop requirement for nutrients are minimal and negative soil moisture deficits dominate across much of the agricultural land (Betts, 1997). Nevertheless, Holden *et al*. (2004) concluded that based on the frequency of rainfall, slurry spreading in winter would be possible for 3-4 years in 10 years in the northern part of Ireland. Indeed it is possible that the risk of P losses during the ‘closed period’ might not be any greater than the risk which exist outside of this period, provided slurry is applied at times when soil and weather conditions are ‘suitable’ for spreading.

MATERIALS AND METHODS

Site description

This experiment was conducted at the Agri-Food and Biosciences Institute, Hillsborough, NI (54^{o}27’N; 06^{o}04’W) between 7 December 2009 and 28 April 2010. The 192 m^2 experiment site was established on a permanent pasture located on a drumlin hill slope (average slope of 4.5%) with a north-easterly aspect. The area soil type was classified as a Soil Water Gley Class 1 overlying Silurian Shale (DANI Soil Survey of Northern Ireland) (FAO classification: Dystric Gleysol). The area has a Hydrology of Soil Types (HOST) classification of 24 which is indicative of poorly drained soils with high runoff rates, with this classification accounting for 46% of the land area in NI (Higgins, 1997). The soil had an average Olsen P content of 57.7 mg/l, twice the agronomic optimum, and an average bulk density of 0.83 g/cm^{-3} in the 0-5 cm horizon. Average annual rainfall and duration of the growing season at the site was 890 mm and 254 days for the periods 1971-2000 and 1951-1990, respectively (Betts, 1997). Daily meteorological data for the site were supplied by a UK Meteorological Office weather station located 300 m from the field site. Prior to the start of this experiment, grass was harvested from the area on 22 September and again on 29 October 2009 using a hand operated mower (3600BM, Agria, Möckmühl, Germany) and removed by hand.

Treatments

Treatments examined in this 3 x 4 factorial design experiment comprised three slurry treatments (Control, Splash-plate and Trailing shoe) and four slurry application dates. The forty-eight experimental plots (each 0.5 m^2) were arranged in four blocks in a split plot design, with each block ‘split’ into four sub-blocks (4.5 m wide x 1.0 m deep), and with each sub-block comprising three 0.5 m^2 plots. Plots within each sub-block were situated 1.0 m apart, while each block was separated by a 3.0 m buffer strip. The three plots within each sub-block represented the three slurry treatments, with one of the four sub-blocks within each block treated with slurry on each of the four application dates.

The target interval between slurry application dates was 42 days, commencing 7 December 2009, thus allowing two applications within the NI closed period for slurry application, and two applications outside of this closed period. However, slurry was only applied on the planned dates if conditions on that day complied with legislation contained within the Nitrates Action Programme (Northern Ireland) Regulations (EHS, 2007). The spreading criteria outlined in this legislation prohibit the land application of manure under the following circumstances: ‘on waterlogged soils, flooded land, or land liable to flood; on frozen or snow covered ground; and if heavy rain is forecast’. Using these criteria the following conditions were set to ensure spreading only took place when there was minimal risk of pollution following application:

1. No snow cover
2. Soil temperature above 0ºC
3. Forecast rainfall on day of application below 2.5 mm
4. Total forecast rainfall for the following two days below 10 mm
5. Soil moisture levels below or within +2% of field capacity (40.1%)

By adhering to these criteria, the following spreading dates were adopted: 7 December 2009, 18 January 2010, 1 March 2010 and 12 April 2010, with all intervals between spreading dates being 42 days.

Approximately two weeks prior to each slurry application (so as to minimise soil disturbance), a shallow trench was excavated along the down-slope edge of each plot and a stainless steel V-shaped collection tray (0.5 x 0.1 x 0.1 m) placed in the trench to act as a runoff collector. The upslope edge of each tray was fitted with a 0.07 m horizontal lip, and this was driven horizontally (to a distance of approximately 0.05 m) into the soil directly underneath each plot, at a depth of approximately 0.03 m below the soil surface. A 0.02 m diameter outlet at the base of the each collection tray allowed runoff collected from each plot to drain into a two-litre high density polyethylene (HDPE) collection container via a 0.5 m length of underground pipe.

The dairy cattle slurry used within this study was collected following the mechanical mixing of an above ground slurry store. Slurry was collected three weeks prior to the first slurry application date and stored at <4°C in HDPE containers throughout the duration of the experiment. No slurry was applied to the Control treatment plots on any application date. With the other two treatments slurry was applied by hand at a rate equivalent to 20 m^3/ha (one litre per 0.5 m^2 plot). Within the plots slurry was not applied within 0.05 m of the plot edges so as to minimise the impact of any disturbance created by the introduction of the plot surrounds following slurry application. Splash-plate spreading was simulated using a plastic jug to pour slurry onto a wooden board which caused it to splash across the plot area. The Trailing shoe treatment consisted of three slurry tramlines (each 0.9 m long) running in the same direction as the slope, an earlier study having found direction of spreading to have no effect on P losses in runoff (McConnell *et al*., 2013). Tramlines were spaced 0.225 m apart, with the two outer tramlines situated 0.025 m away from the sides of the plot and terminating 0.05 m from the top and bottom of each plot. To simulate the Trailing shoe spreading technique, grass was parted by hand and held in place with wooden boards. Slurry was then applied to the base of the sward using a thin spouted plastic jug. Plots were covered with translucent plastic sheeting (positioned approximately 0.2 m above the ground) between slurry application and the first rainfall simulation event, a period of approximately 48 hr.

On the same day that slurry was applied, the sub-plots were hydrologically isolated from overland and shallow sub-surface flow by inserting stainless steel surrounds into the soil along the sides and across the up-slope end of each plot to a depth of 0.05 m.

Rainfall simulation

Rainfall simulations were performed at 2 (RD2) and 9 (RD9) days following the December slurry application, and at 2, 9 and 16 (RD16) days following the January, March and April slurry applications. Two Amsterdam drip-type rainfall simulators (Bowyer-Bower and Burt, 1989) were employed to supply rainfall at a constant rate of 20 mm/hr. This rainfall intensity has a return period equivalent to a one-in-five-year event for NI (Betts, 1997). Thirty minutes of runoff was collected at each rainfall simulation in 2 x 15 minute fractions. Runoff volume and time taken to generate runoff was recorded. Water used in the rainfall simulations was passed through a DC9 general deionising cylinder (Purite limited) to reduce its P concentration. The cylinder delivered deionised water with an average dissolved reactive P (DRP) and nitrate-N concentration of 9.3 μg/l and 253 μg/l, respectively.

Water quality analysis

Water samples were placed in a fridge at 3°C within 4 hr of sampling. One sub-sample was analysed for DRP, total dissolved P (TDP) and total P (TP) within 24 hr of sampling. Samples for DRP and TDP were filtered through 0.45 μm filters (MF-Millipore, Billerica, MA) before analysis. Dissolved reactive P was determined colorimetrically using the ascorbic acid reduction technique described by Murphy and Riley (1962). Acid-digestion techniques (Eisenreich *et al*., 1975) were used to convert TDP and TP content to DRP. These samples were then analysed using the ascorbic acid reduction method outlined above (Murphy and Riley, 1962). Particulate P (PP) was calculated as the difference between TP and TDP.

Sward measurements

On each slurry application date extended tiller measurements were taken using a 30 cm ruler at 10 randomly selected positions within each 0.5 m^2 plot.

Soil measurements

Soil moisture readings were taken using a volumetric soil moisture probe (HH2, Delta-T Devices Ltd., Cambridge, UK). Three soil moisture readings were taken per plot at each slurry application. Average daily soil moisture values were provided by a continuous data logger attached to a HH2 volumetric soil moisture probe located 300 m from the study site (WS-STD1, Delta-T Devices Ltd., Cambridge, UK) and provided soil moisture reading for each rainfall simulation event.

Slurry analysis

Slurry was sampled at each application date and analysed for dry matter (DM), nitrogen (total Kjeldahl N), ammonium-N and P content. Dry matter content was determined gravimetrically after oven-drying at 100°C for 48 hr. Slurry nitrogen and ammonium-N content were determined by analysing fresh manure, as described in

Jensen (1991). Total P was determined on a dried sample of slurry by the methods described in APHA (1995).

Statistical analysis

Data were analysed using Genstat Version 12 software (VSN International, 2009, UK). Runoff days 2, 9, and 16 were treated as independent events following the completion of regression analysis using Microsoft Excel (Microsoft Corporation, 2003, WA) which determined no significant correlation between the amount of rainfall a plot received prior to a rainfall simulation event, and the subsequent runoff volume generated or the time required to generate runoff, at that event. Subsequently, data from each runoff day were analysed separately using ReML repeated measures analysis which included both application date and slurry method. This was adopted rather than a two-way Analysis of Variance to take account of the random effect of the main plot in the split plot design. A power city-block distance model was applied to the data. The presence of non-normality in some variables in the data set was addressed by fitting logarithm base 10 transformations to these variables before analysis. Herbage results were also analysed using ReML repeated measures analysis with application date and spreading method as factors.

RESULTS

Slurry applied during December, January, March and April had a DM content of 68.3, 67.7, 65.9 and 67.6 g/kg, an ammonium-N content of 1.91, 1.92, 1.99 and 2.18 g/kg, a total nitrogen content of 2.98, 2.99, 3.14 and 3.17 g/kg, and a total P content of 5.69, 5.56, 5.74 and 5.52 g/kg DM, respectively.

Daily rainfall on each slurry application date was less than one millimetre (Table 1). Rainfall during the 48-hr periods following the December and January applications was 1.0 and 6.3 mm, respectively, with no rainfall during the 48-hr period following the March and April applications. The average air temperature was 5.0, 4.1, 2.7 and 5.3^{o}C on the day of the December, January, March, and April applications, respectively, while soil temperature at 10 cm depth ranged from 1.5^{o}C (March application) to 7.6^{o}C (April application). Volumetric soil moisture (VSM) content on the day of slurry application in December, January, March, and April, was 41.3, 38.1, 41.6 and 35.3%, respectively. The average extended tiller grass height ranged from 8.5 cm at the March application to 11.7 cm at the April application.

