Nutrients Action Programme (NAP)
2019 Derogation Report for Northern Ireland

December 2020

Department of Agriculture, Environment and Rural Affairs
www.daera-ni.gov.uk

Agri-Food and Biosciences Institute

Northern Ireland Environment Agency
www.daera-ni.gov.uk

College of Agriculture, Food & Rural Enterprise

CONTENTS **Page No**

This report has been prepared by scientific, regulatory, advisory and policy staff from the:-

Department of Agriculture, Environment and Rural Affairs (DAERA); www.daera-ni.gov.uk

Northern Ireland Environment Agency (NIEA) (an agency within DAERA); www.daera-ni.gov.uk/northern-ireland-environment-agency

College of Agriculture, Food and Rural Enterprise (CAFRE) (a division within DAERA); www.cafre.ac.uk/

Agri-Food and Biosciences Institute (AFBI) (a Non-Departmental Public Body sponsored by DAERA); www.afbini.gov.uk

For further information please contact:-
Environmental Farming Branch
Department of Agriculture, Environment and Rural Affairs
R652 Dundonald House
Upper Newtownards Rd
Ballymiscaw
Belfast
BT4 3SB

Email: linda.mcgoldrick@daera-ni.gov.uk

1. INTRODUCTION

The Nitrates Directive (91/676/EEC) (the Directive) is currently implemented in Northern Ireland through the 2019-2022 Nutrients Action Programme (NAP) contained in the Nutrient Action Programme Regulations (Northern Ireland) 2019 (the 2019 NAP Regulations) and subsequent amending regulations[1]. From 2015-2018 the Directive was implemented through the NAP contained in the 2014 NAP Regulations and subsequent amending regulations[2]. The Regulations limit the amount of nitrogen (N) from livestock manure that can be applied to land to 170 kg N/ha/year on all non-derogated farms and are the responsibility of the Department of Agriculture, Environment and Rural Affairs (DAERA).

The measures contained in the 2014 NAP Regulations were carried forward into the 2019 NAP Regulations. However, a range of additional measures were included in 2019 and updated guidance[3] on the NAP was produced for farm businesses.

In 2007, the United Kingdom (UK), with regard to Northern Ireland, was first granted derogation (until 31 December 2010) by Commission Decision 2007/863/EC to permit an increase in the amount of grazing livestock manure that may be applied to land from 170 kg N/ha/year up to a limit of 250 kg N/ha/year on grassland farms which meet certain criteria. After application the derogation was renewed by Commission Decisions 2011/128/EU, 2015/346/EU and 2019/1325/EU. Measures relating to the 2019 Decision have been included in the 2019 NAP Regulations.

Each year Northern Ireland produces a report on implementation of the derogation, water quality and evaluation practice It includes maps showing the percentage of grassland farms, percentage of livestock and percentage of agricultural land covered by an individual derogation in each district of Northern Ireland, as well as maps of local land use. Details are also provided on controls at farm level and information on non-compliant farms based on results of administrative and field inspections.

Reports are produced annually for the preceding year, and include maps for the current year. Therefore, this report provides the annual report on implementation of the derogation in 2019 and maps for 2020. The categories within the maps have been standardised to allow for trend analysis.

[1] The Nutrient Action Programme (Amendment) Regulations (Northern Ireland) 2019
[2] The Nitrates Action Programme (Amendment) Regulations (Northern Ireland) 2015
[3] Nutrients Action Programme 2019-2022 Guidance – www.daera-ni.gov.uk/articles/nutrient-action-programme-regulations-2019-2022

2. MAPS

In 2019, 441 farm businesses out of approximately 24,594 direct aid claimants (i.e. 1.8%) operated under an approved derogation in Northern Ireland, compared to 474 (i.e. 1.9%) in 2018.

Table 1 on page 12, shows the predicted grassland area and livestock manure loadings of farm businesses which applied to operate under the terms of the derogation in years 2011 to 2019.

Under the Water Framework Directive (2000/60/EC) (WFD), Northern Ireland shares three International River Basin Districts (IRBDs) with the Republic of Ireland and there is one River Basin District (RBD) entirely within Northern Ireland. In the 'UK Article 3 Report on the WFD', Northern Ireland was further subdivided into 31 sub-catchments which form the basis of the maps presented below.

2.1 Percentage of grassland farms covered by an individual derogation in 2020

The map in Figure 1 shows the percentage of grassland by maximum eligible area (MEA) of farms who applied for an individual derogation in 2020, broken down by location of the land within the 31 sub-catchments. Across Northern Ireland this equates to 4.6% of the available grassland. The highest percentage in any sub-catchment is 16.13%.

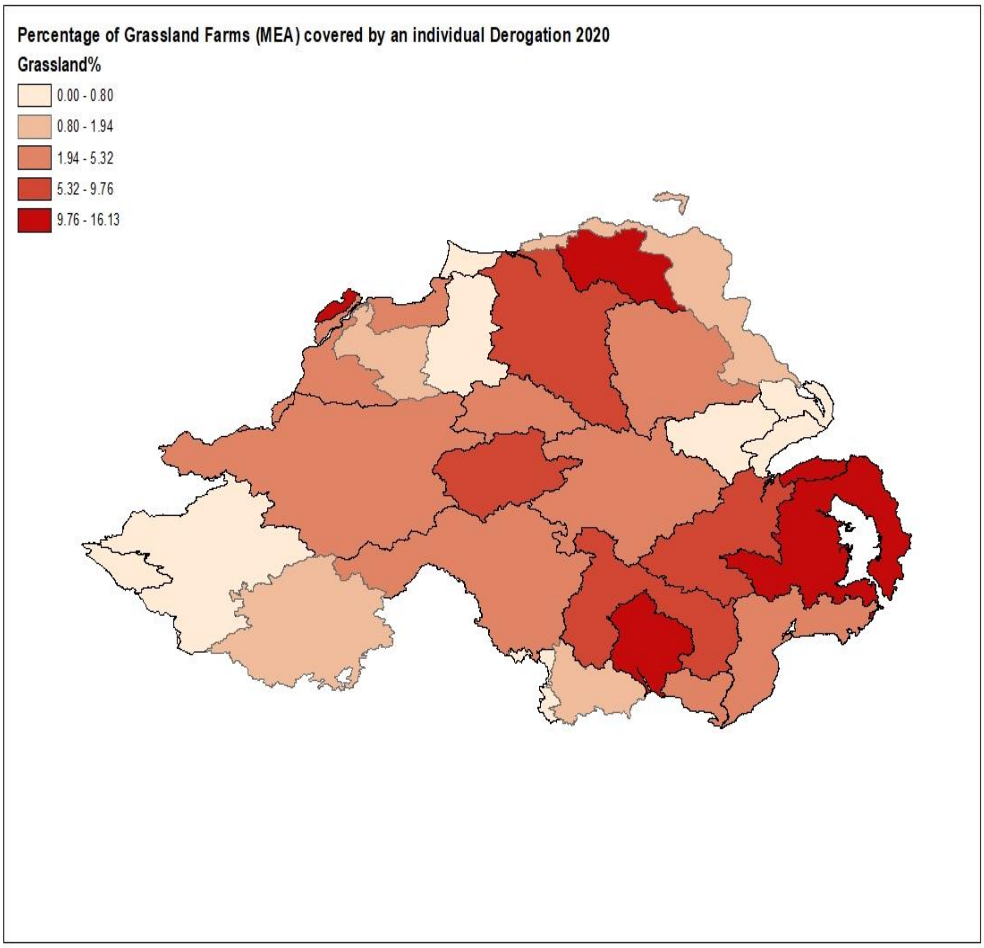


Figure 1: Percentage of grassland covered by an individual derogation in 2020

2.2 Percentage of cattle livestock covered by an individual derogation in 2020

Figure 2 illustrates the percentage of cattle livestock covered by an individual derogation in 2020. This has been calculated on the basis of nitrogen (N) produced and is broken down by location of the farm business address within the 31 sub-catchments. Across Northern Ireland this equates to 10.7% of cattle livestock N produced. The highest percentage in any sub-catchment is 34.07%.

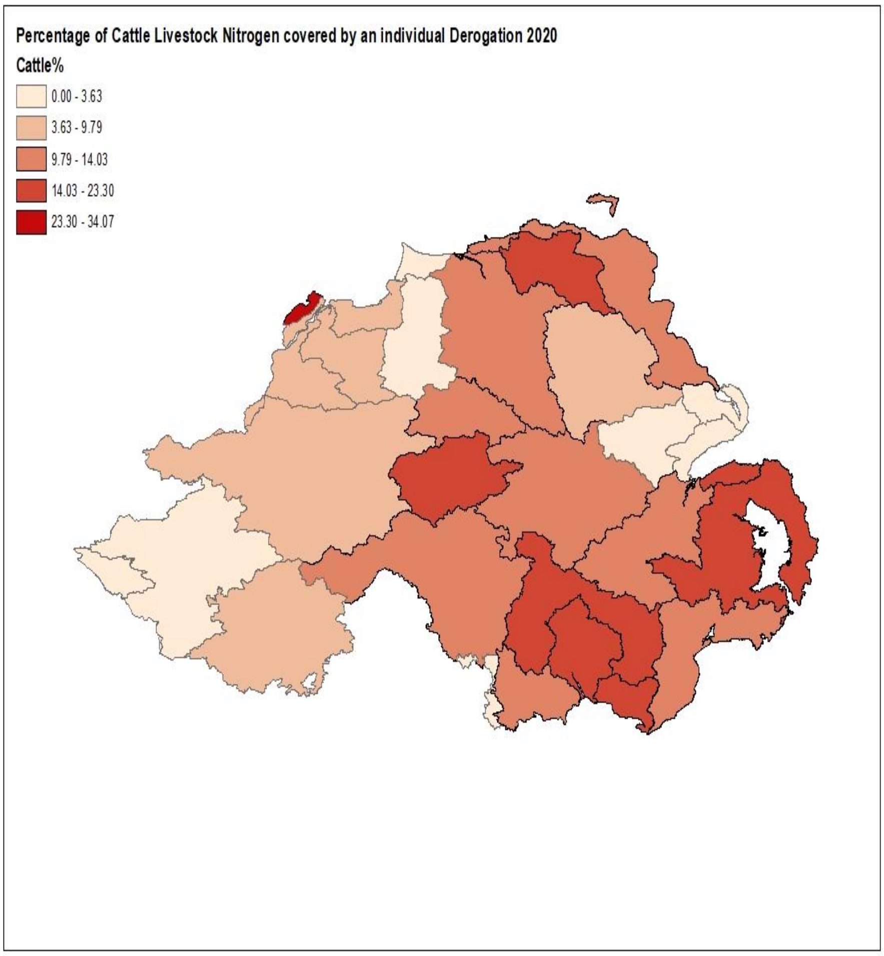


Figure 2: Percentage of cattle livestock covered by an individual derogation in 2020

2.3 Percentage of agricultural land covered by an individual derogation in 2020

Figure 3 illustrates the percentage of agricultural land covered by farm businesses who applied for an individual derogation in 2020. This has been calculated utilising the location of the farm business address within the 31 sub-catchments. Across Northern Ireland this equates to 3.9% of agricultural land. The highest percentage in any sub-catchment is 13.84%

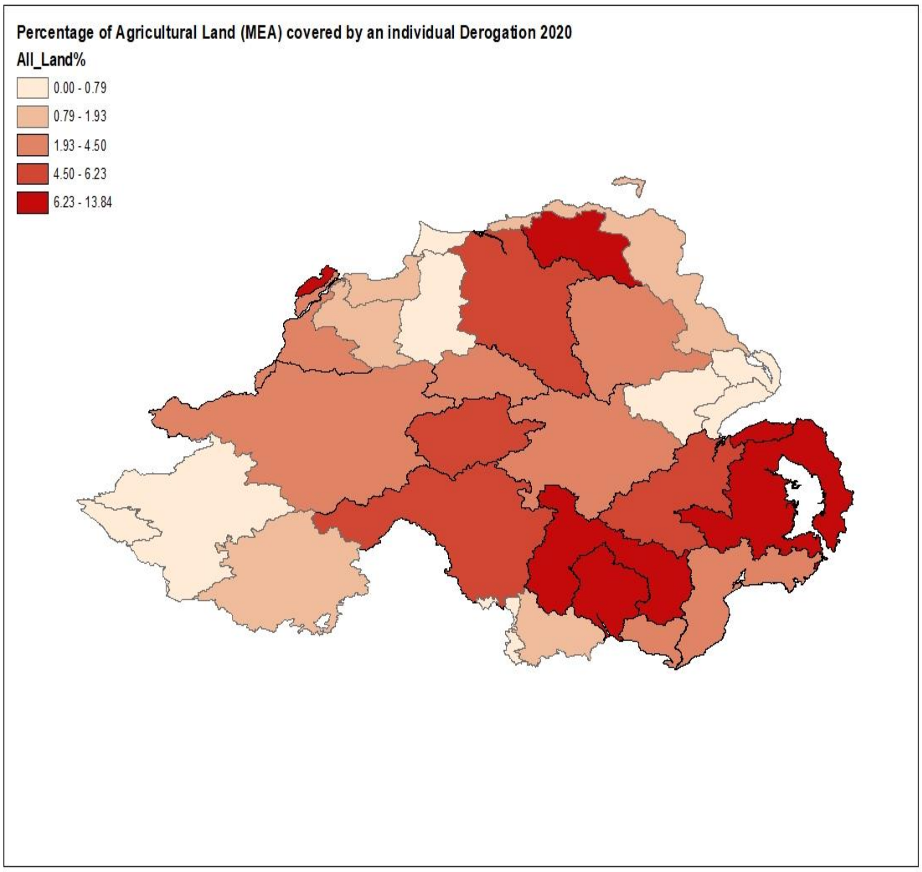


Figure 3: Percentage of agricultural land covered by an individual derogation in 2020

2.4 Map of local land use for 2019

Agricultural land use in Northern Ireland is dominated by grassland farming systems. According to the Northern Ireland Agricultural Census (June 2019) (https://www.daera-ni.gov.uk/publications/agricultural-census-northern-ireland-2019), managed grassland accounted for approximately 79% of a total agricultural area of 1,023,200 ha. Arable and other crops accounted for 4% of the total and rough grazing for 14%. The map in Figure 4 shows the declared land-uses across Northern Ireland from the 2019 SAF application.

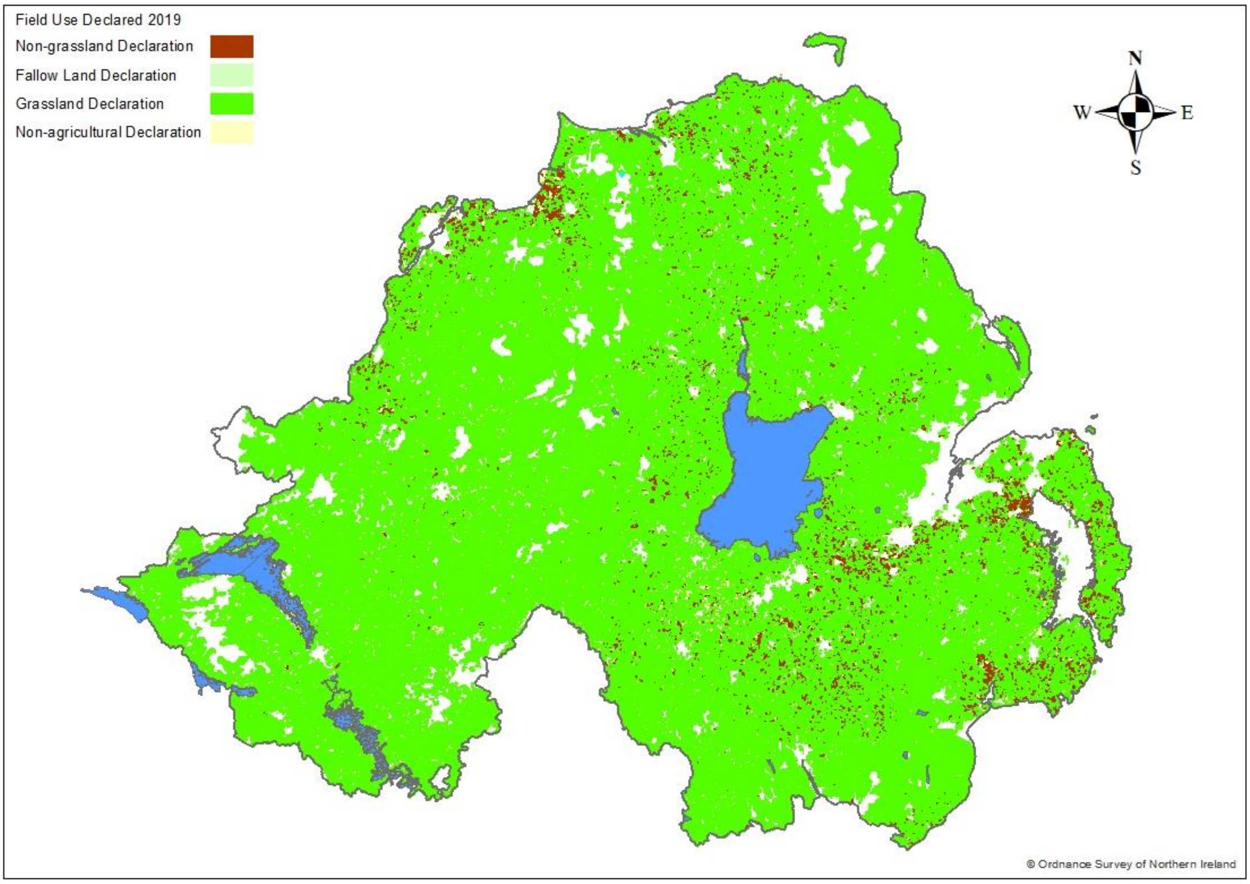


Figure 4: Map of local land use for 2019

Table 1: Predicted average (and minimum-maximum) grassland areas and livestock manure loadings of farm businesses which applied for derogation in years 2011 to 2019

Parameter	2011	2012	2013	2014	2015	2016	2017	2018	2019
Grassland area (%)	98 (81-100)	97 (81-100)	98 (82–100)	98 (82–100)	98 (83–100)	98 (82–100)	98 (81–100)	98.322 (80-100)	98 (80-100)
Farm size (ha)	84 (11–261)	83 (14–260)	85 (14–280)	88 (14–272)	86 (10–370)	88 (7-334)	90 (16-348)	85 (7-358)	86 (7-359)
Total livestock manure nitrogen loading (kg N/ha/year)	206 (155–250)	204 (119–249)	205 (36–249)	205 (122–246)	205 (12–250)	213 (23-250)	215 (22-250)	187 (0 – 275)	210 (0-250)
Grazing livestock manure nitrogen loading (kg N/ha/year)	206 (195–250)	204 (119–249)	205 (36–249)	205 (122–246)	206 (12–250)	213 (23-250)	215 (22-250)	191 (0 – 275)	250 (250-250)

3. WATER QUALITY

In accordance with Article 10 of the 2019 Decision, the results of monitoring and a concise report on water quality are transmitted to the Commission annually in this Derogation report.

The following section provides information on the measured nitrate and phosphorus levels and evolution of water quality in rivers, streams, lakes and groundwater over the period 2012 to 2019. Results are assessed both for Northern Ireland as a whole, and for the sub-catchments (Crawfordsburn, Strangford, Ballinderry, Clanrye and Upper Bann) where the concentration of derogated farms was highest (>7% of agricultural land covered by an individual derogation) in 2019 (high derogation catchments) as reported in the 2018 Derogation Report.

Groundwater monitoring data are not available for the Crawfordsburn catchment which is not considered representative of Northern Ireland, because of its urban nature. The percentage of farm land in comparison to total catchment area in the Crawfordsburn catchment (23 %) is low in comparison to the other catchments: Strangford (51 %), Clanrye (56 %), Upper Bann (61 %) and Ballinderry (71 %). There are currently no groundwater monitoring stations within the Clanrye catchment.. The list of catchments with the highest percentage of derogated farms is updated annually and 2016 was the first time that the Clanrye catchment was considered a high derogation catchment.

In this report, comparisons of the mean annual average data for the period 2012-2015 (as reported in the Northern Ireland 2016 Nitrates Article 10 Report[4]) and the most recent annual average data for the current reporting year (2019) are presented. In each period, surface water data were only included where sufficient numbers of samples over the four years (2012-2015) and one year (2019) were available[5]. In the four-year (2012-2015) period all groundwater data were included. For the current reporting year 2019, all available groundwater data were included; consisting on average of three samples per monitoring site.

Presentation of the four-year data set (2012-2015) as reported in the Northern Ireland 2016 Nitrates Article 10 Report, provides continuity with other Nitrates Directive 91/676/EEC (ND) reporting requirements and it provides a clear indication of how the water quality is evolving since the Article 10 Report for the period 2012-2015. The NI Article 10 Report for the period 2016-2019 was completed in summer 2020 and data submitted to the Commission. Data will be summarised and presented for the period 2016-2019 in next year's Derogation report.

[4] Member States must report every four years to the Commission on the status of water quality in accordance with Article 10 of the Nitrates Directive (91/676/EEC). The 'Nitrates Directive Development Guidance Notes for Member States' issued in 2011, indicate that for the purposes of reporting, data may be averaged over more than one year. The UK 2016 Nitrates Article 10 Report was completed in July 2016 and data was summarised and presented for 2012-2015.

[5] Sufficient numbers of samples, for annual average, in the four-year period 2012-2015 were considered to be ≥20 samples and ≥10 for the current reporting year (2019).

The results of a single year analysis must be treated with caution; due to the relatively low numbers of samples involved and possible variability due to climatic influences (for the purposes of this report, adjustments have not been made for weather or other varying annual effects). When considering the following results it should, therefore, be remembered that variations in annual precipitation or in seasonal patterns of rainfall can increase nutrient run-off and thereby potential nutrient input to surface waters. Average monthly precipitation for Northern Ireland for 2012-2015 was 84.1 mm and for 2019 was higher at 91.5 mm (source: Met Office). In particular, in 2019 Northern Ireland neared record winter temperatures, 8 or 9 degrees higher than what is typical for February[6]. However, five named storms affected the UK during 2019[7]. Significant average monthly precipitation was noted in March (148.2 mm) and August (145.9 mm), both significantly above the 1971-2000 average.

A review of the surface freshwater monitoring programme was undertaken for the second cycle of the River Basin Management Plans (RBMPs). As a result, changes were implemented in 2015 through better targeting and by adopting a risk based approach to monitoring. In 2019, the average number of monthly samples analysed for nutrients was 503.

Groundwater quality in Northern Ireland is assessed in accordance with NIEA's groundwater monitoring programme through the collection of water samples from boreholes and springs that are mostly owned and operated by third parties. The public water supply provider in Northern Ireland (NI Water Ltd) does not currently utilise groundwater with the exception of Rathlin Island, a small island off the north coast of Northern Ireland. Hence, NIEA rely mostly on third party owned boreholes and the co-operation of land/property owners to continue sampling from their groundwater sources for the chemical/nutrient monitoring.

This means that the composition of the ground water monitoring network can change due to businesses closing or changing their groundwater usage and in addition, datasets for trend assessments are often small. The monitoring network consists mainly of industrial boreholes where groundwater is utilised for manufacturing or food/drinks production. A small number of springs or boreholes purpose-installed by NIEA, which are purged prior to sampling, are also monitored. The selection of groundwater monitoring sites to date has been based on a pressure-pathway assessment of the groundwater bodies and the availability of potential monitoring points.

[6] www.climatenorthernireland.org.uk/
[7] www.metoffice.gov.uk/binaries/content/assets/metofficegovuk/pdf/weather/learn-about/uk-past-events/summaries/uk_monthly_climate_summary_annual_2019.pdf

3.1 Nitrate concentrations in surface freshwater

In the period 2012-2015, NIEA monitored nitrate concentrations at 337 surface freshwater monitoring stations across Northern Ireland. The annual average nitrate concentration at these stations was 5.2 mg NO_3/l. In 2019, nitrate concentrations were monitored at 505 surface freshwater stations giving an annual average nitrate concentration of 6.3 mg NO_3/l. Table 2 shows annual average nitrate concentrations in surface freshwater across Northern Ireland in 2012-2015 and 2019. Table 3 shows the average nitrate concentrations in surface freshwater in the five high derogation catchments in 2012-2015 and 2019.

Table 2: Annual average nitrate concentrations (based on number and % of monitoring stations) of surface freshwater across Northern Ireland, 2012-2015 and 2019

Average nitrate concentration (mg NO_3/l)	2012-2015 (337 Stations)	2019 (505 Stations)
0-9.99	89.3% (301)	80.8% (408)
10-24.99	10.7% (36)	19.0% (96)
25-39.99	0	0.2% (1)
40-49.99	0	0
>50	0	0

Table 3: Annual average nitrate concentrations (based on number of monitoring stations) of surface freshwater in the high derogation catchments, 2012-2015 and 2019

Catchment		Average Nitrate (mg NO_3/L)				
		0-9.99	10-24.99	25-39.99	40-49.99	>50
Ballinderry Catchment	2012-15 (22 Sites)	81.8% (18)	18.2% (4)	0	0	0
	2019 (23 Sites)	69.6% (16)	30.4% (7)	0	0	0
Strangford Catchment	2012-15 (19 Sites)	63.2% (12)	36.8% (7)	0	0	0
	2019 (21 Sites)	38.1% (8)	61.9% (13)	0	0	0
Clanrye Catchment	2012-15 (9 Sites)	44.4% (4)	55.6% (5)	0	0	0
	2019 (10 Sites)	10% (1)	90% (9)	0	0	0
Crawfordsburn Catchment	2012-15 (1 Site)	100% (1)	0	0	0	0
	2019 (1 Site)	0	100% (1)	0	0	0
Upper Bann Catchment	2012-15 (12 Sites)	100% (12)	0	0	0	0
	2019 (24 Sites)	54.2% (13)	45.8% (11)	0	0	0

In 2012-2015, 63 of the monitored surface freshwater stations were located in the five catchments with the highest proportion of derogated farms and an annual average nitrate concentration of 7.8 mg NO_3/l was recorded. In 2019, 79 stations were monitored in these catchments with an annual average concentration of 10.6 mg NO_3/l.

Figure 5 shows the distribution of nitrate in surface freshwater across Northern Ireland and the high derogation catchments in 2019 (as reported in 2018 Derogation Report). Average nitrate concentrations in 2019 were generally low across Northern Ireland, with 99.8% of surface water stations below 25 mg NO_3/l.

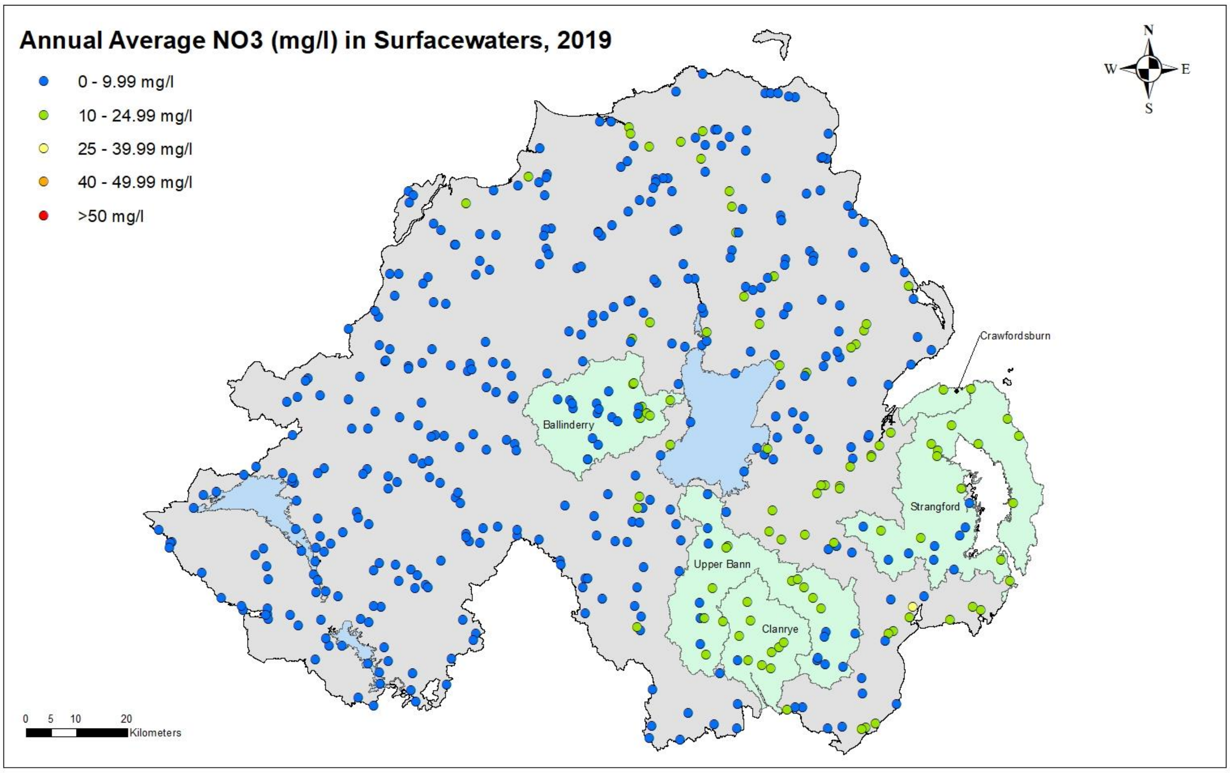


Figure 5: Distribution of annual average nitrate concentrations at surface freshwater stations in 2019

Nitrate concentration trends in Table 4 and Figure 6 indicate that the annual average nitrate concentrations in common surface freshwater stations across Northern Ireland were stable or decreasing at 58.1% of surface freshwater sites (including the five high derogation catchments) between the two reporting periods, 2012-2015 and 2019. This compares to 100% of sites in the 2016 report, 87% of sites in the 2017 report and 69% of sites in the 2018 report.