Following each slurry application, weather conditions were monitored during the 16-day period during which runoff measurements were undertaken (Figure 1). During this period, average VSM contents following the December, January, March, and April slurry applications were 38.8, 39.2, 35.2 and 33.9%, respectively. The 16-day period following the January application was the wettest period during the experiment, with 44.0 mm of rainfall falling during this period. In contrast, 8.6, 0.9 and 9.2 mm of rainfall fell during the 16-day periods following the December, March, and April applications, respectively. The average air temperature during the 16-day periods following the December, January, March, and April slurry applications was 3.2, 3.0, 4.5 and 9.0^{o}C, respectively. Following slurry application on 7 December, air temperature fell below freezing on 18 December, the start of a 23-day cold period during which ground remained almost permanently frozen. As a result, the RD16 rainfall simulations following the December slurry application were not undertaken.

Slurry treatment had no significant effect on either runoff generation time or the volume of runoff produced over a 30-minute period at RD2 and RD9 (Table 2). While runoff generation time at RD16 was unaffected by slurry treatment, runoff volume was higher with the Trailing shoe treatment than with the Control treatment (P<0.05) at this time. In contrast, runoff generation time differed with application date at RD2 and RD9 (P<0.001), being longest with the April application date. The volume of runoff produced was unaffected by application date at RD9 and RD16, while being highest with the January and March applications at RD2 (P<0.05).

At each of RD2, RD9 and RD16, flow-weighted mean concentrations of DRP, PP and TP in runoff were significantly lower (P<0.05) with the Control treatment than with either the Splashplate or Trailing shoe treatments (Table 3). At RD2, concentrations of DRP, PP and TP in runoff were significantly lower (P<0.05) with the Trailing shoe treatment than with the Splashplate treatment, while these differences had largely disappeared at RD9 and RD16. The exception to this was DRP concentrations in runoff at RD16 which was significantly higher (P<0.05) with the Trailing shoe treatment than with the Splashplate treatment. Application date had a significant effect ($P<0.05$) on DRP, PP and TP concentrations in runoff at each of RD2, RD9 and RD16 (with the exception of PP concentrations in runoff at RD2). In general, PP and TP concentrations decreased from December through to April, while DRP concentrations were highest in December and March. There was a significant Slurry treatment x Application date interaction at each of RD2, RD9 and RD16, for DRP ($P=0.028$, $P<0.001$ and $P=0.003$, respectively) and TP ($P=0.018$, $P=0.029$ and $P<0.001$, respectively) concentrations in runoff.

At each application date total P export at RD2 was greater ($P<0.05$) from the Splashplate treatment than from the Trailing shoe treatment, with TP exports from both treatments higher ($P<0.05$) than from the Control treatment (Figure 2). Total P export rates at RD2 were significantly greater ($P<0.05$) following the January and March slurry applications than following the December and April slurry applications. Total phosphorus exports at RD9 were higher with the Splashplate and Trailing shoe treatments ($P<0.05$) than with the Control treatment, whilst TP export at RD9 following the December application was higher ($P<0.05$) than either the January, March, or April applications. By RD16, TP export was significantly greater ($P<0.05$) from the Trailing shoe treatment than from the Splashplate treatment. Application date also had a significant effect ($P=0.013$) on TP export at RD16, with the January and March applications having a higher TP export than the April application. Following each slurry application, the majority of the TP exported from the Control treatment was in particulate form, with the proportion of PP exported as TP gradually reduced from 74% in December to 37% in April. In contrast both the Splashplate and Trailing shoe treatments exhibited lower proportions of TP export in particulate form, with PP from the slurry treatments accounting for 46, 51, 58 and 29% of TP following the December, January, March and April applications, respectively.

DISCUSSION

Runoff generation

Slurry was only applied when soil and weather conditions were in agreement with guidance contained within the Code of Good Agricultural Practice (DARD, 2008) and the Nitrates Action Programme (Northern Ireland) Regulations (EHS, 2007). Nevertheless, there was considerable variation in runoff characteristics between the four application dates, and between runoff days 2, 9 and 16. In general, when VSM content was high, runoff volume was also high. Vegetation cover may also have influenced runoff volume, with the low runoff volumes in April coinciding with the maximum grass tiller heights recorded. As plant cover increases, interception by vegetation has a greater effect on lowering the kinetic energy of raindrops, and this is thought to facilitate greater infiltration thus reducing runoff generation (Barfield *et al*., 1979). Small scale changes in soil structure and soil micro-topography between individual rainfall simulation plots may also have caused some variation in runoff generation within each runoff event.

There is conflicting evidence concerning the impact of slurry application on runoff generation. While Smith *et al*. (2001) suggested that slurry can reduce infiltration rates and promote runoff by 'sealing' the soil surface, slurry application had no significant impact on runoff generation in the current experiment, in agreement with findings of similar grassland-based studies (Srinivasan *et al*. 2007; Johnson *et al*., 2011).

Effect of slurry treatment on P losses in runoff

As expected, P concentrations in runoff from the slurry treated plots were higher than from the Control plots at all runoff day events, with this highlighting the risk of P losses in surface runoff following intensive rainfall events, even up to 16 days post slurry application. However, this experiment provides clear evidence that the use of the trailing shoe slurry spreading technique during the winter-early spring period can reduce P concentrations in runoff, relative to traditional splashplate spreading. For example, DRP, PP and TP concentrations were reduced by 41, 25 and 32% respectively at a rainfall event two days after slurry application. This is likely due to a smaller slurry-rainfall contact area with slurry applied using the trailing shoe technique compared to the Splashplate technique. As a consequence of this smaller contact area, the potential for the dissolution of slurry P to runoff is reduced with the trailing shoe (Leinweber *et al*., 2002), while the cumulative raindrop impact experienced by the slurry will also be reduced. In addition, when slurry is placed at the base of the sward using the trailing shoe technique, the grass plants intercept and deflect raindrops, thus reducing their impact on the slurry. Structural breakdown of slurry aggregates by raindrop impact can result in both particle detachment and the exposure of previously occluded P (Kleinman *et al*., 2002; Leinweber *et al*., 2002). As a result of the lower cumulative raindrop impact associated with the Trailing shoe treatment, the potential for both dissolved and particulate P losses is reduced.

These findings are consistent with those reported by McConnell *et al*. (2013) in a study involving slurry application to a grass stubble during the summer period. In this study DRP concentration in runoff following a rainfall event two days after slurry application was 37% lower when slurry was applied using the trailing shoe technique compared to the splashplate technique. Thus the current study clearly demonstrates that during the winter and early spring periods the trailing shoe technique can be equally effective as during the summer months in reducing the risk of P losses in runoff.

Effect of application date on runoff P

High DRP concentrations in runoff (RD2) following the December and March slurry applications coincided with high VSM levels (above field capacity) at the time of slurry application. Sommer and Jacobsen (1999) noted that at higher soil water contents, the mass flow of liquid from surface broadcast pig slurry into the soil was reduced. Thus the high VSM levels at the time of slurry application in the current study most likely hindered the infiltration of the liquid fraction of slurry into the soil during the first 48 hr after application, leaving a higher proportion of the applied slurry P on the soil surface. Indeed, McGechan and Lewis (2000) noted that applying slurry to fields with high soil hydraulic conductivities provided one of the greatest opportunities to minimise the loss of P in runoff during the winter period, as this would both aid the infiltration of slurry P and reduce the potential for runoff generation. Similarly, Vadas *et al*. (2011) noted that under drier soil conditions, the time taken to generate runoff at an individual rainfall event increases, resulting in a greater interaction time between slurry and rainfall, thus allowing the rainfall to transfer P to the upper soil horizons where it undergoes rapid sorption by soil particles. The impact of this process was particularly evident following the March application of slurry, where the highest runoff:rainfall ratio (0:40) recorded during the experiment was observed at RD2. This was reflected in a short time interval to the start of runoff, minimal translocation of P into the soil, and the highest DRP concentrations and exports recorded throughout the experiment. In contrast, a low runoff:rainfall ratio (0:13) and a low VSM content at the time of slurry application (35.3%), as observed following the April slurry application, will have facilitated rapid translocation of P into the underlying soil, and resulted in lower concentrations and exports of P.

At each application date throughout this experiment, the Trailing shoe technique significantly reduced the magnitude of DRP concentrations in runoff relative to the splashplate technique, regardless of weather conditions. The absence of significant variation between runoff DRP concentrations from the Trailing shoe treatment across the four application dates, suggests that the lower rainfall-slurry contact offered by the Trailing Shoe helps mitigate against adverse weather conditions thus minimising DRP concentrations in runoff. In contrast, the high levels of DRP in runoff from the Splashplate treatment under high VSM levels (December and March), suggests that splashplate spreading methods can exacerbate the magnitude of runoff P loss under wetter weather conditions.

Although a number of authors (McGechan and Lewis, 2000; Jordan *et al*., 2007) have highlighted that the 'safe' application of slurry during the winter period can often be limited due to the absence of suitable weather conditions, a study by Holden *et al*. (2004) predicted that in five years out of a ten-year period, suitable weather conditions for slurry spreading would be available for at least 10% of the winter period in Northern Ireland. The findings of the current study clearly demonstrate that provided spreading conditions are suitable, P concentrations in runoff from slurry applied during the 'winter' period (18 January) can be lower than from slurry spread in early Spring (1 March), thus highlighting that calendar-based 'closed periods' might on occasions actually delay slurry applications until a higher risk period. Nevertheless, crop requirements for nutrients during the winter are low. Thus, while it may be possible to apply slurry during the winter without exacerbating the risk of P loss in runoff compared to slurry applied later in the year, nitrogen use efficiency will be greater if slurry applications are delayed until grass growth has commenced. This was reflected in the trend in yield of herbage harvested from plots in the current experiment on 25 May (1.82, 2.16, 2.46 and 2.50 t DM/ha for the December, January, March and April application dates respectively: SED, 0.1732, *P*=NS).

Runoff P losses over successive rainfall events

The reductions in runoff P concentrations over the three successive runoff events (RD2, RD9 and RD16) in the current experiment are in line with findings from earlier studies (Kleinman and Sharpley, 2003; O'Rourke *et al*., 2010; Vadas *et al*., 2011). While a number of mechanisms may account for the decline in P concentrations in runoff over time, P availability at the soil surface is one of the primary factors controlling P loss (Kleinman and Sharpley, 2003). Phosphorus availability is determined by the extent of P removal during earlier runoff events, and by the translocation of P into the soil via infiltration and bioturbation, whereby rapid sorption of P occurs. Within the current experiment, P removal during previous runoff events most likely contributed in part to the lower P concentrations in runoff at RD9 and RD16. Indeed higher DRP concentrations in runoff at RD2 from the Splashplate treatment relative to the Trailing shoe treatment indicate a greater depletion of slurry P at the soil surface and most likely resulted in the lower DRP concentrations in runoff from the Splashplate treatment (0.96 mg/l) relative to the Trailing shoe treatment (1.18 mg/l) by RD16. Consequently, though the Splashplate appears to pose a greater short term threat to water quality, the Trailing shoe may perhaps pose a longer term threat, with the rate of decline of P loss over successive runoff events being lower than that of the Splashplate treatment. However, this is an issue that requires further clarification.

While earlier rainfall events contributed to lower P concentrations at later events in the current experiment, in reality only a small proportion of the total P applied in slurry (<10%) was lost in this way. Similarly, Kleinman and Sharpley (2003) noted that while runoff DRP concentrations fell by 90% over the course of three rainfall simulation events (at 3, 10 and 24 days after slurry application), the total DRP exported only accounted for 4 to 13% of the slurry water extractable P applied. Thus a large part of the decrease in concentrations of runoff P over time must be due to the assimilation of slurry P into soil. Up to 60% of P from dairy cow slurry can infiltrate into soil provided the manure has a DM content of less than 15% (Vadas *et al*., 2007), with up to two-thirds of infiltration occurring during the first four days after slurry application (Vadas, 2006). Once infiltration takes place the manure P is subject to rapid sorption by soil colloids, thus reducing its availability to runoff water. For example Sharpley (1997) noted a positive correlation between the initial soil sorption capacity of soil, and the difference in runoff DRP concentrations between the first and final rainfall simulation events after slurry application.

CONCLUSIONS

In agreement with previous research undertaken during the summer months, the use of the trailing shoe technique during the winter and early spring period was effective in reducing P losses in runoff from applied slurry, when compared to the splash plate. Phosphorus concentrations in runoff appeared to be primarily driven by soil moisture levels, and not by date of slurry application *per se*, thus highlighting the importance of considering soil conditions at the time of spreading when seeking to minimise nutrient losses from applied slurry. In addition to optimal timing of slurry application the use of trailing shoe during winter and early spring should be considered as a mitigation measure to minimise the risk of nutrient loss during this period. However, further research is required to test the scale dependency of the finding of this study.

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Table 1: Weather and soil conditions, and extended tiller heights at each slurry application

Application date	Rainfall total (mm)		Temperature (°C)			Volumetric soil moisture content (%)	Extended tiller heights (cm)
			Air		Soil @10 cm		
	Day	+48 hr	Max	Min			
7 December	0.6	1.0	8.0	3.4	5.2	41.3	9.6
18 January	0.3	6.3	8.5	3.4	4.8	38.1	9.4
1 March	0.0	0.0	6.4	-2.6	1.5	41.6	8.5
10 April	0.0	0.0	18.0	3.7	7.6	35.3	11.7

Table 2: Effect of slurry treatment and slurry application date on the time taken to generate runoff, the volume of runoff produced, and the average runoff:rainfall ratio throughout the study

		Runoff day 2			Runoff day 9			Runoff day 16		
		Time to runoff (seconds)	Runoff volume† (litres)	Runoff: rainfall ratio	Time to runoff (seconds)	Runoff volume† (litres)	Runoff: rainfall ratio	Time to runoff (seconds)	Runoff volume† (litres)	Runoff: rainfall ratio
Slurry treatment	Control	925	1.84	0.24	1014	1.19	0.15	991	1.48[a]	0.19
	Splashplate	973	1.83	0.24	975	1.50	0.20	872	1.65[ab]	0.22
	Trailing shoe	800	1.95	0.27	856	1.69	0.23	764	1.88[b]	0.26
	SED	99.1	0.173		128.1	0.206		98.2	0.136	
	P	NS	NS		NS	NS		NS	0.016	
Application date	7 December	718[a§]	1.45[a]	0.21	506[a]	1.78	0.28	-	-	-
	18 January	841[a]	2.18[b]	0.30	937[b]	1.07	0.14	643	2.17	0.32
	1 March	630[a]	2.69[b]	0.40	773[ab]	1.67	0.23	1055	1.67	0.21
	12 April	1408[b]	1.16[a]	0.13	1578[c]	1.31	0.14	928	1.16	0.15
	SED	123.3	0.344		155.3	0.364		175.0	0.408	
	P	<0.001	0.003		<0.001	NS		NS	NS	

† Runoff volume collected in the first 30 minutes following the initiation of runoff

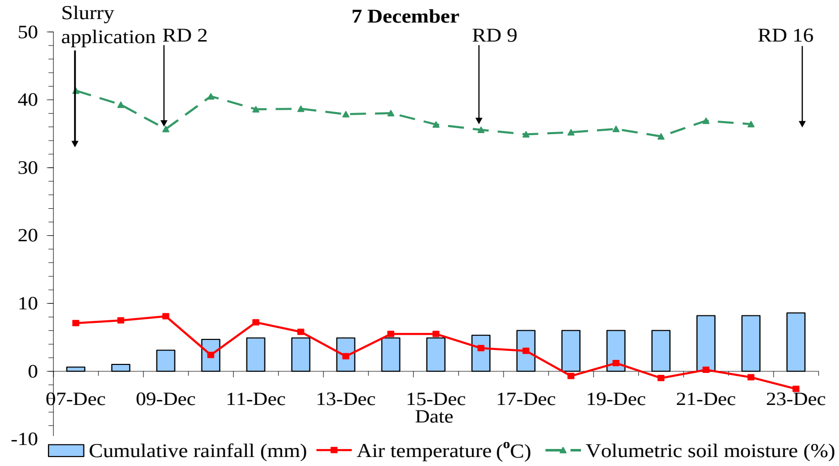
‡ Runoff:rainfall ratio calculated as the ratio of runoff collected in a 30 minute to the total amount of simulated rainfall applied from the start of rainfall until the end of this 30-minute period

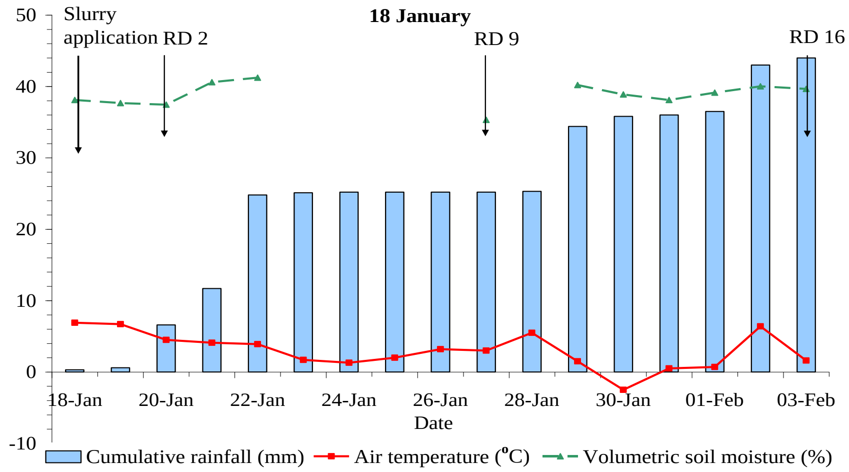
§ Means with the same superscript within columns, within factors, are not significantly different (*P*>0.05)

Table 3: The effect of slurry treatment and slurry application date on flow-weighted mean concentrations of dissolved reactive phosphorus (DRP), particulate phosphorus (PP) and total phosphorus (TP) in runoff throughout the experiment

		Runoff day 2			Runoff day 9			Runoff day 16		
		DRP (mg/l)	PP (mg/l)	TP (mg/l)	DRP (mg/l)	PP (mg/l)	TP (mg/l)	DRP (mg/l)	PP (mg/l)	TP (mg/l)
Slurry treatment	Control	0.16[a‡]	0.91[a]	1.24[a]	0.16[a]	0.39[a]	0.71[a]	0.19[a]	0.29[a]	0.57[a]
	Splashplate	3.34[c]	4.33[c]	9.10[c]	1.85[b]	1.34[b]	3.93[b]	0.96[b]	0.43[b]	1.63[b]
	Trailing shoe	1.96[b]	3.26[b]	6.21[b]	1.57[b]	1.14[b]	3.40[b]	1.18[c]	0.48[b]	1.90[b]
	SED	0.128	0.289	0.292	0.040[†]	0.089[†]	0.059[†]	0.038[†]	0.053[†]	0.033[†]
	P	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Application date	7 December	1.99[b]	3.48	6.74[c]	1.02[b]	2.00[c]	4.07[c]	-	-	-
	18 January	1.47[a]	2.99	5.14[b]	0.63[a]	1.14[b]	1.99[b]	0.54[a]	0.86[a]	1.52[c]
	1 March	2.14[b]	2.80	5.76[b]	0.78[b]	0.57[a]	1.82[ab]	0.75[b]	0.23[c]	1.18[b]
	12 April	1.68[a]	2.13	4.42[a]	0.72[ab]	0.39[a]	1.37[a]	0.54[a]	0.30[b]	0.98[a]
	SED	0.208	0.461	0.349	0.046[†]	0.102[†]	0.068[†]	0.037[†]	0.037[†]	0.033[†]
	P	0.014	NS	<0.001	0.002	<0.001	<0.001	<0.001	<0.001	<0.001
Interaction[§]	*P*	0.028	NS	0.018	<0.001	NS	0.029	0.003	NS	<0.001