Increasing nitrate concentrations were reported in 41.8% of sites across Northern Ireland. 29.4% of these increases coincided with catchments with high proportion of derogated farms, especially Strangford, Upper Bann and Clanrye.

Table 4: Change in average nitrate concentrations (based on % and number of common monitoring stations) of surface freshwater across Northern Ireland and in the high derogation catchments, between 2012-2015 and 2019

Difference in average Nitrate concentration (mg NO_3/L) 2012-15 - 2019	% and number of common monitoring stations		
	Decrease[1]	**Stable**[2]	**Increase**[3]
Northern Ireland (325 Stations)	0.9% (3)	57.2% (186)	41.8% (136)
Ballinderry Catchment (23 Sites)	4.3% (1)	73.9% (17)	21.7% (5)
Strangford Catchment (18 Sites)	0	11.1% (2)	88.9% (16)
Clanrye Catchment (9 Sites)	0	0	100% (9)
Crawfordsburn Catchment (1 Site)	0	0	100% (1)
Upper Bann Catchment (12 Sites)	0	25.0% (3)	75.0% (9)

Difference is assessed by change in concentration – [1]Decrease ≤ -1 mg/l, [2]Stable -1 to +1 mg/l, [3]Increase ≥ +1 mg/l

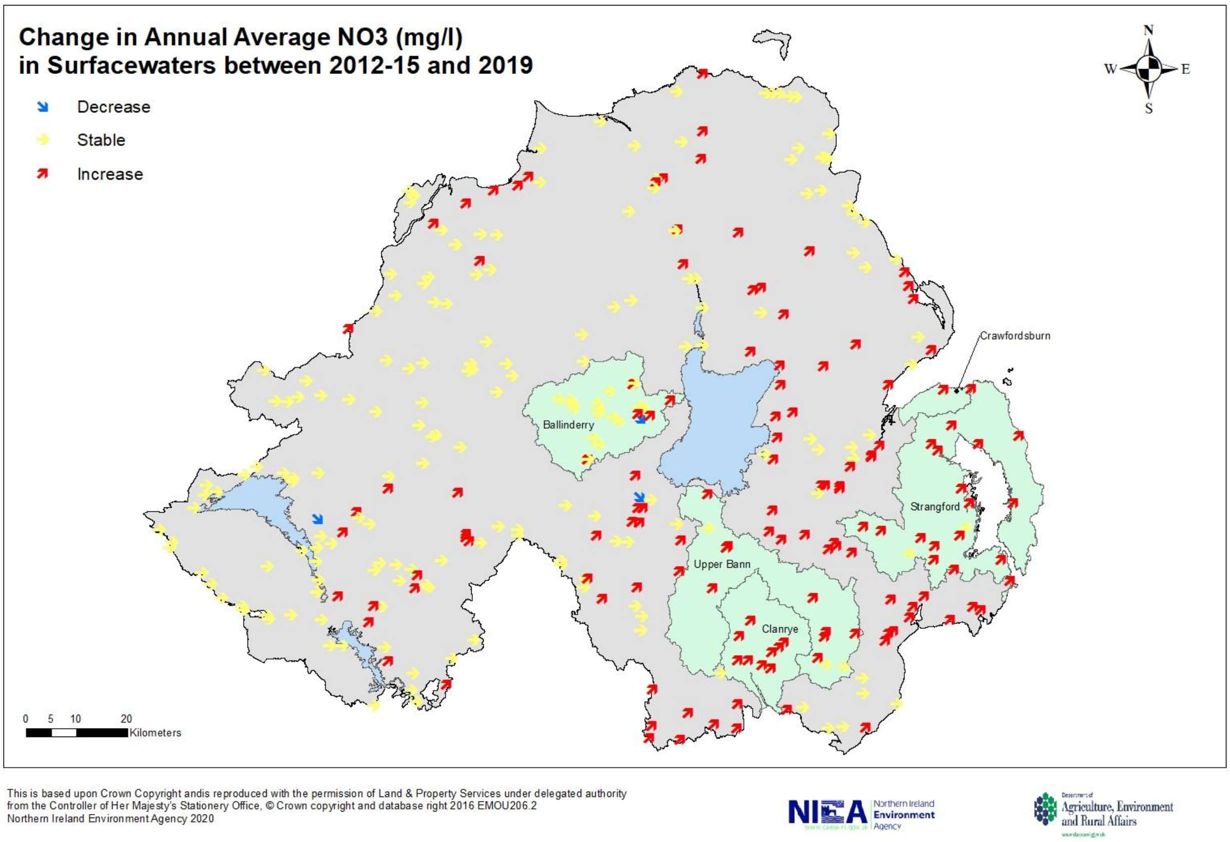


Figure 6: Change in annual average nitrate concentrations at surface freshwater stations between 2012-2015 and 2019

Stations across Northern Ireland showing increasing trends in nitrate will be subject to further data analysis, and targeted action. Further investigations and actions in these catchments will be implemented as part of targeted catchment projects under WFD in the NIEA River Basin District Groups (RBDs). This will include engagement with the sewerage regulator, home owners and farmers in the local areas, to follow up actions arising from reported pollution incidents and improve water protection.

3.2 Nitrate concentrations in groundwaters

In the period 2012 to 2015, NIEA monitored nitrate concentrations at 56 groundwater monitoring sites across Northern Ireland in which the average nitrate concentration was 6.26 mg NO_3/l. In 2019, nitrate concentrations were monitored at 54 groundwater sites across Northern Ireland giving an average concentration of 6.58mg NO_3/l. The difference in the number of groundwater monitoring sites is a result of NIEA's reliance on third parties to assist in providing monitoring points. Overall, two less boreholes were in use in the 2019 period than during the period from 2012 to 2015. Where such changes have occurred within the high derogation catchments they have not been found to influence the catchments status.

Table 5 shows the range of average nitrate concentrations in groundwater across Northern Ireland and in the high derogation catchments for 2012-2015 and 2019.

Table 5: Average nitrate concentrations (based on number of monitoring sites) of groundwater across Northern Ireland and in the high derogation catchments with the highest proportion of derogated farms, 2012-2015 and 2019.

Catchment	Groundwater body	Average nitrate concentration (mg NO_3/l)	0-24.99	25–39.99	40–50	>50
Northern Ireland		2012-2015 56 Sites	55	0	0	1
		2019 54 Sites	52	1	0	1
Strangford	Ards Peninsula	2012-2015 1 Sites	0	1	0	0
		2019 1Site	1	0	0	0
	Belfast East	2012-2015 5 Sites	4	0	0	1
		2019 4 Sites	3	0	0	1
Upper Bann	Aughnacloy	2012-2015 1 Site	1	0	0	0
		2019 1 Site	1	0	0	0
	Tandragee	2012-2015 2 Sites	2	0	0	0
		2019 1 Site	1	0	0	0
Ballinderry	Cookstown	2012-2015 5 Sites	4	1	0	0
		2019 4 Sites	3	1	0	0
	Moneymore	2012-2015 1 Sites	1	0	0	0
		2019 1 Site	1	0	0	0

In 2012-2015 55 of the 56 monitoring stations across Northern Ireland had nitrate concentrations in the 0-24.99 NO_3/l bracket. Therefore 1 of the 56 stations across Northern Ireland had an average nitrate concentration over 25 mg/l NO_3/l. In 2019, 52 out of 54 stations across Northern Ireland had nitrate concentrations in the 0-24.99 NO_3/l bracket. Therefore 2 of the 54 stations across Northern Ireland had nitrate concentrations over 25 mg/l NO_3/l. The two stations in the higher brackets (i.e. between 25->50mg/l NO_3/l and +50mg) are located in the high derogation catchment of Strangford.

In the Strangford catchment 6 stations were used to calculate an average concentration for 2019 of 19.94 mg/l NO_3/l. The average concentration for the Strangford catchment 2012-2015 was 21.3 mg/l NO_3/l. From the stations monitored 1 station in the Belfast East groundwater body has had a nitrate concentration greater than 50 mg/l NO_3/l in 2012-2015, 2018 and 2019. This is demonstrated in Figure 7. This station is located in a former nitrate vulnerable zone before Northern Ireland was designated total territory. The station was purposely installed in that location to monitor groundwater quality in this area of arable farming.

In the Ballinderry catchment 5 stations were used to calculate an average concentration for 2019 of 9.38 mg/l NO_3/l. The average concentration for 2012-2015 was 7.1 mg/l NO_3/l. From the 4 stations monitored in the Cookstown groundwater body all had nitrate concentrations less than the range of 25-39.99 mg/l NO_3/l (whereas in 2012-2015 and 2018 there was 1 station in this range). The Cookstown groundwater body had an average concentration of 11.27 NO_3/l for 2019, therefore indicating a general increase in the catchment average, despite the station which was in the higher range moving to the category below. This is demonstrated in Figure 7.

In the Upper Bann catchment 2 stations (one in each groundwater body) were used to calculate an average concentration for 2019 of 0.88 mg/l NO_3. The average concentration for Upper Bann in 2012-2015 was 2.00 mg/l NO_3/l.

The Lagan, Blackwater and Lower Bann catchments all have average concentrations in the 0-24.99 NO_3/l bracket. The Crawfordsburn, Clanrye and Bush are not recorded in Table 5 because there are currently no groundwater chemical monitoring stations in these catchments. This is principally because the NIEA are reliant on third parties for access to land and boreholes. NIEA continues to work towards expanding the groundwater monitoring network across Northern Ireland. The Bush catchment and Clanrye catchment are considered high priority areas in the development of this with the aim of locating monitoring points within these areas.

The trend assessments average of the groundwater bodies within each catchment are compared from the 2012-2015 and 2019 time periods and are presented in Table 6. The results are also presented spatially in Figure 8.

Nitrate concentration trends in groundwater across Northern Ireland indicate a decrease or stabilisation in Upper Bann, Lagan, Blackwater, Ballinderry and Lower Bann derogation catchments in 2019 compared to 2012-2015. The Strangford catchment shows a slight decrease in average nitrates concentration in 2019 when compared to 2012 to 2015 for the Belfast East groundwater body and a larger decrease in average nitrates concentration in 2019 when compared to 2012 to 2015 for the Ards Peninsula groundwater body. Cumulatively these results from the two groundwater bodies give a modest overall decrease in nitrate for the Strangford catchment since the 2012-2015 period.

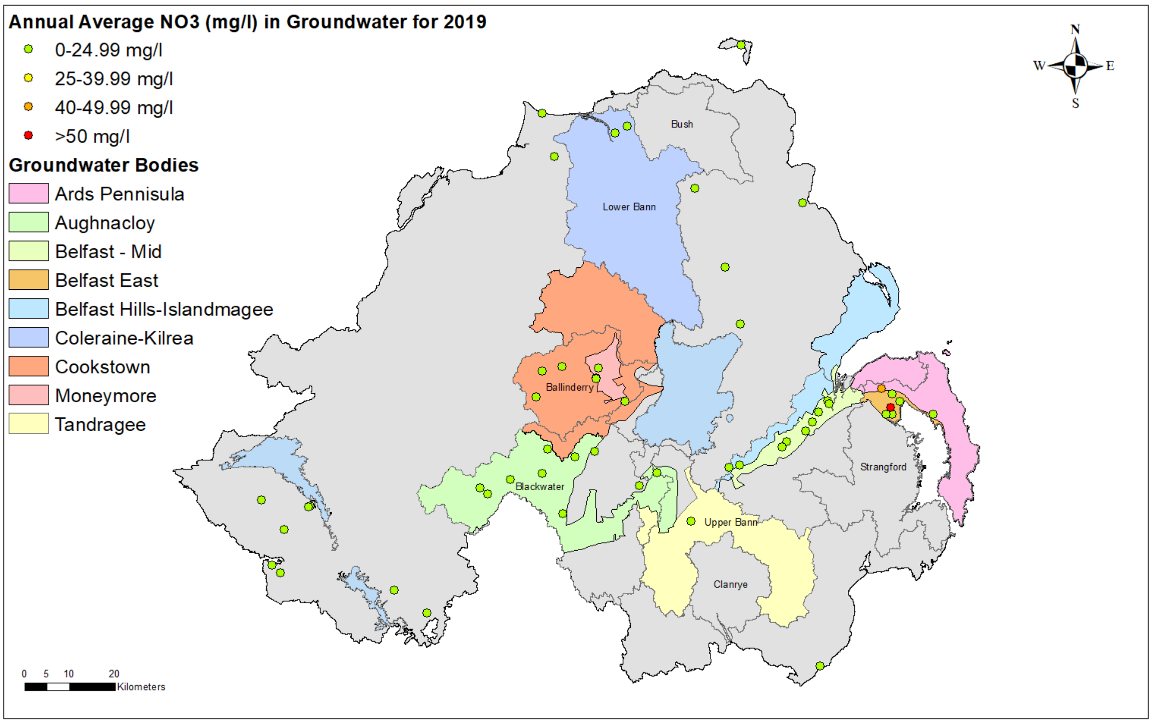


Figure 7: Distribution of annual average nitrate concentrations at groundwater stations in 2019. High Derogation Catchments are labelled and outlined with black line. Sampled groundwater bodies within these catchments are coloured according to legend.

Table 6: Change in average nitrate concentrations (based on averages per groundwater body in each catchment) across Northern Ireland and in the high derogation catchments, between 2011-2015 and 2019

Catchment	**Groundwater body**	**Difference in average nitrate concentration (mg NO_3/l)**	**< -5**	**-1 to -5**	**-1 to +1**	**+1 to +5**	**> +5**
Strangford	Ards Peninsula		•				
	Belfast East			•			
Upper Bann	Aughnacloy				•		
	Tandragee				•		
Ballinderry	Cookstown				•		
	Moneymore				•		

Difference is assessed by change in concentration – Strong Decrease ≤ -5 mg/l, Decrease >-5 to ≤ -1 mg/l, Stable >-1 to <+1 mg/l, Increase ≥ +1 to <5 mg/l, Strong Increase ≥ +5 mg/l

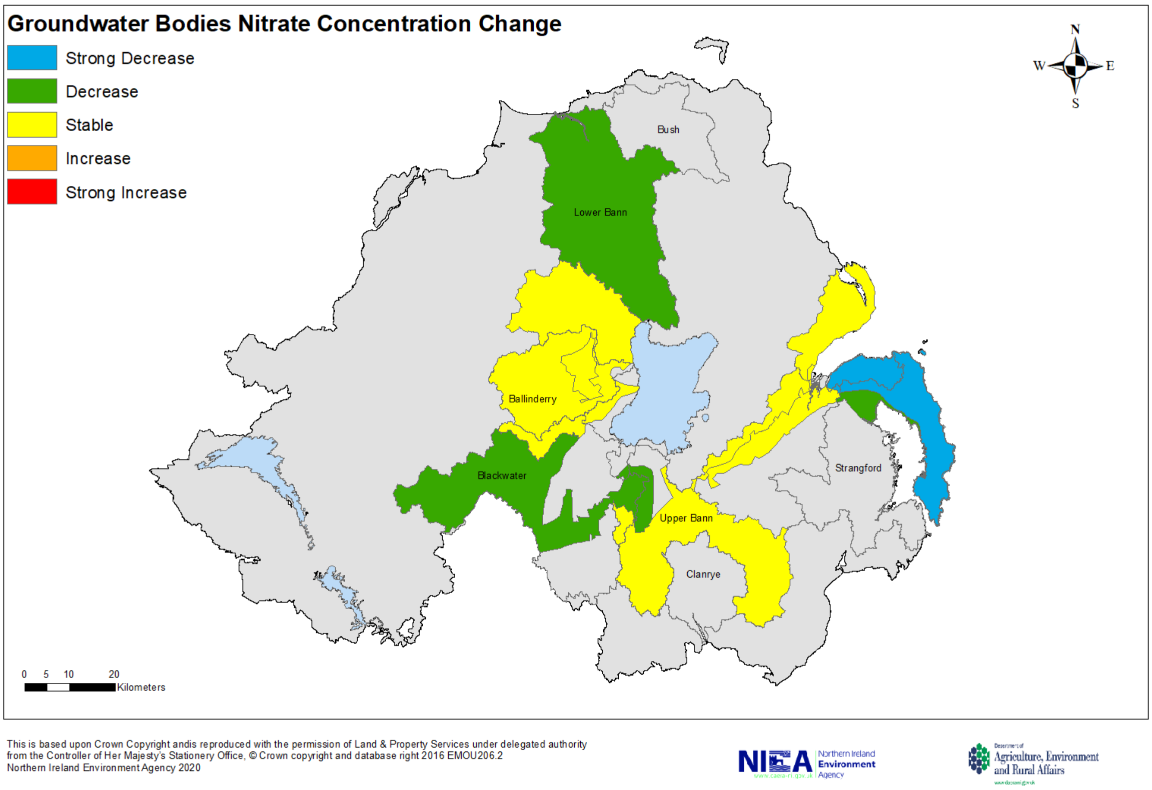


Figure 8: Changes in nitrate concentration averages per groundwater body in each catchment between the periods of 2012-2015 and 2019

3.3 Eutrophic indicators - phosphorus concentrations in rivers and streams

Since the adoption of the WFD, methodologies for assessment of eutrophication in rivers and lakes have changed. Historically, waters were assessed for trophic status using guidance issued by the UK authorities in 2002. Under the current WFD methodology, transposed in Northern Ireland as The Water Environment (Water Framework Directive) Regulations (Northern Ireland) 2017[8] and the Water Framework Directive (Classification, Priority Substances and Shellfish Waters) Regulations (Northern Ireland) (2015)[9], freshwater bodies are assessed for trophic status using WFD standards for both phosphorus (SRP) and biological indicators.

For the purposes of this report SRP is considered on its own and without the supporting biological parameters normally required to classify status. To be consistent in the approach and for temporal comparative purposes Northern Ireland has assessed data from 2012-2018 using the SRP standards calculator as set out in the 2015 Regulations noted above, to obtain a site specific WFD SRP Classification for each site. They have been derived using an approach to setting phosphorus standards that produces site specific estimates of natural phosphorus concentrations, taking account of a site's alkalinity and altitude (UKTAG, 2015).

[8] http://www.legislation.gov.uk/nisr/2017/81/pdfs/nisr_20170081_en.pdf
[9] http://www.legislation.gov.uk/nisr/2015/351/pdfs/nisr_20150351_en.pdf

The revised standards represent a major step forward in matching nutrient concentration to ecological change. The standards are more precautionary than previous SRP standards used in the first cycle of River Basin Management Plans as UKTAG found these to be insufficiently stringent, with High or Good status phosphorus classifications being produced for water bodies where there are clear ecological impacts of nutrient enrichment.

In the period 2012-2015, NIEA monitored annual soluble reactive phosphorus (SRP) concentrations at 391 surface freshwater river stations across Northern Ireland. The annual average SRP concentration at these sites was 64.8 µg SRP/l. In 2019, SRP concentrations were monitored at 471 surface freshwater river stations, giving an annual average SRP concentration of 75.1 µg SRP/l.

In 2012-2015, 67 of the monitored surface freshwater monitoring stations were located in the five catchments with the highest proportion of derogated farms and an average phosphorus concentration of 109.4 µg SRP/l was recorded. In 2019, 74 stations were monitored in these catchments with an annual average concentration of 112 µg SRP/l. Table 7 shows the WFD SRP status in rivers and streams across Northern Ireland for 2012-2015 and 2019.

Table 7: WFD SRP status (based on % and number of monitoring stations) of surface freshwater in rivers and streams across Northern Ireland, 2012–2015 and 2019

WFD SRP Class	Northern Ireland	
	2012-2015 (391 Stations)	2019 (471 Stations)
HIGH	127 (32.5%)	80 (17.0%)
GOOD	132 (33.8%)	184 (39.1%)
MODERATE	110 (28.1%)	170 (36.1%)
POOR	22 (5.6%)	37 (7.8%)
BAD	0	0

Results in Table 7 show that in the 2012-15 reporting period, 66.3% of river sites were classified as High or Good for SRP status. The remaining 33.7% of river sites had a WFD SRP classification of less than Good status and are considered to be at risk from eutrophication or eutrophic. Of these sites, 5.6% were classed as Poor status for SRP.

In 2019, 56.1% of river sites were classified as High or Good for SRP status. 43.9% of river sites had a WFD SRP classification of less than Good status. Of these, 7.8% were classified as Poor status for SRP, indicative of nutrient enrichment. No sites were classed as Bad status in either reporting period. Compared with the previous reporting period (2012-15), there was a decrease in the percentage of sites that were classed as High or Good. Figure 9 shows the distribution of WFD SRP status across Northern Ireland and the five derogation catchments in 2019.

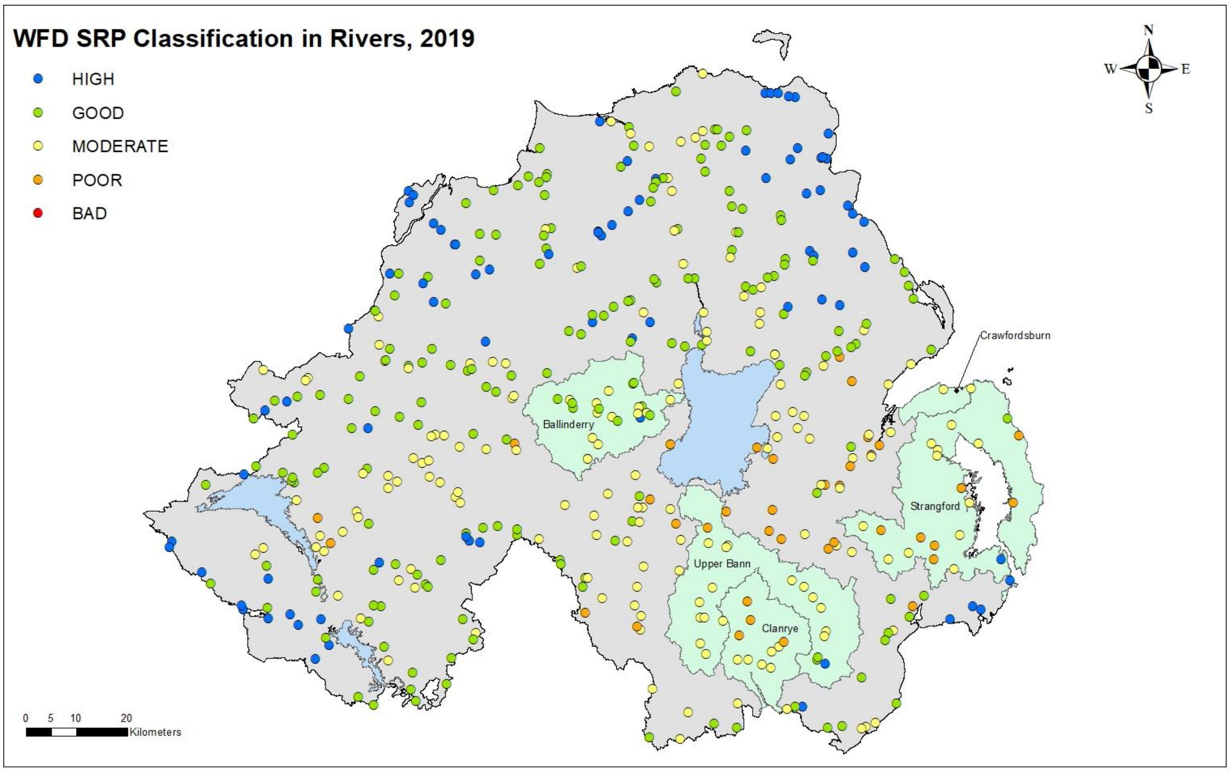


Figure 9: Distribution of WFD SRP status at surface freshwater stations in rivers and streams in 2019

Table 8 shows the WFD SRP status of surface freshwater monitoring stations in rivers and streams (based on common stations) in the high derogation catchments in 2012-2015 and 2019.

Average SRP concentrations at sites in the Ballinderry catchment ranged from 19 to 119 µg SRP/l in 2012-2015, with 65.2% of sites class as High or Good status and 34.8% classed as Moderate status. No sites were classed as Poor or Bad status. In 2019, average SRP concentrations ranged from 28 to 107 µg SRP/l, with 43.5% of sites classed as High or Good status and 56.5% classed as Moderate status. No sites were classed as Poor or Bad status.

Average concentrations at sites in the Strangford catchment ranged from 41 to 519 µg SRP/l in 2012-2015, with 5.9% of sites class as High status and 47.1% classed as Moderate and 47.1% classed as Poor status. No sites were classed as Good or Bad status. In 2019, average SRP concentrations ranged from 42 to 423 µg SRP/l, with 5.9% of sites classed as High status and 52.9% classed as Moderate and 41.2% classed as Poor status. No sites were classed as Good or Bad status.

Average SRP concentrations at sites in the Clanrye catchment ranged from 63 to 181 µg SRP/l in 2012-2015, with 88.9% of sites class as Moderate status and 11.1% classed Poor status. In 2019, average SRP concentrations ranged from 60 to 216 µg SRP/l, with 66.7%

of sites class as Moderate status and 33.3% classed Poor status. No sites were classed as High, Good or Bad status in either reporting period.

The average concentration at the single site in the Crawfordsburn catchment was 103 µg SRP/l in 2012-2015 and 104 µg SRP/l in 2019, with no change in Moderate status between reporting periods.

Average SRP concentrations at sites in the Upper Bann catchment ranged from 11 to 274 µg SRP/l in 2012-2015, with 23.5% of sites class as High or Good status and 76.5% classed as Moderate or Poor status. In 2019, average SRP concentrations ranged from 10 to 270 µg SRP/l, with 17.7% of sites classed as High or Good status and 82.3% classed as Moderate or Poor status. No sites were classed as Bad status either reporting period.

Table 8: WFD SRP status (based on number of common stations) of surface freshwater in rivers and streams in the high derogation catchments, 2012-2015 and 2019

WFD SRP Class		HIGH	GOOD	MODERATE	POOR	BAD
Ballinderry Catchment (23 Sites)	2012-15	4.3% (1)	60.9% (14)	34.8% (8)	0	0
	2019	4.3% (1)	39.1% (9)	56.5% (13)	0	0
Strangford Catchment (17 Sites)	2012-15	5.9% (1)	0	47.1% (8)	47.1% (8)	0
	2019	5.9% (1)	0	52.9% (9)	41.2% (7)	0
Clanrye Catchment (9 Sites)	2012-15	0	0	88.9% (8)	11.1% (1)	0
	2019	0	0	66.7% (6)	33.3% (3)	0
Crawfordsburn Catchment (1 Site)	2012-15	0	0	100% (1)	0	0
	2019	0	0	100% (1)	0	0
Upper Bann Catchment (17 Sites)	2012-15	5.9% (1)	17.6% (3)	70.6% (12)	5.9% (1)	0
	2019	5.9% (1)	11.8% (2)	76.5% (13)	5.9% (1)	0

The change in annual average SRP concentrations were historically reported in previous Derogation Reports according to Nitrates Directive guidance circulated in 2011. This identifies if a change is significant at ±50 µg/l, but this is a relatively coarse assessment. By also reporting the change in WFD SRP status using site specific standards, these will be more responsive to changes occurring and highlight any deterioration. This may be due to natural fluctuations as relatively small changes in SRP concentration can result in a change in class.

Trends in annual average SRP concentration shown in Table 9 and Figure 10 indicate a decline or stabilisation in SRP levels at 97.8% of common surface freshwater monitoring stations in rivers and streams between 2012-2015 and 2019 across Northern Ireland when assessed according to Nitrates Directive guidance. The criteria used to report change is ±50 µg/l, which is a relatively high threshold.