[†] Logarithmic base 10 standard error of difference
[‡] Means with the same superscript within columns, within factors, are not significantly different (P>0.05)
[§] Slurry treatment x application date interaction
NS = P>0.05


Slurry application
7 December
RD 2
RD 9
RD 16
50
40
30
20
10
0
-10
07-Dec
09-Dec
11-Dec
13-Dec
15-Dec
17-Dec
19-Dec
21-Dec
23-Dec
Date
Cumulative rainfall (mm)
Air temperature (°C)
Volumetric soil moisture (%)


Slurry application
18 January
RD 2
RD 9
RD 16
50
40
30
20
10
0
-10
18-Jan
20-Jan
22-Jan
24-Jan
26-Jan
28-Jan
30-Jan
01-Feb
03-Feb
Date
Cumulative rainfall (mm)
Air temperature (°C)
Volumetric soil moisture (%)

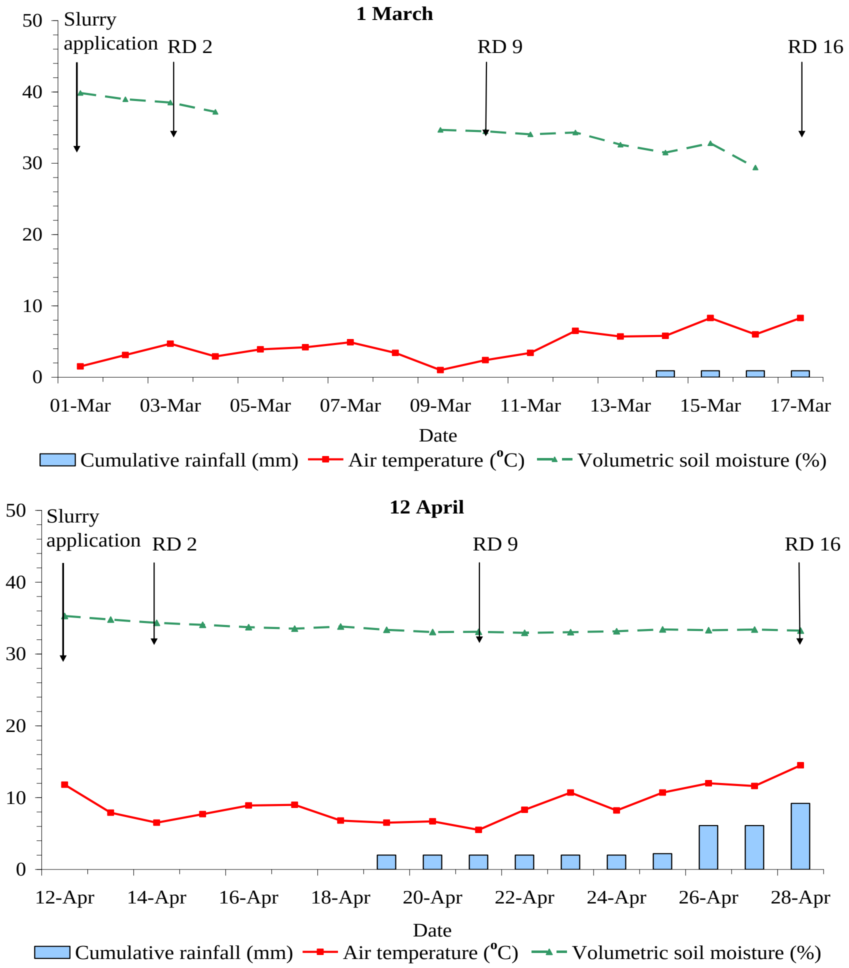


Figure 1: Cumulative rainfall totals, average air temperature and average soil moisture content at the study site during the 16-day period following each slurry application date

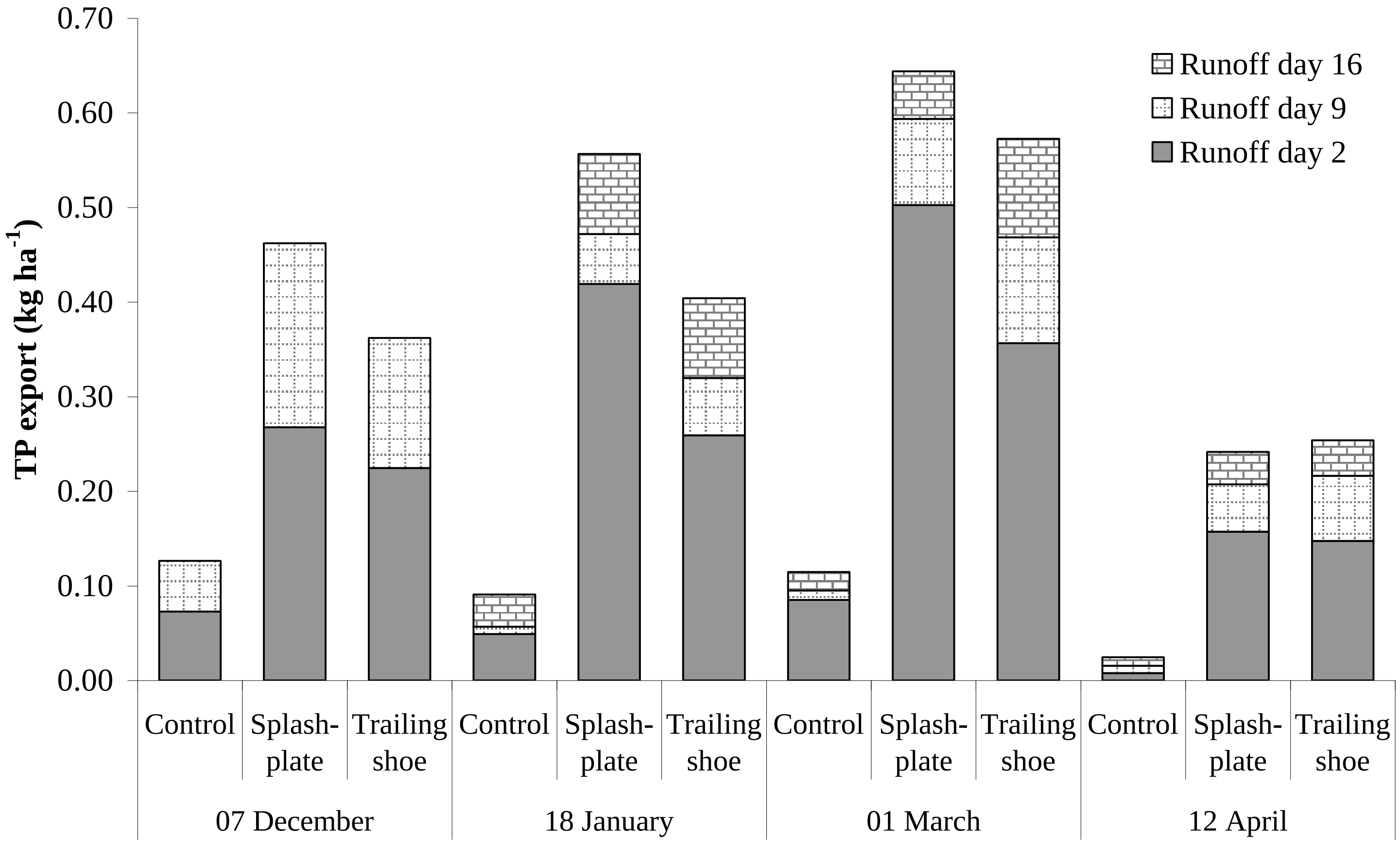


Figure 2: The effect of application date and slurry treatment on total phosphorus (TP) export rates in runoff, collected 2, 9, and 16 days post slurry application

Small Scale Experiment 4

Phosphorus loss in runoff following the application of anaerobically digested slurry to grassland.

ABSTRACT

The microbial processes which take place during anaerobic digestion alter both the physical and chemical properties of slurry, including changing the soluble phosphorus (P) content. Incidental P losses following slurry application to grasslands remain a major threat to water quality, and consequently changes in the soluble P content of slurry following anaerobic digestion have potential environmental consequences. An experiment was conducted to examine the effect of anaerobic digestion of dairy cow slurry on P losses in surface runoff following manure application to grassland. Three treatments were examined in this plot trial with each treatment replicated 4 times in a randomised block design. Treatments comprised a control (no slurry), anaerobically digested slurry (AD) and undigested slurry (UD), with manure applied to 0.5 m^2 plots at a rate equivalent to 1000 mg total P per plot. Simulated rainfall was applied to each plot at a rate of 30 mm/hr on day two and nine post slurry applications, with runoff samples collected and analysed for a range of P fractions. Despite AD slurry having a higher (P<0.001) water extractable P content than UD slurry, dissolved reactive P concentrations in runoff were unaffected (P>0.05). In contrast, both dissolved unreactive P and particulate P concentrations in runoff from the AD treatment were lower (P<0.05) than from the UD slurry treatment. Consequently the results of this experiment highlight that the anaerobic digestion of slurry does not increase the risk of P being lost in runoff following slurry application.

INTRODUCTION

The European Union's Renewable Energy Directive (EC/28/2009), aims to increase the proportion of energy generated from renewable resources to 20% by 2020. It advocates the use of anaerobic digestion, which through the bacterial breakdown of agricultural, sewage, and municipal organic wastes, produces methane which can be burned to provide a renewable energy resource (Smith *et al*., 2007). Within the United Kingdom (UK), the agricultural sector provides the largest potential source of organic matter for anaerobic digestion, with an average production of over 90 million tonnes of organic wastes per annum (DEFRA, 2009). Similarly in Northern Ireland (NI) animal manures account for 92% of annual organic matter production, with the anaerobic digestion of this matter having the potential to supply 7% of NI electricity requirements (Frost and Gilkinson, 2010).