Table 9: Change in average soluble reactive phosphorus concentrations (based on % and number of common monitoring stations) of surface freshwater in rivers and streams across Northern Ireland and in the high derogation catchments, between 2012-2015 and 2019

Difference in average SRP concentration (µg NO_3/L) 2012-15 - 2018	**Decrease[1]**	**Stable[2]**	**Increase[3]**
Northern Ireland (358 Stations)	1.7% (6)	96.1% (344)	2.2% (8)
Ballinderry Catchment (23 Stations)	0	100% (23)	0
Strangford Catchment (17 Stations)	23.5% (4)	76.5% (13)	0
Clanrye Catchment (9 Stations)	0	88.9% (8)	11.1% (1)
Crawfordsburn Catchment (1 Stations)	0	100% (1)	0
Upper Bann Catchment (17 Stations)	5.9% (1)	94.1% (16)	0

Difference is assessed by change in concentration – [1]Decrease ≤ -50 µg/l, [2]Stable > -50 to < +50 µg/l, [3]Increase ≥ +50 µg/l

Increasing phosphorus levels leading to a deterioration in WFD SRP class occurred at 28.5% of sites across Northern Ireland (Table 10 and Figure 11). Of these increases, 8.7% occurred within catchments with high proportion of derogated farms in the Ballinderry, Clanrye, Strangford and Upper Bann.

The changes in summary are:

- 27.4% (98 sites) deteriorated by one class for WFD SRP status;
- Four sites (1.1%) deteriorated by 2 classes between the two reporting periods. The Owenkillew River at Drumlea, Coneyglen Burn at Coneyglen Bridge and the Owenalena River at Owenalena Bridge all deteriorated from High to Moderate between the reporting periods. The Salry River at Salry deteriorated from Good to Poor status between the reporting periods
- 68.7% (246 sites) remained stable in WFD SRP status; and

- 2.8% (10 sites) exhibited an improvement in class between the two reporting periods. 0.3% (1 site - Hillsborough Park Lake Stream at Gowdy's) improved from Poor to Good status between the two reporting periods.

A number of these changes occurred within catchments with high numbers of derogated farms. The single site in the Crawfordsburn catchment remained stable with Moderate WFD SRP status between the two reporting periods. 82.4% (14 sites) remained stable in the Strangford catchment whilst 5.9 % (1 site) deteriorated by one class for WFD SRP status. 77.8% (7 sites) remained stable in the Clanrye catchment, whilst 22.2% (2 sites) deteriorated by 1 class. 94.1% (16 sites) remained stable in the Upper Bann catchment whilst 5.9% (1 sites) deteriorated by 1 class for WFD SRP status. 78.3% (18 sites) remained stable in the Ballinderry catchment whilst 21.7% (5 sites) deteriorated by one class for WFD SRP status.

As previously highlighted, these results should be treated with a degree of caution as natural variation in nutrient concentration is expected year to year due to seasonal and climatic changes.

Table 10: Change in WFD SRP classification (based on % and number of common monitoring stations) of surface freshwater in rivers and streams across Northern Ireland and in the high derogation catchments, between 2012-2015 and 2019

WFD SRP Classification	Strong Decrease[1]	Weak Decrease[2]	Stable[3]	Weak Increase[4]	Strong Increase[5]
Northern Ireland (358 Stations)	0.3% (1)	2.5% (9)	68.7% (246)	27.4% (98)	1.1% (4)
Ballinderry Catchment (23 Stations)	0	0	78.3% (18)	21.7% (5)	0
Strangford Catchment (17 Stations)	0	11.8% (2)	82.4% (14)	5.9% (1)	0
Clanrye Catchment (9 Stations)	0	0	77.8% (7)	22.2% (2)	0
Crawfordsburn Catchment (1 Station)	0	0	100% (1)	0	0
Upper Bann Catchment (17 Stations)	0	0	94.1% (16)	5.9% (1)	0

[1] *Strong Decrease = ≥2 improvements in class*
[2] *Weak Decrease = 1 improvement in class*
[3] *Stable = No change in class*
[4] *Weak Increase = 1 deterioration in class*
[5] *Strong Increase = ≥2 deteriorations in class*

All monitoring stations showing higher concentrations of SRP or a decline in WFD status for SRP will be subject to further data analysis to establish if there are any specific factors, such as geographic area or season that may suggest a cause for the change. This will be followed by investigations and actions as part of the relevant targeted catchment projects under WFD for each of the NIEA RBDs where the changes are most significant. This will include engagement with the sewerage regulator, home owners and farmers in the local areas, to follow up actions arising from reported pollution incidents and improve water protection.

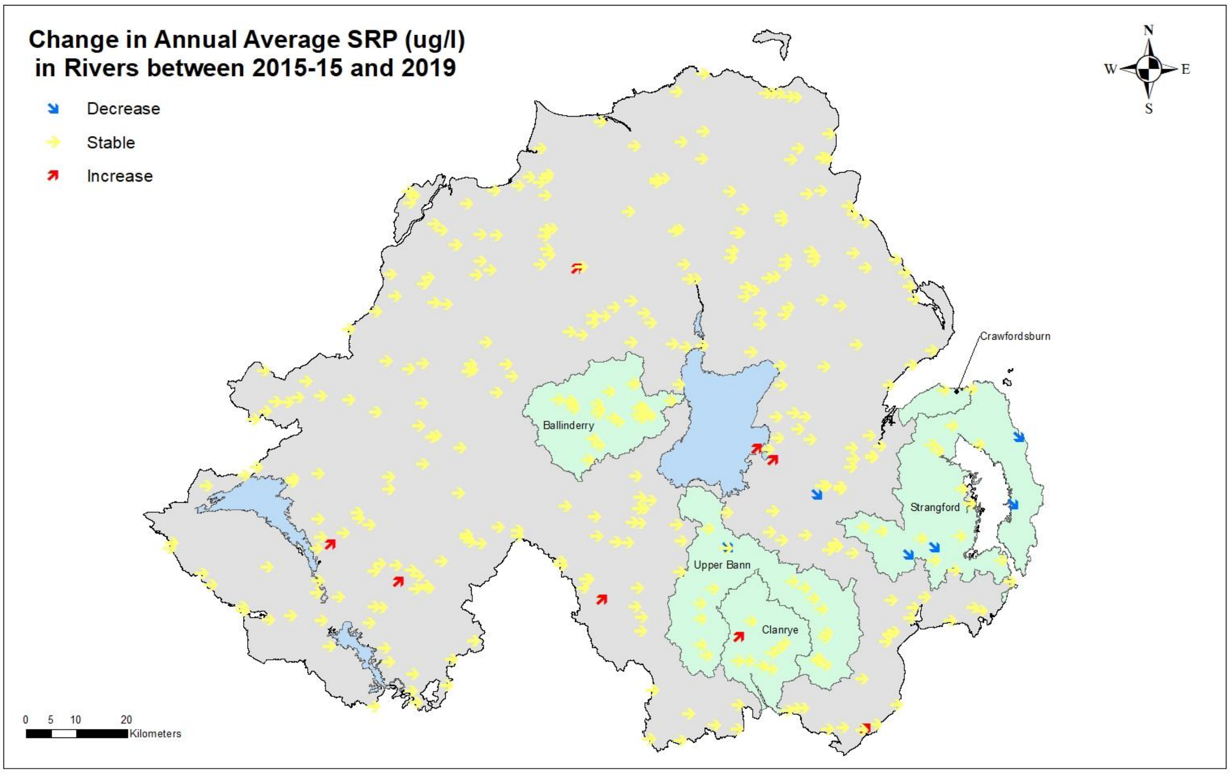


Figure 10: Change in SRP concentrations in surface water between 2012-2015 and 2019 when assessed according to Nitrates Directive guidance.

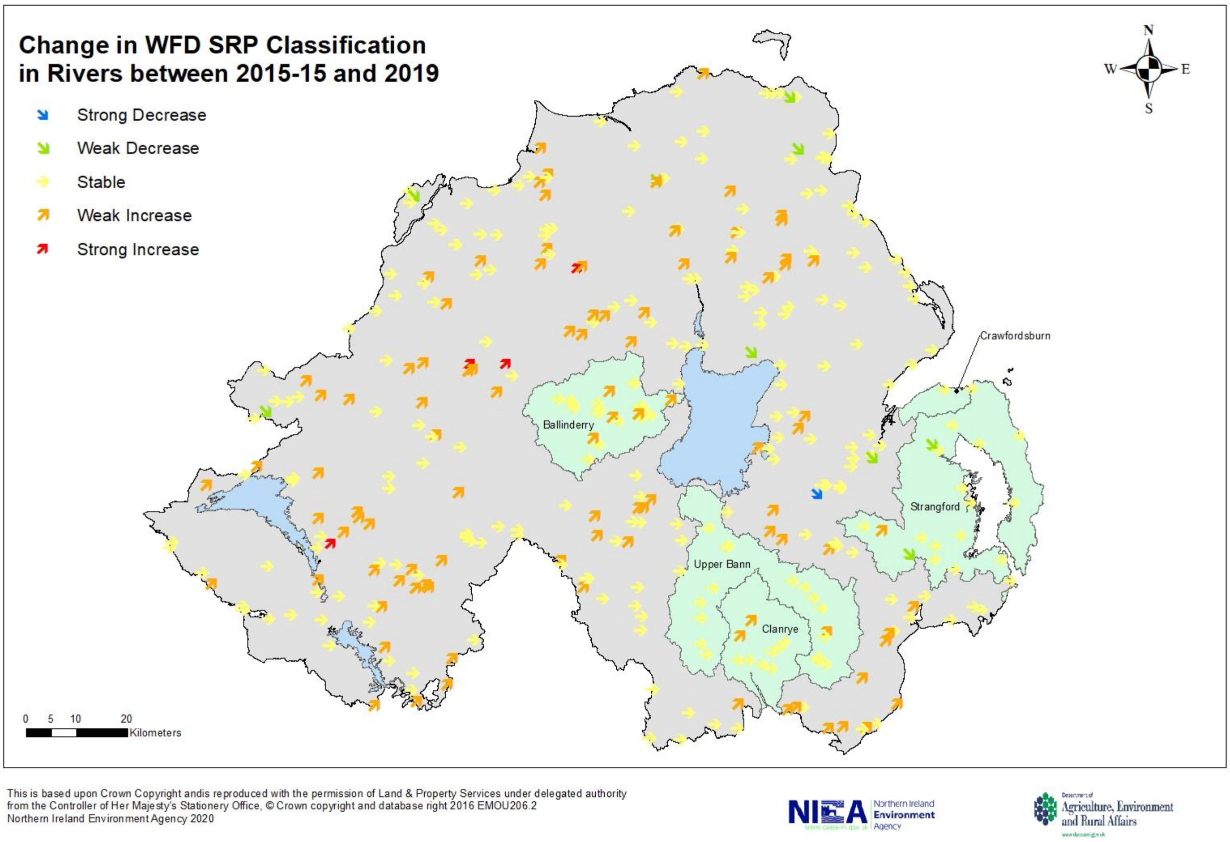


Figure 11: Change in WFD SRP classification in river monitoring sites between 2012-2015 and 2019

In this report, both the assessments using Nitrates Directive and WFD criteria show deterioration as indicated by the percentage of sites that are exhibiting increasing SRP levels. This is a serious cause for concern. Therefore, DAERA has included an SRP indicator for Water Quality in the draft Programme for Government (PfG). The SRP indicator used is the annual average SRP (µg/L) for 93 surveillance river sites and is not a WFD status assessment. For PfG it is required that a threshold for change is set. The criteria used to report changes for this indicator is ±0.01mg/l (against the baseline year of 2015). This is published annually in the NI Environmental Statistics Report[10]. The latest update in 2020 identified an increase of 0.004mg/l.

3.4 Eutrophic indicators - phosphorus concentrations in lakes

For the purposes of this report total phosphorus (TP) is considered on its own as a eutrophication indicator for lakes and without the supporting data on chlorophyll-α and biology normally required to classify under the WFD.

In the WFD classification period 2012-2014, NIEA monitored annual TP concentrations at 21 WFD lake and reservoir monitoring stations (Lower Lough Erne is divided into two water bodies) across Northern Ireland, with a surface area greater than 50ha (known as

[10] https://www.daera-ni.gov.uk/publications/northern-ireland-environmental-statistics-report-2020

surveillance lakes). In 2019, the same 21 lake and reservoir monitoring stations were monitored. The annual average TP concentration (based on geometric mean) for the 21 common surveillance stations was 65 µg TP/l for the period 2012-2014 and 83 µg TP/l for 2019. Although it will not be discussed further in this report, more recent detailed analysis undertaken for classification under the WFD[11] has shown that TP levels in all lakes have increased in the 3 year period 2017-2019.

For the previous baseline for lakes (2012-2014), 9 out of 21 lakes were classed as High or Good whilst 12 lakes were assessed as Moderate or worse. Table 11 and Figure 12 show that in 2019, four lakes (Fea, Lower Lough MacNean, Scolban and Silent Valley) were classed as Good WFD status whilst 17 were classed as Moderate, Poor or Bad WFD status, indicative of nutrient enrichment. Of these, three lakes (Lough Neagh, Portmore Lough and Stoneyford) were classed as Bad WFD TP status. No lake water bodies were classed as High status in 2019.

Figure 13 shows the changes in WFD TP status. One lake (Clea Lakes) showed an improvement from Poor to Moderate status between the two reporting periods (2012-14 and 2019) and 10 lakes remained stable for WFD TP status. Overall 10 lakes deteriorated in WFD TP class as set out below:

Eight lakes exhibited deterioration by one class in TP status between the two reporting periods. Fea, Scolban and Silent Valley deteriorated from High to Good TP class. Lower Lough Erne (Kesh), Upper MacNean and Melvin deteriorated from Good to Moderate TP class. Lower Lough Erne (Devenish) deteriorated from Moderate to Poor TP class. Stoneyford Lough deteriorated from Poor to Bad TP status.

Two lake water bodies exhibited deterioration by 2 classes in TP status between the two reporting periods. Castlehume Lough and Spelga, deteriorated from High to Moderate TP status.

For lakes within catchments with high numbers of derogated farms, the following occurred. Clea Lakes is in the Strangford derogated catchment and showed improvement from Poor to Moderate between the two reporting periods. Three lakes are in the Upper Bann derogated catchment. Lough Gullion was classed as Poor in both reporting periods, Lough Island Reavy was classed as Moderate in both reporting periods and Spelga Dam deteriorated from High to Moderate status between reporting periods. Lough Beg is in the Lower Bann derogated catchment and was classed as Poor in both reporting periods. All lakes exhibiting eutrophic conditions will be subject to further investigations and actions as part of the relevant RBD programme of targeted catchment projects under WFD.