Whilst the primary focus of anaerobic digestion is the supply of renewable energy, the process has also been found to significantly increase the ammonium-N content of animal slurry (Smith *et al*., 2007) resulting in an increase in herbage response to slurry application (Frost and Gilkinson, 2010). For example, Sommer and Birkmose (2007) reported a 30% increase in plant N utilisation following application of AD slurry to grassland relative to its undigested counterpart. Likewise, while a number of studies have observed negligible changes in slurry phosphorus (P) content following anaerobic digestion (Martin, 2004; Wright, 2004), there is some evidence to suggest that the process can increase the water extractable phosphorus (WEP) concentration of slurry (Wright, 2004; Moody *et al*., 2009). For example, in a study of five cow slurry digestion plants, Martin (2004) observed increases in slurry orthophosphate concentrations between 8.0 and 46.3%. However, other authors have noted a decrease in slurry orthophosphate concentration following anaerobic digestion due to the formation of struvite (Güngör and Karthikeyan, 2005; Möller *et al*., 2008.

The application of slurry to agricultural land increases both the basal soil P levels and the risk of pollution through incidental P transfers to nearby waterways. The magnitude of incidental P lost is dictated by both weather and soil conditions at the time of slurry application (McGechan *et al*., 2008), and by variations in manure physical and chemical characteristics (Vadas *et al*., 2007). In recent years, large quantities of P in livestock diets have resulted in the application of P rich animal slurry to agricultural soil; for example in 2010 the NI dairy industry generated 1650 tonnes of P in animal slurry (DARD, 2011). In this context, the potential impact of AD on the WEP content of slurry has consequences for incidental losses of P, with a number of authors observing a significant correlation between the WEP content of applied slurry, and dissolved reactive P (DRP) concentrations in runoff (Kleinman *et al*., 2002; Withers, *et al*., 2001).

To date no work has been conducted on nutrient losses following the land application of anaerobically digested slurry to agricultural soils. Following the implementation of Renewable Energy Directive anaerobic digestion of animal slurry is likely to increase.

Consequently, further consideration needs to be given to the potential impacts of this process on nutrient export from agriculture and the associated environmental impacts. This is particularly important in light of the water quality objectives outlined in the EU Water Framework Directive (2000). Consequently, the aim of this experiment was to examine the effect of anaerobic digestion of slurry on P losses in surface runoff following slurry application to grassland.

MATERIALS AND METHODS

Site Description

This experiment was undertaken (7 September – 23 September 2010) at the Agri-Food and Biosciences Institute, Hillsborough, NI (54°27'N; 06°04'W). The 192 m^2 experiment area was located on a drumlin hill slope with an average slope of 4.5% and a north-easterly aspect. The area soil type was classified as a Soil Water Gley Class 1 soil overlying Silurian Shale (DANI Soil Survey of Northern Ireland) (FAO classification: Dystric Gleysol). The area has a Hydrology of Soil Types (HOST) classification of 24 which is indicative of poorly drained soils with high runoff rates, with this classification accounting for 46% of the land area in NI (Higgins, 1997). The soil has an average Olsen P content of 57.7 mg/l, twice the agronomic optimum, and an average bulk density of 0.83 g/cm^3 in the 0-5 cm horizon. Average annual rainfall and average annual evapotranspiration recorded at the site equated to 890 mm and 524 mm, respectively, for the period 1971-2000 (Betts, 1997). The average annual duration of the growing season was 254 days for the period 1951-1990 (Betts, 1997). Daily meteorological data for the site were supplied by a UK Meteorological Office weather station located 300 m from the field site.

The field site was last reseeded in 1997 with perennial ryegrass (*Lolium perenne*). Historically, silage has been cut at the site since 1985 supplying 2-4 harvests per annum with surplus winter grass grazed by livestock. From September 2009 until the start of the experiment (7 September 2010) livestock and machinery were excluded from the site to minimise the potential for 'hot spots' of compaction or high soil nutrient concentrations. To maintain an appropriate sward cover, grass throughout the period of animal exclusion was harvested from the plots at regular intervals using a self propelled mower (3600BM, Agria, Möckmühl, Germany) and gathered by hand. Between April 2010 and August 2010 chemical fertilisers in the form calcium ammonium nitrate was applied by hand on three occasions (12 April 2010, 16 June 2010 and 6 August) at rates equivalent to 100, 100 and 60 kg N/ha, respectively. Plots were cut on 7 September 2010 using Gardena battery-operated clippers (Accu 6, Kress and Kastner, Weiterstadt, Germany) fitted with a metal stand to standardise grass cutting height to 4.0 cm, with herbage removed by hand.

Four blocks (5.5 x 2 m) were constructed within the field site, with each block consisting of three plots (1.0 x 0.5 m) spaced 1.0 m apart across the block. The blocks were separated by a three metre deep buffer zone upslope of each block. Within each block, each plot was hydrologically isolated from its surrounding area by three stainless steel surrounds placed 0.05 m into the soil along the upslope edge and side of each plot. A V-shaped, stainless steel runoff collection tray (0.5 x 0.1 x 0.1 m) was inserted 0.03 m below the soil surface at the down-slope end of each sub-plot, perpendicular to the slope. Each collection tray had a 0.01 m shelf which was pushed into the soil beneath each plot (approximately 0.05 m), parallel to the soil surface. The runoff collection trays were connected to a two litre high density polyethylene (HDPE) plastic collection bottle by a 0.5 m length of braided PVC pipe buried underground. Collection trays were inserted two weeks before the experiment began to minimise disturbance to the soil. The stainless steel surrounds were inserted immediately following slurry application.

Slurry

The dairy cow slurry used in this experiment was collected from the Anaerobic Digestion unit at the Agri-Food and Biosciences Institute, Hillsborough, NI. The unit comprises of a 660 m^3 mesophilic digester which has a hydraulic retention time of 27 days, and is supplied with 20 m^3 of dairy cow slurry daily. A full description of the digester is available in Frost and Gilkinson (2010). The undigested slurry was collected from a 'reception tank' which fed the digester. The corresponding digestate for each UD sample was collected 27 days later from a similar outlet reception tank. Samples were collected on a weekly basis for a four-week period. These samples were then combined to create one bulk AD slurry and one UD slurry sample. Both slurries were macerated before entering the digestion plant. Samples were stored in

HDPE containers below 4ºC for 3 months prior to the beginning of the experiment. Two days prior to application the bulk samples were agitated and the appropriate quantities of each slurry measured out and stored in two-litre HDPE containers.

Treatments

Three treatments were examined in this randomised block design experiment: no slurry (control), undigested (UD) slurry, and anaerobically digested (AD) slurry. Each treatment was replicated four times and each block contained one replicate of each treatment. Slurry was applied on 7 September 2010 to the 12 0.5 m^2 plots. Slurry was poured against a rubber board above the plots and allowed to 'splash' onto the ground to simulate a splashplate spreading technique. Slurry was not applied to the area adjacent to the plot surrounds to minimise any edge effects which would arise from the insertion of the surrounds. As a result slurry was applied to an area 0.9 x 0.45 m inside the plot area. Both of the slurry treatments (AD and UD) slurry was applied so as to supply 1,000 mg of total P to each plot, a rate equivalent to 20 kg total P/ha. This represented a manure application rate of 33.6 and 32.4 m^3/ha for the AD and UD treatments, respectively, with the maximum permissible slurry application rate in NI equivalent to 50 m^3/ha (EHS, 2006).

Rainfall simulation

Rainfall simulations were performed on day 2 (RD2), 9 (RD9), and 16 (RD16) post slurry application. Two Amsterdam drip-type rainfall simulators, as described by Bowyer-Bower and Burt (1989), were employed to supply rainfall at a constant rate of 30 mm/hr. This rainfall intensity has a return period equivalent to a one in twenty-five year event for Northern Ireland (Cruickshank, 1997) and was selected to generate runoff during the dry weather conditions at the time of application. Water used in the rainfall simulations was passed through a DC9 general deionising cylinder (DC9, Purite Ltd, Thames Oxon, UK) to reduce its P concentration. The cylinder delivered deionised water with an average dissolved reactive phosphorus (DRP) and nitrate-N concentration of 11.7 μg/l and 181 μg/l, respectively. Wooden boards (1.2 m^2) were placed along two sides of the rainfall simulators to act as a wind shield throughout the experiment, and to prevent water droplets being blown outside the plot boundaries. Runoff volume and time taken to generate runoff were recorded. Thirty minutes of runoff was collection at each rainfall simulation in 2 x 15 minute fractions.

Water analysis

Water samples were refrigerated at 3°C within 4 hr of sampling and analysed for DRP, total dissolved P (TDP) and total P (TP) within 24 hr of sampling. Samples for nitrite-N (NO_2^--N), nitrate-N (NO_3^--N), and ammonium-N (NH_4^+-N) were frozen at -21°C and their analysis completed within two months of sampling. Total dissolved nitrogen (TDN) was determined on refrigerated, acidified samples within 3 weeks of collection. Samples for DRP, TDP, NO_2^--N, NO_3^--N, NH_4^+-N, and TDN were filtered through 0.45 μm filters (MF-Millipore, Billerica, MA) before analysis. Dissolved reactive P was determined by the acidic molybdate-ascorbic acid method of Murphy and Riley (1962). Total dissolved P and TP were determined by digestion with potassium persulphate and sulphuric acid, followed by analysis of the digest as outlined above for DRP (Eisenreich *et al*., 1975). Particulate P (PP) was calculated as the difference between TP and TDP. Dissolved unreactive P (DUP) was calculated from the difference between TDP and DRP.

Nitrite-N content was determined colorimetrically using a continuous flow analyser (Quaatro, Bran and Luebbe, Norderstedt, Germany) at 520 nm (Wood *et al*., 1967). Nitrate-N was determined through the cadmium-copper reduction to nitrite-N as outlined in Wood *et al*. (1967). Ammonium-N was also determined colorimetrically (Quaatro, Bran and Luebbe, Norderstedt, Germany) using the indophenol method of Scheiner (1976). Dissolved inorganic nitrogen (DIN) was calculated as the sum of NO_2^--N, NO_3^--N, and NH_4^+-N. Dissolved organic-N (DON) was inferred as the difference between TDN and DIN. Total dissolved nitrogen was measured using an Apollo 9000 TN module running in series with a total organic carbon combustion analyser (Apollo 9000, Teledyne Tekmar, Mason, Ohio). Following the combustion of TDN to nitrogen monoxide under a platinum catalyst, the reaction of nitrogen monoxide with ozone to give a chemiluminescent reaction was measured to give a corresponding value for TDN.

Suspended sediment concentration in the runoff samples was determined under suction filtration. A known volume of samples was passed through 1.2 um glass microfibre filters (GF/C; Whatman, Ltd, Maidstone, Kent). The filters were oven dried at 105°C and ashed at 550°C, both for 24 hr. Organic sediment was determined as the difference between the furnace and oven dried filter weights. Inorganic sediment was identified as the weight of the remaining ash residue.

Slurry analysis

Both the UD and AD slurries used in this experiment were analysed for dry matter (DM) content, WEP, total P and total N content. Dry matter content was determined gravimetrically by oven drying at 60ºC for 48 hr. Water extractable P was determined by the method outlined in Vadas (2006). One gram dry-weight equivalent of slurry was placed on an orbital shaker (60 min, 150 rpm) at a water:dry matter ratio of 200:1. Mixtures were filtered through 0.45 μm filter paper and following digestion by alkali persulfate, samples were analysed colorimetrically via the Murphy and Riley (1962) molybdate blue method. Total P was determined on a 2 gram ashed (500ºC) sample of slurry via digestion with hydrochloric acid and ammonium molybdate/tin (II) chloride reduction. The phosphor molybdenum blue complex was measured colorimetrically at 700 nm using a UV spectrometer (Lambda 2, PerkinElmer Inc., MA). Total N content was determined on a dried sample using the Kjeldahl technique outlined by Association of Official Agricultural Chemists (1983). Steam distillation following liberation by sodium hydroxide, allowed the titrimetric determination of ammonium-N (Association of Official Agricultural Chemists, 1983). The pH of both slurries was assessed using a pH electrode (736 GP Tritrino, Metrohm AG, Switzerland).

Statistical analysis

Statistical analysis was completed using Genstat version 12 software (VSN International Ltd, 2009, UK). Flow-weighted mean nutrient concentrations and total nutrient export quantities were calculated using the two 15-minute fractions of runoff collected. Nutrient concentrations in runoff and export rates from RD2 were analysed independently using a one-way analysis of variance (ANOVA). The effect of slurry treatment at the second runoff day (RD9) could not be analysed in this way, as these rainfall simulations were not independent from the previous runoff event at RD2. Consequently, the runoff generation data and runoff nutrient concentrations from RD2 and RD9, were analysed using a repeated measures analysis and an antedependence order 1 correlation. Slurry analysis was completed using the Students T-test.

RESULTS

Anaerobic digestion increased the DM content (*P*<0.001) and WEP content (*P*<0.001) of slurry by 14 and 22%, respectively (Table 1). Water extractable P accounted for 34% and 40% of total P in the UD and AD slurry, respectively, with the inorganic P fraction accounting for 88% of WEP in both slurries. The total P content of slurry remained unchanged following anaerobic digestion. Similarly, anaerobic digestion had no effect on the total N content of the slurry (*P*>0.05) however the ammonium-N content of slurry was significantly greater following anaerobic digestion (*P*<0.001).

On the day of slurry application, total rainfall, average air temperature, and volumetric soil moisture content (VSM) were 3.8 mm, 14.2^{o}C and 36.7%, respectively (Figure 1). As a result of two dry days following slurry application, the average VSM content fell to 29.7% at RD2. Thereafter a number of prolonged natural rainfall events occurred, causing fluctuations in VSM content but by RD9 the VSM content had recovered to 33.9%. From the time of slurry application (7 September 2010) until the second rainfall simulation event (RD9; 16 September 2010), total rainfall, average daily temperature, and the average VSM content were 37.2 mm, 13.9^{o}C and 33.7%, respectively. During the seven-day period following RD9, there was a further 51.7 mm of rainfall, with 24.5 mm of this rainfall falling on the day prior to the planned RD16 measurements (RD16, 23 September 2010). As a result, the rainfall simulations on RD16 were not carried out due to water-logging at the experimental site.

There was no significant effect (*P*>0.05) of slurry treatment on either runoff volume or the time taken to generate runoff at either RD2 or RD9 (Table 2). The average runoff volumes across both RD events for the Control, UD, and AD treatments were 0.93, 0.83 and 1.25 litres, respectively. Likewise, the average time taken to generate runoff did not differ between RD2 and RD9, with average runoff generation times of 911 and 853 seconds, respectively. Runoff volume at RD9 tended to be higher than at RD2 (1.231 *vs.* 0.734 litres) however, this was only at the 10% significance level.

At RD2, DRP, DUP, PP and TP concentrations in runoff were significantly greater (P<0.05) from the slurry (UD and AD) treatments than the Control treatment (Table 3). For the Control treatment 66% of TP in runoff at RD2 was present in dissolved form, of which 88% was present as DRP. Dissolved P accounted for 51% of the TP concentration in runoff from the UD treatment, of which 75% was present as DRP. In contrast, dissolved P accounted for 78% of TP concentrations in runoff from the AD treatment, 90% of which was present as DRP. Although only significant at the 10% level (P=0.083), concentrations of DRP in runoff at RD2 were 68% higher from the AD treatment (4.67 mg/l) than the UD treatment (2.78 mg/l). In contrast, DUP and PP concentrations in runoff from the AD treatment were 45 and 59% lower (P<0.05), respectively, than the UD treatment at RD2, however, TP concentrations in runoff did not differ between these two treatments (P>0.05).

Throughout the duration of the experiment (RD2 + 9), P concentrations in runoff were significantly greater (P<0.05) from the UD and AD slurry treatments than the control treatment (Table 3). Neither DRP nor TP concentrations in runoff differed between the UD and AD treatments (P>0.05), while both DUP and PP concentrations in runoff from the UD treatment were higher than the AD treatment (P<0.05).

Dissolved reactive phosphorus, DUP, PP and TP concentrations in runoff were significant greater (P<0.05) at RD2 in comparison to RD9 (Table 3). No significant difference in runoff P concentrations was evident for the Control treatment between the two RD events. On average, P concentrations from the slurry treatments at RD9 were 53, 47, 86 and 64% lower than RD2 for DRP, DUP, PP and TP, respectively. A significant (P<0.001) slurry treatment x runoff day interaction was evident for PP concentrations in runoff where, in contrast to RD2, PP concentrations from the UD and AD treatments at RD9 did not differ from the Control.

Neither DRP nor TP export in runoff was affected by slurry treatment at RD2 or RD2 + RD9 (P>0.05; Table 4). In contrast, DUP export rates from the Control treatment were lower than from either the UD or AD treatments (P<0.05) at both RD2 and RD2 + RD9, while DUP export did not differ between the UD and AD treatments (P>0.05).

Particulate P export rates at RD2 and RD2 + RD9 from the UD treatment were higher than from the Control treatment ($P<0.05$).

Nitrate-N, ammonium-N and organic-N concentrations in runoff at RD2, and RD2 + RD9, were significantly higher ($P<0.05$) from the UD and AD treatments in comparison to the Control treatment (Table 5) while concentrations did not differ between treatments UD and AD ($P>0.05$). In contrast there was no significant effect ($P>0.05$) of treatment on nitrite-N concentrations in runoff at RD2 or RD2 + RD9. Nitrite-N ($P<0.05$), ammonium-N ($P<0.001$) and organic-N ($P<0.001$) concentrations in runoff were greater at RD2 than at RD9.

There was no significant effect of slurry treatment on suspended sediment concentrations in runoff at either RD2 or RD2 + RD9 (Table 6). Inorganic sediment concentrations did not differ between RD2 and RD9, whilst organic sediment ($P<0.01$) and total suspended sediment ($P<0.05$) concentrations were significantly greater at RD2 in comparison to RD9. There was no slurry treatment x runoff day interaction for any of these parameters ($P>0.05$).