[11] https://www.daera-ni.gov.uk/publications/northern-ireland-water-framework-directive-statistics-lake-quality-update-2020

Table 11: WFD status based on average TP concentrations (based on number of common monitoring stations) of WFD surveillance lakes and reservoirs across Northern Ireland, 2012-2014 and 2019

WFD TP Class	2012-2014 (21 Stations)	2019 (21 Stations)
High	5	0
Good	4	4
Moderate	3	8
Poor	7	6
Bad	2	3

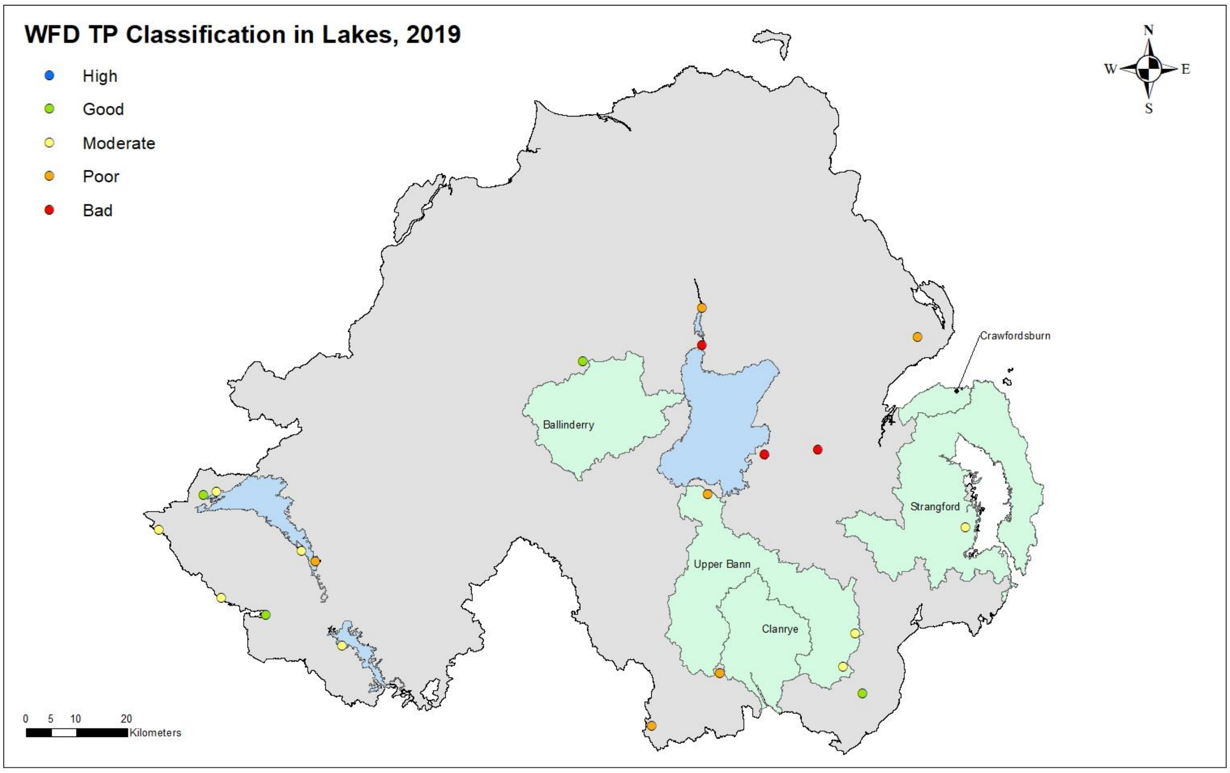


Figure 12: Distribution of WFD TP status in surveillance lakes and reservoirs in 2019

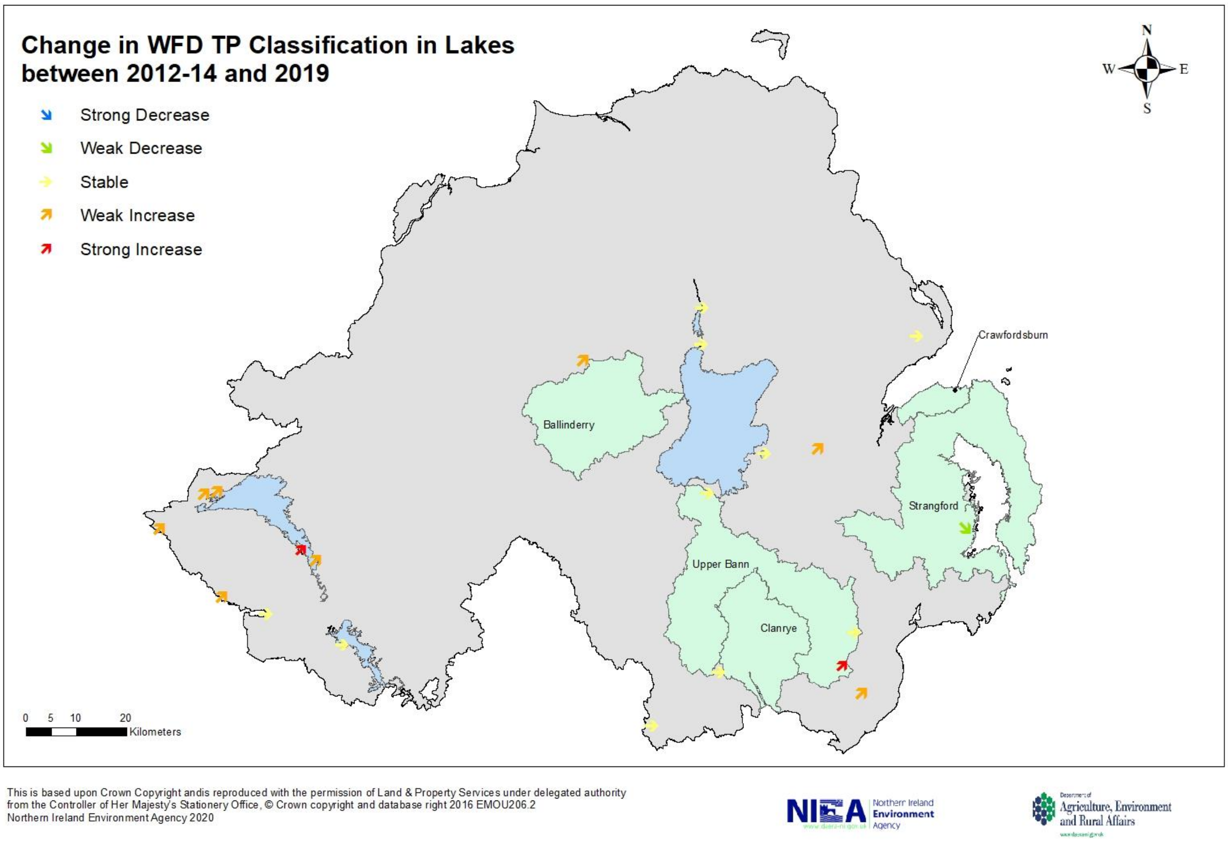


Figure 13: Change in WFD TP classification in surveillance lakes and reservoirs between 2012-2014 and 2019

4. SOIL MONITORING

To meet the requirements of Article 8(2) of the 2019 Derogation Decision, a detailed monitoring programme has been established. It will provide relevant soil P data for modelling P losses from derogated and non-derogated farms on the main soil types of Northern Ireland and information on farming practices etc. on derogated and non-derogated farms. A pair of sub-catchments has been identified in the Upper Bann River Catchment (Figure 14). One sub-catchment has a significant proportion of derogated farmland (120 out of 329 fields are on derogated farms) and the other has no derogated farmland (Figure 15). In accordance with Article 8(2) of the 2019 Derogation Decision, the most important soil profile types in Northern Ireland, i.e. Gleys (57%), and also the most important Hydrology of Soil Types (HOST) classes (17-24) particularly 24, i.e. soils developed on slowly permeable material (54%), are well represented within this pair of sub-catchments. The requirement to monitor soils on farms with *"levels of intensity and fertilisation practices"* typical for Northern Ireland has also been fulfilled by having two contrasting small sub-catchments with either some or no derogated farmland.

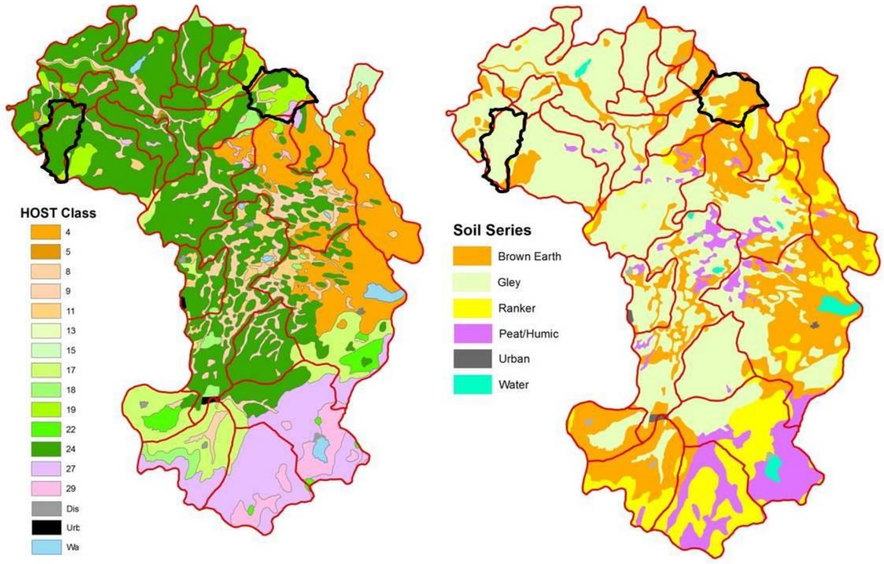


Figure 14: Maps of the Upper Bann River Catchment indicating soil types and Hydrology of Soil Types (HOST) classifications present in selected pairs of derogated and non-derogated sub-catchments.

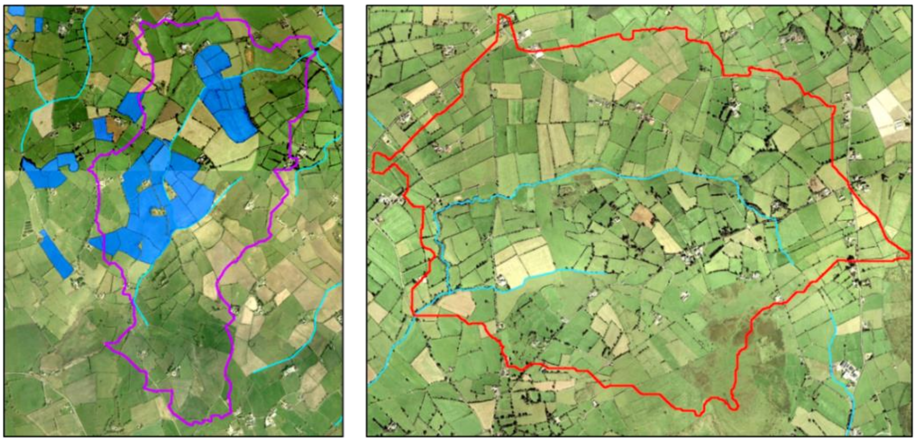


Figure 15: 'Derogated' (UB03) and Non-derogated (UB15) sub-catchments in the Upper Bann (UB) catchment with derogated farmland coloured blue.

Soil sampling in both catchments was undertaken during November-February 2016/2017. In total 945 fields were identified for sampling (including land outside the catchment boundary) of which 884 were sampled.

A further significant contribution to the knowledge base on soil fertility and farming practice in Northern Ireland was made by work undertaken as part of the EU Exceptional Adjustment Aid Soil Sampling and Analysis Scheme. In an Open Scheme accessible to farmers across Northern Ireland, 12,218 fields on 522 farms were soil sampled. An additional Catchment Scheme, which focussed on the Upper Bann catchment area sampled 513 farms (7,772 fields) across 11 sub-catchments that have biological and chemical water quality monitoring programmes in place. This Soil Sampling and Analysis Scheme was completed on 1st April 2018 and analysis is now underway to interrogate relationships with farm scale, derogation and landscape and climatic factors across the Upper Bann catchment and Northern Ireland.

4.1 Nitrogen and Phosphorus concentrations in soil water under derogated and non-derogated conditions

Article 8(2) of the 2019 Derogation Decision requires assessments to be made of N and P concentrations in soil water, to facilitate model-based estimates of nitrate and P losses from farms benefiting from derogation. However, these assessments are not considered to be appropriate in grassland situations in Northern Ireland for the following reasons.

The procedures for measuring N (*nitrate and ammonium*) and P (*SRP and total P*) concentrations in soil water (*solution*) are both difficult and problematic necessitating either vacuum plate extraction of soil solutions from individual intact soil cores, or the installation of ceramic suction cups in soil profiles to collect soil water/solution samples in situ. In commercial farming situations the ceramic cup apparatus can easily be damaged by farm machinery or livestock.

Furthermore, obtaining representative samples of soil solution is difficult because of preferential flow pathways which form along root channels or crevices allowing some downward flowing water to bypass the samplers (Ryan *et al*., 2006). An additional difficulty is the fact that the soil water samples collected by either technique are point-specific, since they are taken from single points within a field. Consequently, to allow for the high degree of spatial heterogeneity in N and P concentrations across grassland fields (Cuttle *et al*., 2001; McCormick *et al*, 2009), scores of points would have to be sampled, and this would simply not be cost-effective or practicable.

Temporal heterogeneity is also a problem, as nutrient levels in soil solution are subject to appreciable short-term fluctuations, owing to rainfall dilution etc. (Magid & Neilsen, 1992). Assessments of N and P concentrations in soil water therefore, do not provide a basis for predicting mean annual N and P losses to water at field and farm scales on grassland farms in Northern Ireland.

The factor most responsible for 'poor' water quality in Northern Ireland is P mobilisation from farmland into freshwater ecosystems and the resultant upsurge in algal growth in this P-limited rather than N-limited environment (Parr & Smith, 1976; Gibson & Stevens, 1979).

Consequently, to assess the impact of derogation on water quality, primary emphasis will be placed on quantifying the risk of P loss, rather than N loss, from farmland. In this regard, researchers in New Zealand have demonstrated that soil Olsen-P, in fields receiving nutrient inputs, is significantly correlated with both dissolved reactive phosphorus (DRP) and total P concentrations in overland flow from grassland sites on a broad range of soil types (McDowell *et al*., 2003).

Importantly, this soil parameter (*Olsen-P*) can be measured on bulked subsamples of soil (*0-75 mm depth*) easily collected from multiple locations across whole fields, as opposed to the single point locations associated with measurements of soil water P concentrations, thus minimising problems owing to spatial heterogeneity in soil P, as noted above. Moreover, Olsen-P assessments appear to be temporally quite stable (Shi *et al*., 2002).

In addition, the bulked soil samples could also be analysed for $CaCl_2$- P, which can provide a proxy estimate of DRP concentration in soil sub-surface flow (McDowell *et al*., 2003). These complementary soil P assessments, together with information and data on soil hydrology and connectivity, and on farm nutrient management practices, will then be used to model and compare P losses from derogated and non-derogated farms.