DISCUSSION

The changes in slurry composition induced by the anaerobic digestion process in this experiment are generally in agreement with the current literature on the digestion of animal manures. The reduction in the DM content of slurry following anaerobic digestion reflects the breakdown of organic matter in the digester and consequently the loss of carbon from the substrate to carbon dioxide and methane production (Smith *et al*., 2007). This 14% reduction in DM content falls within the range of observed DM reductions (range 8-34%; mean 19%) witnessed in the AFBI digester over a 18-month period (Frost and Gilkinson, 2010) but is considerably smaller than the average reduction of 25% in dry matter content witnessed by Smith *et al*. (2007) in a wider survey of 12 slurry-fed anaerobic digesters. The increase in pH (0.7 units) observed in the current experiment was also concurrent with other studies of anaerobic digestion and is associated with the destruction of volatile fatty acids by the methanogenic organisms present in the digester (Smith *et al*., 2007; Topper *et* *al.,* 2006). Smith *et al*. (2007) noted an average pH increase of 0.4 units across 11 digesters whilst Möller and Stinner (2010) also observed a 0.8 unit increase in pH following anaerobic digestion of cattle slurry. In agreement with other studies anaerobic digestion of slurry did not alter slurry total P content (Smith *et al*., 2007; Güngör *et al*., 2007). In contrast, anaerobic digestion significantly altered the fractionation of P within the slurry used in the current experiment. However, elsewhere no consensus has been reached on the effect of anaerobic digestion on slurry P fractions, with differences thought to reflect variations in substrate mineral composition (Güngör and Karthikeyan, 2005). Nonetheless, the majority of the literature on AD describes an increase in WEP and orthophosphate in slurry following digestion (Smith *et al*., 2007; Marti *et al*., 2008), with the 22% increase in WEP witnessed in the current experiment similar to values reported by Martin (2004). This increase in orthophosphate is primarily driven by the breakdown of organic complexes by organisms in the digester leading to the solubilisation of P and an increase in both inorganic and WEP forms. Wright (2004) in a study of five dairy cow slurry anaerobic digestion systems, recorded variable increases (8-61%) in slurry orthophosphate concentrations between digesters. Moody *et al*. (2009) also noted a 26% increase in orthophosphate from anaerobic digestion of swine slurry, however, this was not significantly different from the undigested slurry. Finally, whilst total N content of slurry has not been found to change under anaerobic digestion, a higher proportion of the total N has been found to be present as ammonium-N following the breakdown of protein molecules in the slurry. Smith *et al*. (2007), Möller and Stinner (2010), and Frost and Gilkinson (2010) all present increases (20-26%) in ammonium-N content following anaerobic digestion, similar to that recorded in the current experiment (22%).

The absence of a significant effect of slurry application on runoff volume or time taken to generate runoff in the current experiment is consistent with findings from other authors (Kleinman and Sharpley, 2003; Smith *et al*., 2001). A few authors have observed increases in runoff volume following the surface application of slurry by splashplate application methods as a result of the surface sealing of soil by slurry, preventing the infiltration of rainfall (Little *et al*., 2005; Ramos *et al*., 2006). However, in the current experiment, the low DM content of slurry would have resulted in the rapid infiltration of the liquid fraction of slurry into the soil, decreasing the possibility of surface sealing.

The absence of a significant difference between DRP concentrations in runoff from the UD and AD treatments, despite significantly higher levels of WEP in the AD slurry was unexpected. It is hypothesised that this is due to the differences in DM between the UD (DM = 67.7 g/kg fresh) and AD (DM = 58.5 g/kg fresh) slurry. At lower slurry DM contents, less inorganic WEP will remain on the soil surface, for example Vadas (2006) reporting a significantly lower proportion of slurry matter remaining on the soil surface 24 hr after application from swine slurry (6.3% DM) in comparison to dairy slurry (8.0% DM). In their experiment, the lower DM content of the swine slurry was thought to facilitate a higher rate of infiltration than the dairy slurry resulting in a lower proportion of applied inorganic WEP remaining on the soil surface. Once infiltrated into the soil, this inorganic WEP is thought to rapidly sorb to soil complexes rendering it inaccessible to surface runoff (Pierzynski *et al*., 2005).

In addition, although not significant, numerically DRP concentrations and export rates were 168% and 267% higher, respectively, from the AD treatment at RD2 in comparison to the UD treatment. This was most likely a direct response to the elevated inorganic WEP concentrations identified in the AD slurry following the digestion process. Inorganic WEP is predominantly the largest fraction of WEP in slurry, accounting for over 80% WEP (Vadas *et al*., 2007; Chapuis-Lardy *et al*., 2004) and the rapid nature with which dissolution of inorganic WEP occurs under contact with water means it is highly susceptible to runoff transfers (He *et al*., 2004; Dou *et al*., 2000). Indeed, a number of authors have highlighted the strong positive relationship between manure WEP content and DRP concentrations in runoff (Kleinman *et al*., 2002; Withers *et al*., 2001) to the extent that manure WEP can in many cases be used as predictor for DRP concentrations in runoff (Vadas *et al*., 2007; Vadas, 2006).

In the current experiment, the higher DUP concentrations in runoff at RD2 from the UD treatment are also unexpected in light of the significantly higher organic WEP concentrations in the AD slurry relative to the UD slurry. This difference may also be attributed to differences in infiltration rates between the AD and UD treatments. Like inorganic WEP, the absorption of organic WEP onto soil particles and minerals takes place following the infiltration of slurry into soil, making is less accessible to runoff (Condron *et al*., 2005). In some cases, authors have noted that upon infiltration of slurry into soil, soluble organic P compounds in manure are preferentially sorbed onto soil compounds in place of inorganic P, with this process going so far as to displace some orthophosphate previously bound to soil structures (Marshall and Laboski, 2006; Anderson *et al*., 1974). In addition organic WEP is known to be a highly dynamic fraction of P and can vary with experimental conditions (Vadas, 2006). The higher concentrations of PP in runoff from the UD treatment relative to the AD treatment in the current experiment are most likely indicative of the higher proportion of slurry P in non water-extractable forms in the UD slurry, however to date, there is no literature to support this finding.

The reduction in P concentrations in runoff and P export rates from the UD and AD treatments between RD2 and RD9 across all fractions of P can be attributed to slurry infiltration into the soil. As previously discussed, the infiltration of the liquid slurry fraction into the soil between RD2 and RD9 facilitates the rapid sorption of P onto the surface of soil molecules, reducing the amount of both DRP and DUP available for transport at RD9. In addition, the removal of P in the previous rainfall simulation event at RD2 would have resulted in a depletion of the store of P on the soil surface and consequently the amount of P available for transport in runoff at RD9 (Kleinman and Sharpley, 2003). However, as P transport in runoff at RD2 was less than 2% of that applied in slurry, this is most likely not the dominant control on P losses. With regard to PP, the decrease in PP concentrations between RD2 and RD9 may also be both a result of manure decomposition and assimilation into the soil (DARD, 2011; Vadas *et al*., 2011), and a response to the increasing level of sward interception from the sward regrowth which had developed on each plot since grass was harvested immediately prior to slurry application.

The higher ammonium-N concentrations recorded in the AD slurry did not result in elevated levels of ammonium-N in runoff from the AD treatment plots. This may have reflected both the potentially higher levels of ammonia volatilisation from these AD plots upon application and the rapid utilisation of ammonium-N upon infiltration to the soil in the first 48 hr post slurry application (Chadwick and Chen, 2002). The rapid decrease in nitrite-N and ammonium-N concentrations in runoff between RD2 and RD9 highlights the highly volatile nature of inorganic N compounds and its susceptibility, once infiltrated, to be either utilised by the plant or translocated down the soil profile away from the soil surface (Edwards *et al*., 2000). In contrast, nitrate-N concentrations in runoff did not fall between RD2 and RD9 from either slurry treatment. This was most likely a result of the breakdown of organic N into nitrate-N species between RD2 and RD9, fuelling the high nitrate-N concentrations witnessed at RD9. Similar variations in nitrate-N patterns were also identified by Smith *et al*. (2007) who found elevated concentrations of nitrate-N in runoff eight days after slurry application. Likewise Sharpley (1997) whilst identifying a significant reduction in ammonium-N concentrations over time, found no effect of rainfall frequency or duration on nitrate-N concentrations in runoff.

Although not significantly different, the high levels of organic sediment in runoff from the UD and AD treatments at RD2 provide a good indication of the level of contamination of runoff water with slurry when compared with that of the Control. This is similar to findings by Pote *et al*. (2003) who noted elevated levels of suspended solids in runoff following the surface application of poultry litter to grasslands in comparison with a control. The absence of a significant difference in sediment concentrations between the two slurries despite different DM contents may be due to the fact that both slurries were macerated before application leading to a more uniform particle size distribution in the slurry. The significant reduction in organic suspended sediment concentrations between RD2 and RD9, again highlights the integration of slurry into the soil through the processes of decomposition and assimilation (Vadas *et al*., 2011).

CONCLUSIONS

The anaerobic digestion process significantly altered the physical and chemical properties of the dairy cow slurry used in this experiment notably causing a reduction in DM content and an increase in the WEP content of the slurry. However, this increase in WEP content did not result in significantly higher DRP concentrations in runoff at day 2 or day 9 post slurry application. It is likely that a higher level of slurry infiltration, associated with the lower DM content of the AD slurry, promoted the infiltration of WEP into the soil, hence allowing the rapid sorption of WEP in the upper soil horizons to take place. Particulate P concentrations in runoff were higher from the UD slurry treatment reflecting the higher organic matter content, and PP in the UD slurry. Whilst high concentrations of PP do not have an immediate detrimental impact on receiving waterbodies, over time the conversion of PP to bio-available P may present a pollution risk in aquatic environments. The current study showed no significant effect of slurry treatment on N concentrations in runoff, however, the long signal of nitrate-N in runoff, in comparison with other N fractions, highlights again the potential risk of water pollution associated with application of slurry to grasslands. Likewise, there was no significant effect of slurry treatment on suspended sediment concentrations in runoff. Consequently, the results of this study suggest that the continued expansion of the AD industry will not pose a significant threat to water quality with relation to P runoff.

ACKNOWLEDGEMENTS

This work was co-funded by the Department of Agriculture and Rural Development for Northern Ireland and AgriSearch, Northern Ireland. The authors thank the staff of the dairy unit for technical assistance and members of the AFBI agriculture and freshwater laboratories for undertaking chemical analysis.