N is rarely limiting to algal growth in freshwater bodies in Northern Ireland (Parr & Smith, 1976; Gibson & Stevens, 1979). Mean nitrate concentrations in surface and ground waters are generally low and in almost all cases are well below the EU maximum admissible limit for drinking water (Northern Ireland NAP Review Report 2014). Modelling N losses from derogated farmland is still important to ensure that derogation measures are effective in preventing any deterioration in water quality attributable to N losses linked to farming.

As outlined above, soil N assessments are both problematic and poorly related to N loss by leaching or runoff from grassland. Therefore, information and data on soil type, soil hydrology and connectivity, and details of farming practices will be used to model and compare N losses from derogated and non-derogated farms.

In summary, soil Olsen-P and $CaCl_2$-P concentrations are being monitored in soils instead of P concentrations in soil water. In the absence of suitable soil N metrics, model estimates of nitrate loss from farms will be based primarily on soil typology and hydrology plus local climatic and farm management information.

Results from the first phase of soil sampling in the catchments were used to map soil chemistry within both sub-catchments (Figure 16). Although 884 fields in total were sampled, the total numbers of sampled fields within the catchment boundaries are 169 in the derogated and 324 in the non-derogated catchments, and it is these (n=493) which are the focus of this analysis. Field characteristics differ between catchments with larger field sizes in the derogated catchment (41 % > 2 ha in area) compared to the non-derogated catchment (9 % > 2 ha in area). This may reflect more intensive agriculture which has, over time, led to enlargement and merging of smaller fields in the derogated catchment, but also the difference in elevation and topography between catchments (Derogated Elevation Range: 80 - 170m; Non-Derogated Elevation Range: 125 – 310 m).

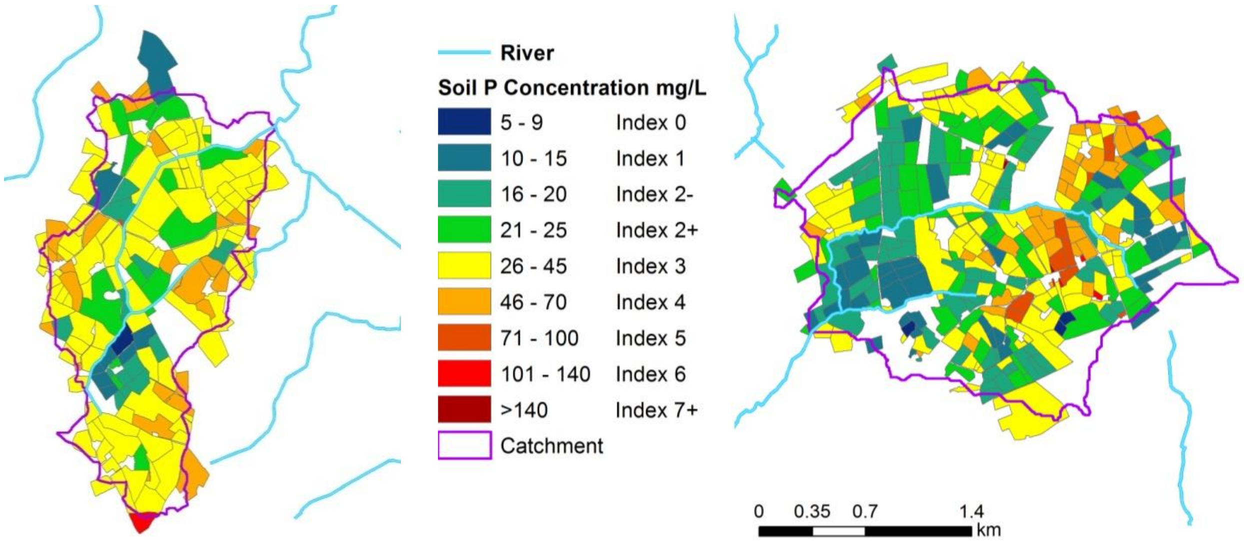


Figure 16: Soil P concentrations for the 'Derogated' (left) and 'Non-derogated' (right) sub-catchments in the Upper Bann catchment. Blank areas within the catchment boundaries were not included in the sampling.

The distribution of soil P between catchments differs considerably (Figure 17). The majority of fields, 79%, in the derogated catchment are at index 3 or above, compared to 43% of fields in the non-derogated catchment. The number of fields with excessively high soil P is greater in the non-derogated catchment with 4.9% of fields (n=16) in excess of Index 4, compared to 1.2% of fields (n=2) in the derogated catchment.

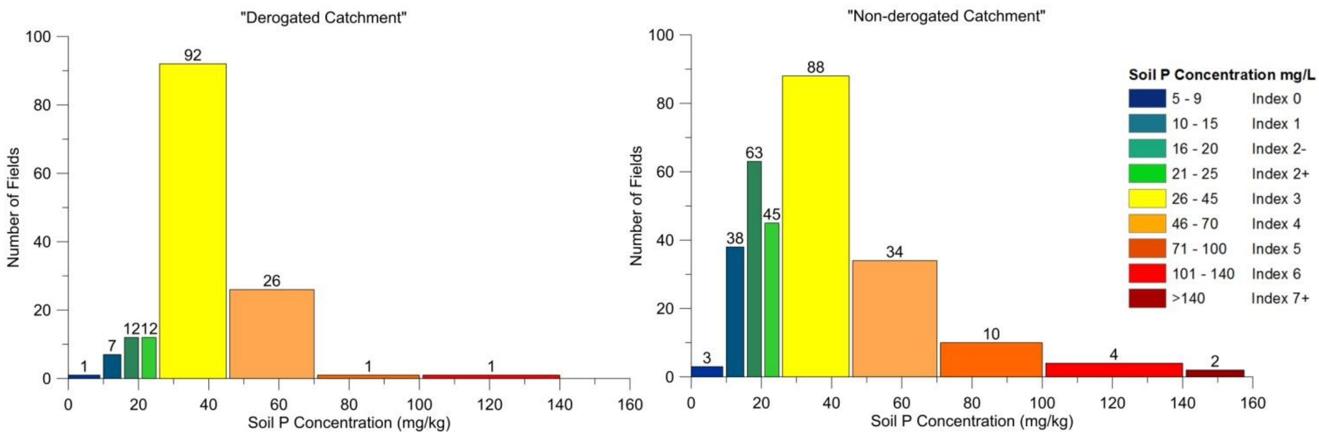


Figure 17: Histograms of P Index category for the 'Derogated' (n=169 fields) and 'Non-derogated' (n=324 fields) sub-catchments in the Upper Bann (UB) catchment.

The higher numbers of highly P-enriched fields (Index 5, 6 &7) in the non-derogated catchment (c.f. the derogated catchment), especially in proximity to farmyards, may indicate that ease of manure transport to particular fields is a stronger determinant of nutrient loading. The more extensive nature of agriculture in the non-derogated catchment is reflected however, by the proportion of fields lower than Index 3 (45.9%) compared to the equivalent proportion in the derogated catchment (18.9%). Furthermore, the average farm-gate P balance for a representative selection of farms within the derogated sub-catchment was only -1 kg P/ha (*primarily as a result of manure-P export from derogated farms*), c.f. 8 kg P/ha for a representative selection of farms within the non-derogated sub-catchment. In other words, while derogated farmland has currently the greatest proportion of fields with soil P indices > 2^+, corrective action is underway which has immediately reduced manure-P pressure on farmland (via manure export), and which in time should help to bring soil P on all P-enriched land (P index > 2^+), into the optimum Index 2^+ range.

The data from the Catchment Scheme within the EU EAA SSAS, has added an additional 7,772 fields across 11 sub-catchments, and allowed observations to be validated at a larger scale and across a more diverse agricultural landscape (these findings are presented in part in Section 7.1).

References

Cuttle, S.P., Scurlock, R.V. & Davies, B.M.S. (2001). Comparison of fertilizer strategies for reducing nitrate leaching from grazed grassland, with particular reference to the contribution from urine patches. *Journal of Agricultural Science*, **136**: 221-230.

Gibson, C.E. & Stevens, R.J. (1979). Changes in Phytoplankton Physiology and Morphology in response to dissolved nutrients in Lough Neagh, Northern Ireland. *Freshwater Biology*, **9**: 105-109.

Magid, J. & Nielsen, N.E. (1992). Seasonal variation in organic and inorganic phosphorus fractions of temperate-climate sandy soils. *Plant and Soil*, **144**: 155-165.

McCormick, S., Jordan, C. & Bailey, J.S. (2009). Within and between-field spatial variability in soil phosphorus levels within a permanent grassland area. *Precision Agriculture*, **10**: 262-276.

McDowell, R.W., Monaghan, R.M. & Morton, J. (2003). Soil phosphorus concentrations to minimise potential P loss to surface waters in Southland. *New Zealand Journal of Agricultural Research*, **46**: 239-253.

NAP Review Report (2014). Review of 2011-2014 Action Programme for the Nitrates Directive in Northern Ireland and associated regulations. Accessed at: www.daera-ni.gov.uk/sites/default/files/publications/daera/ni-nap-review-report-2014.PDF

Parr, M.P. & Smith, R.V. (1976). The identification of phosphorus as a growth limiting nutrient in Lough Neagh using bioassays. *Water Research*, **10**: 1151-1154.

Ryan, M., Brophy, C., Connolly, J., McNamara, K. & Carton, O.T. (2006). Monitoring of nitrogen leaching on a dairy farm during four drainage seasons. *Irish Journal of Agricultural and Food Research*, **45**: 115-134 Shi, Z., Wang, K., Bailey, J.S., Jordan, C. & Higgins, A.J. (2002). Temporal Changes in the Spatial Distribution of some Soil Properties on a Temperate Grassland Site. *Soil Use and Management*, **18**:353-362.

4.2 Mineral nitrogen in soil profile under derogated and non-derogated conditions

As indicated above, soil sampling and analysis commenced in the autumn of 2016, as part of the sampling programme. However, measurements of mineral nitrogen (N) are not being made, but rather estimates of nitrate loss from farms will be modelled based on soil typology and hydrology plus local climatic and farm management information.

Assessments of mineral N in soil profiles, which involve the collection of deep soil cores (*up to 900 mm depth*), each of which is separately analysed for mineral N, and the periodic soil

N assessments to be made on derogated farms, as specified in Article 5(6) of the 2019 Derogation Decision, are not appropriate for predicting N losses to water from grassland in Northern Ireland for the following reasons:-

a) They provide only a snap-shot in time of the amounts of mineral N present in soil, and it is known that mineral N pools fluctuate appreciably over time owing to a number of competing loss processes, and not just nitrate leaching/runoff. Lysimeter studies in Northern Ireland at the Hillsborough farm research site show that when chemical N inputs to grassland exceed 300 kg N ha^{-1} yr^{-1}, losses of nitrate from soil significantly exceed the amounts released into ground or drainage waters almost certainly because of gaseous (denitrification) N losses (Mills, 1997), which can be substantial (*i.e. > 70 kg N* ha^{-1}) on NI grassland (Jordan, 1989). It is also worth noting that researchers in Ireland failed to find any relationship between mineral N concentrations in grassland soils and nitrate concentrations in ground waters (Humphrey's *et al*., 2008).

b) Because soil cores (*0-900 mm*) taken to assess mineral N in soil profiles are collected at single points within fields, the N values obtained are point-specific. Consequently, large numbers of cores would need to be collected and analysed to accommodate the high degree of spatial heterogeneity in soil mineral N supply and formation (Murphy *et al*., 2013) across fields, particularly in grazing situations (Cuttle *et al*., 2001; Hutchings *et al*., 2007), but also under cutting management (Bailey *et al*., 2001) This would not be cost-effective or practicable.

Therefore, model estimates of nitrate loss from farms will be made based on soil typology and hydrology plus local climatic and farm management information instead of mineral N assessments in soil profiles (described in Section 7).

References

Bailey, J.S., Wang, K., Jordan, C. & Higgins, A.J. (2001). Use of Precision Agriculture Technology to Investigate Spatial Variability in Nitrogen Yields in Cut Grassland. *Chemosphere*, **42**:131-140.

Cuttle, S.P., Scurlock, R.V. & Davies, B.M.S. (2001). Comparison of fertilizer strategies for reducing nitrate leaching from grazed grassland, with particular reference to the contribution from urine patches. *Journal of Agricultural Science*, **136**: 221-230.

Hutchings, N.J., Olesen, J.E., Petersen, B.M. & Berntsen, J. (2007). Modelling spatial heterogeneity in grazed grassland and its effects on nitrogen cycling and greenhouse gas emissions. *Agriculture, Ecosystems & Environment*, **121**: 153-163.

Humphreys, J., Casey, I.A., Darmody, P., O'Connell, K.O., Fenton, O. & Watson, C.J. (2008). Quantities of mineral N in soil and concentrations of nitrate-N in groundwater in four grassland-based systems of dairy production on a clay-loam soil in a moist temperate climate. *Grass and Forage Science*, **63**: 481-494.

Jordan, C., 1989. The effect of fertiliser type and application rate on denitrification losses from cut grassland in Northern Ireland. *Fertiliser Research*, **19**: 45–55.

Mills, C.L. (1997). *The Nutrient Economy of Grazed Grassland*. PhD thesis, Faculty of Agriculture and Food Science, The Queens University of Belfast, Murphy, P.C.N., O'Connell, K., Watson, S., Watson, C.J. & Humphreys, J. (2013). Seasonality of nitrogen uptake, apparent recovery of fertiliser nitrogen and background supply in two Irish grassland soils. *Irish Journal of Agricultural & Food Research*, **52**, 17-38.

5. REINFORCED WATER MONITORING

5.1 Summary of results from reinforced water monitoring in agricultural catchments in proximity to most vulnerable water bodies

5.1.1 Fortnightly monitoring data

Monitoring was re-established in August 2016 with fortnightly grab sampling in 12 sub-catchments in Colebrooke and 13 sub-catchments in Upper Bann, following a hiatus from 2014 (Figure 18). Water quality data from these 25 sites provides supplementary evidence for current and subsequent annual derogation reports under Article 8(3) and Article 10(4) of the 2019 Derogation Decision. The catchments cover a gradient of agricultural intensities (generally lower in Colebrooke than Upper Bann) and are representative of the majority of soil types and soil hydrological classes found in Northern Ireland (Figure 19). Results for the period August 2016 – April 2020 are available for reporting.