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Table 1: Chemical composition of the undigested (UD) and anaerobically digested slurry (AD) applied during the experiment

	UD (g/kg	AD (g/kg)
Dry matter	67.7	58.5
pH	7.24	7.87
Total phosphorus	0.58	0.59
Water extractable phosphorus (WEP)	0.20	0.24
Inorganic WEP	0.17	0.21
Organic WEP	0.02	0.03
Total nitrogen	3.17	3.21
Ammonium-nitrogen	2.17	2.65

Table 2: Effect of slurry treatment and days post slurry application on runoff characteristics observed throughout the study

Runoff day (days post slurry application)	Slurry Treatment	Runoff volume† (litres)	Time taken to generate runoff (seconds)
2	Control	0.66	1118
	UD	0.65	598
	AD	0.90	919
9	Control	1.21	917
	UD	1.01	947
	AD	1.61	695
Slurry treatment	SIG	NS	NS
Runoff day	SIG	NS	NS
Slurry x Runoff‡	SED	0.390	213.6
	SIG	NS	NS

Control = no slurry, UD = undigested slurry. AD = anaerobically digested slurry.
† Runoff collected in the first 30 minutes following initiation of runoff.
‡ Slurry treatment x runoff day interaction.
NS = P>0.05

Table 3: The effect of slurry treatment on flow-weighted mean phosphorus concentrations in runoff generated at days 2 and 9 following slurry application

Runoff day (days post slurry application)	Slurry treatment	DRP (mg/l)	DUP (mg/l)	PP (mg/l)	TP (mg/l)
2	Control	0.51[a†]	0.07[a]	0.21[a]	0.79[a]
	UD	2.78[ab]	0.95[c]	3.63[c]	7.36[b]
	AD	4.67[b]	0.52[b]	1.49[b]	6.67[b]
	SED	1.237	0.170	0.560	1.718
	SIG	*	*	**	*
9	Control	0.22	0.06	0.23	0.51
	UD	1.99	0.47	0.42	2.87
	AD	1.52	0.31	0.29	2.12
Mean of days 2 and 9	Control	0.36[a]	0.07[a]	0.22[a]	0.65[a]
	UD	2.39[b]	0.71[c]	2.03[c]	5.12[b]
	AD	3.10[b]	0.41[b]	0.89[b]	4.40[b]
	SED	0.663	0.071	0.286	1.291
Slurry treatment	SIG	**	***	***	***
Runoff Day	SIG	NS	*	***	***
Slurry x Runoff‡	SED	0.956	0.072	0.404	1.291
	SIG	NS	*	***	NS

† Values with the same letter in each column, within each factor, are not significantly different (P>0.05)
‡ Slurry treatment x runoff day interaction
* P<0.05, ** P<0.01, *** P<0.001, NS = P>0.05
Control = no slurry, UD = undigested slurry. AD = anaerobically digested slurry
DRP = dissolved reactive P, DUP = dissolved unreactive P, PP = particulate P, TP = total P

Table 4: The effect of slurry treatment on phosphorus export rates in runoff at runoff days 2 and 9 throughout the experiment

Runoff day (days post slurry application)	Slurry treatment	Export Rate (g/ha)			
		DRP	DUP	PP	TP
2	Control	6.3	0.2[a†]	2.9[a]	10.1
	UD	38.9	12.4[b]	48.3[b]	99.6
	AD	104.0	10.2[b]	26.7[ab]	141.1
	SED	47.31	2.58	12.96	60.72
	SIG	NS	**	*	NS
9	Control	5.5	1.7	6.5	13.6
	UD	40.3	8.9	8.5	58.0
	AD	36.3	7.6	5.5	49.1
Mean of days 2 and 9	Control	5.9	0.9[a]	4.7[a]	11.9
	UD	39.6	10.7[b]	28.4[b]	78.8
	AD	70.6	8.9[b]	16.1[ab]	95.1
	SED	24.88	2.40	7.10	32.52
Slurry treatment	SIG	NS	**	*	NS
Runoff Day	SIG	NS	NS	*	NS
Slurry x Runoff‡	SED	35.18	3.81	10.00	45.98
	SIG	NS	NS	*	NS

† Values with the same letter in each column, within each factor, are not significantly different (P>0.05)
‡ Slurry treatment x runoff day interaction
* P<0.05, ** P<0.01, *** P<0.001, NS = P>0.05
Control = no slurry, UD = undigested slurry. AD = anaerobically digested slurry
DRP = dissolved reactive P, DUP = dissolved unreactive P, PP = particulate P, TP = total P

Table 5: The effect of slurry treatment on flow-weight mean concentrations of nitrite, nitrate, ammonium, and organic nitrogen in runoff at day 2 and day 9 following slurry application

Runoff day (days post slurry application)	Slurry treatment	Nitrite (mg/l)	Nitrate (mg/l)	Ammonium (mg/l)	Organic Nitrogen (mg/l)
2	Control	0.01	0.70[a†]	0.09[a]	2.62[a]
	UD	0.32	3.81[b]	6.57[b]	7.40[b]
	AD	0.37	2.53[b]	5.42[b]	9.12[b]
	SED	0.156	0.850	1.930	2.041
	SIG	NS	*	*	*
9	Control	0.01	0.14	0.01	0.90
	UD	0.03	3.28	0.17	4.71
	AD	0.04	2.91	0.17	3.62
Mean of days 2 and 9	Control	0.01	0.59[a]	0.05[a]	1.76[a]
	UD	0.18	4.43[b]	3.37[b]	6.05[b]
	AD	0.20	3.39[b]	2.74[b]	6.37[b]
	SED	0.088	1.486	0.919	1.197
Slurry treatment	SIG	NS	**	**	**
Runoff day	SIG	*	NS	***	**
Slurry x RD‡	SED	0.124	0.976	1.285	1.689
	SIG	NS	NS	*	NS

† Values with the same letter in each column, within each factor are not significantly different ($P>0.05$)
‡ Slurry treatment x runoff day interaction
* $P<0.05$, ** $P<0.01$, *** $P<0.001$, NS = $P>0.05$
Control = no slurry, UD = undigested slurry. AD = anaerobically digested slurry

Table 6: The effect of slurry treatment on suspended sediment concentrations measured in runoff at day 2 and day 9 following slurry application

Runoff day (days post slurry application)	Slurry treatment	Inorganic sediment (mg/l)	Organic sediment (mg/l)	Total sediment (mg/l)
2	Control	0.05	0.02	0.06
	UD	0.08	0.27	0.35
	AD	0.07	0.25	0.32
	SED	0.017	0.423	0.426
	SIG	NS	NS	NS
9	Control	0.07	0.04	0.13
	UD	0.05	0.06	0.13
	AD	0.08	0.05	0.08
Mean of days 2 and 9	Control	0.08	0.03	0.10
	UD	0.05	0.15	0.21
	AD	0.07	0.16	0.23
	SED	0.005	0.043	0.052
Slurry	SIG	NS	NS	NS
Runoff day	SIG	NS	**	*
Slurry x Runoff†	SED	0.018	0.060	0.084
	SIG	NS	NS	NS

Control = no slurry, UD = undigested slurry. AD = anaerobically digested slurry
† Slurry treatment x runoff day interaction
* P<0.05, ** P<0.01, NS = P>0.05

Table 7: Effect of slurry treatment on herbage yield, herbage dry matter (DM) content, and herbage phosphorus content 50 days after slurry application

Slurry treatment	Herbage yield (kg DM/ha)	Herbage dry matter content (g/kg)	Herbage phosphorus content (g/kg DM)
Control	1745	147	3.96
	2066	140	4.15
	2239	138	3.93
	356.9	3.4	0.139
	NS	NS	NS

Control = no slurry, UD = undigested slurry. AD = anaerobically digested slurry
NS = *P*>0.05

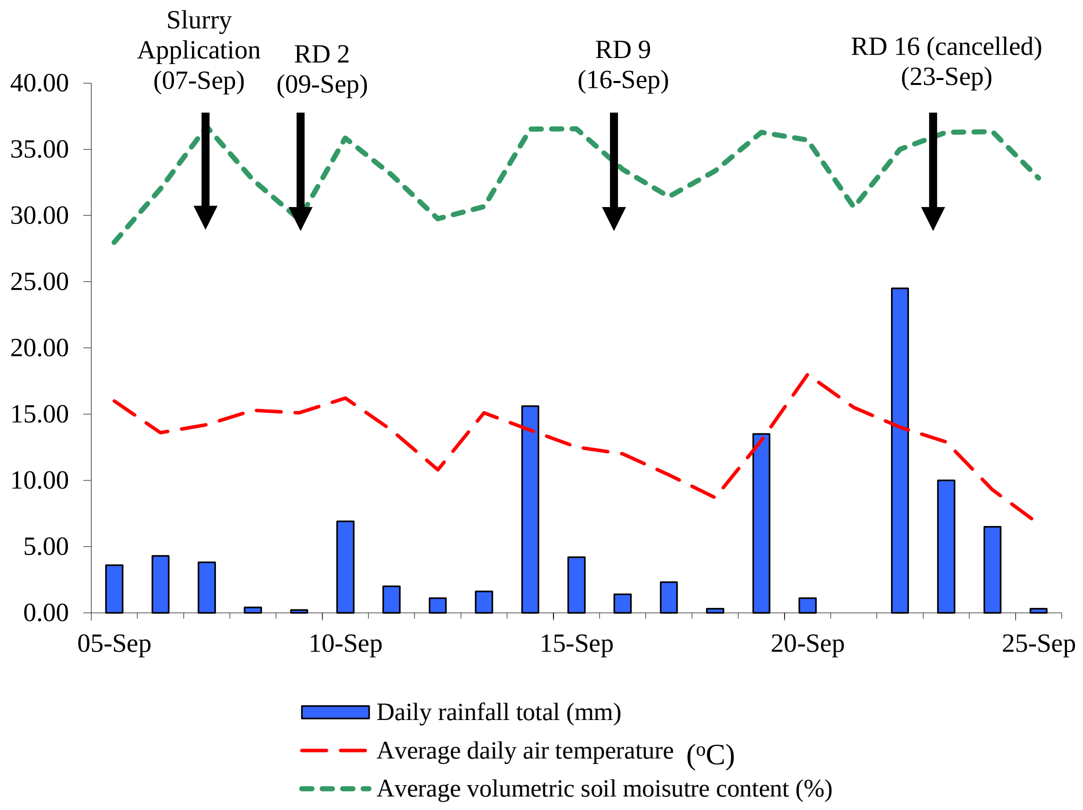

Figure 1: Daily rainfall, average daily air temperature, and average daily soil moisture deficit recorded throughout the experimental period, together with slurry application and runoff days (RD) dates