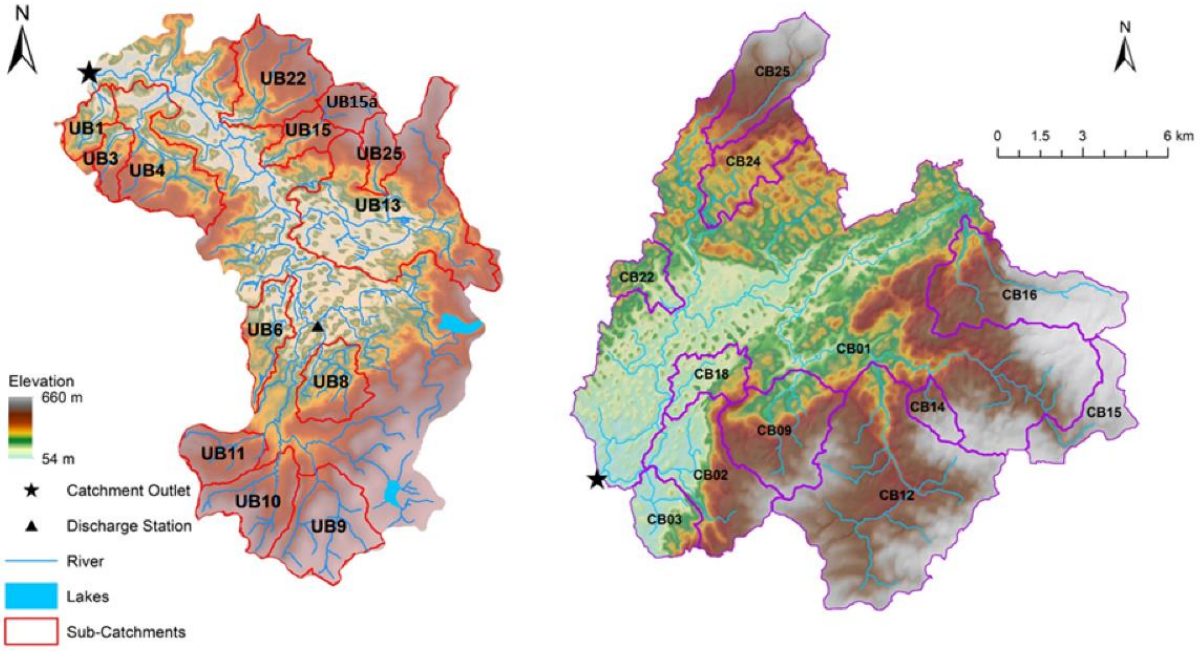


Figure 18: Upper Bann (left) and Colebrooke (right) catchments with monitored sub-catchments outlined.

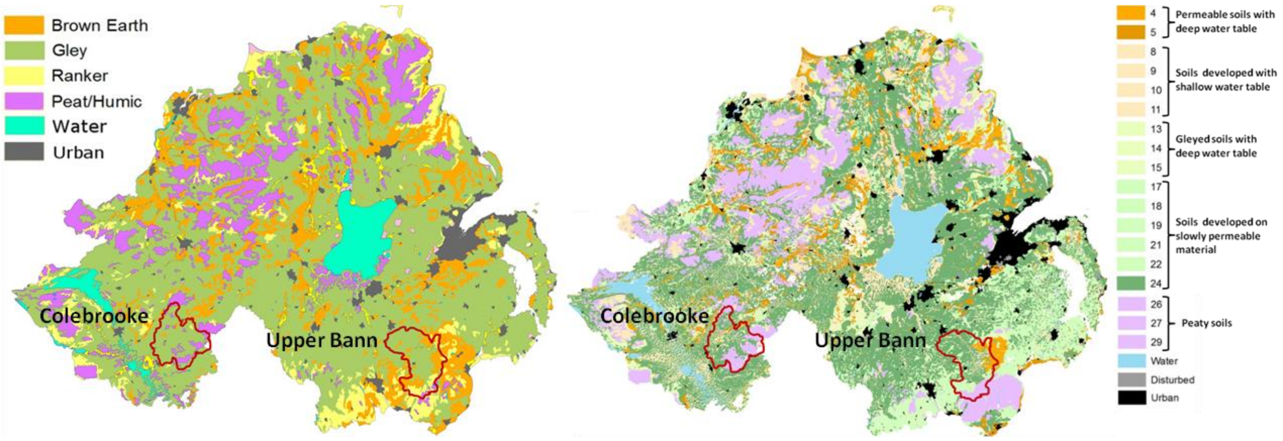


Figure 19: (a) General soil profile types in NI, and **(b)** General Hydrology of Soil Types (HOST) classes in NI

Results to date are indicative of the difference in land use and nutrient pressures between the Colebrooke and Upper Bann catchments as a whole. Figure 20 shows the time series of soluble reactive phosphorus (SRP) for the Upper Bann and Colebrooke sub-catchments from August 2016 to April 2020. Of the 13 sub-catchments monitored in Upper Bann on a fortnightly basis, only 2 catchments (UB 9 and UB10 in the Mourne headwaters) have notably lower median SRP concentrations (12 and 29 ug/L, respectively), compared to a range of 71-136 ug/L in the other catchments (for the August 2016 - April 2020 period). For the other catchments there is a clear annual pattern, repeated in each year of observation (Figure 20). Following the end of the closed period for manure applications, samples across all catchments to April have higher concentrations. Higher rainfall during this period increases the probability that grab sampling coincides with rainfall events where surface nutrients can be mobilised to the stream network. Spreading of nutrients in agricultural areas in all farmed sub-catchments at the start of the growing season provides a potential source for mobilisation during even small rainfall events due to elevated soil moisture following the winter. From April to August in each year there is a steady increase in SRP concentration in the catchments. This is during a period of lower rainfall, falling river levels and a dominance of soil and groundwater in the system. Increased concentrations may be linked to drying and re-wetting of soils through the summer, resulting in SRP release into soil water through soil aggregate breakdown, slaking and dispersion, and has been shown previously in plot and field studies (e.g. Majdalani et al., 2008, Cassidy et al., 2017). Higher rainfall and increased storm events through autumn and winter dilute concentrations throughout the system. Sampling at fortnightly intervals, however, tends to miss short duration storm events when most of the winter nutrient loads are transferred (Mellander et al., 2015).

Concentrations in the Colebrooke follow a similar pattern but are almost an order of magnitude lower than those in Upper Bann, with a median concentration range across catchments of 13-52 ug/L, lowest in the upland eastern sub-catchments which have large areas of forestry and peat and limited extensive farming. The increase in concentrations observed in Upper Bann between April and August is also observed in Colebrooke but with the peak occurring later, in late September/ October. The reasons for this are not clear and further investigation is required.

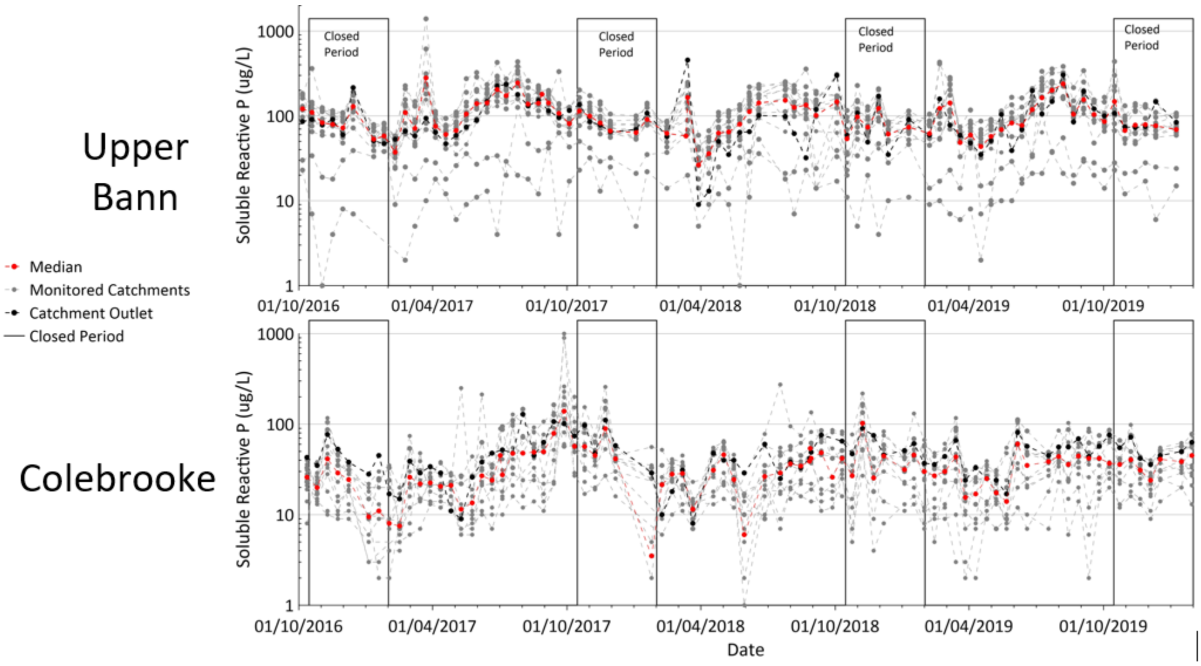


Figure 20: Time series of Soluble Reactive Phosphorus (SRP) for Upper Bann and Colebrooke sub-catchments. Median concentrations across all sub-catchments are indicated in red and the main outlet (UB24 and CB01) concentrations in black.

Though generally low, TON concentrations (Figure 21) showed greater variation among catchments ranging from median concentrations of 0.08 - 1.66 mg/L across Colebrooke sub-catchments and 0.26 – 3.13 mg/L in Upper Bann sub-catchments. Examining year-on-year changes in concentration for the catchments (Figure 21), there is a marked increase in concentrations during autumn/winter 2018/19, with concentrations more than double those in previous years.

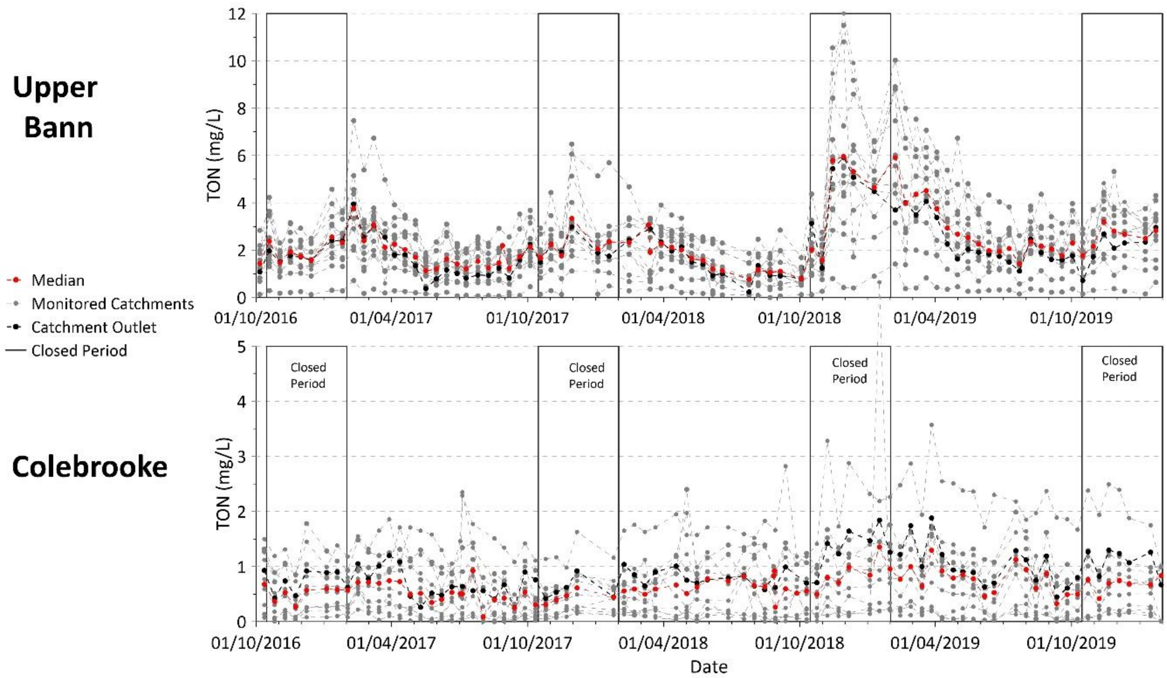


Figure 21: Time series of Total Oxidised Nitrogen (TON) for Upper Bann and Colebrooke sub-catchments. Median concentrations across all sub-catchments are indicated in red and the main outlet (UB24 and CB01) concentrations in black.

This is attributed to the prolonged dry conditions through summer 2018 that resulted in above average soil moisture deficits into autumn. In addition to increased mineralisation of N in organic matter during drying and re-wetting of the soil, denitrification within the catchment was impeded during the drought, with N remaining in the soil or infiltrating to groundwater for later release. This effect is also observed in the Colebrooke water quality time series, though less marked due to lower overall concentrations. A wetter summer in 2019 saw concentrations return to "normal" ranges in autumn/winter 2019/20. The potential impact of prolonged dry periods on increased TON loading from river systems into coastal receiving waters is of interest and should be highlighted in considering future climate impacts on aquatic ecosystems. Further examination of land use and farming intensity in these catchments is ongoing and will be needed to disentangle these relationships further. Differences in sources and pathways for nutrients among catchments (including surface/groundwater contributions) are also being examined.

5.1.2 Amplified monitoring in Derogated and non-derogated sub-catchments.

In 2017, higher resolution monitoring was instigated at 2 catchments in Upper Bann; one (UB03) with a number of derogated farms (the "derogated catchment") and the other (UB15a) a less intensive catchment with no derogated farmland (the "non-derogated catchment") (see locations in Figure 18). While fortnightly grab sampling continues at the outlets, it is now supplemented by flow gauging infrastructure, multiparameter probes (temperature, pH, conductivity, dissolved oxygen (DO)) and nutrient analyses. After initial trials of a battery operated Hydrocycle P analyser proved maintenance heavy and unreliable, automatic sampling at 7 hour intervals with laboratory analysis for P and N fractions was implemented from 1st October 2018 onward (for further information on the 24-7 approach; see Halliday et al. (2012), Neal et al. (2012), Jordan and Cassidy (2011)).

A comparison of the fortnightly monitoring data for the catchments, presented as box-plots for SRP and TON (Figure 22) over each hydrological year (from 1st October), show a number of differences. For SRP, concentrations were higher in the derogated catchment, with median concentrations of 114 ug/L in 2016/17, 98 ug/L in 2017/18 and 111 ug/L in 2018/19 compared to 99 ug/L, 89 ug/L and 86 ug/L, respectively, in the non-derogated catchment. Overlap of the notches (which in a box plot indicate the confidence interval around the median (median +/- 1.57 x $IQR/\sqrt{n}$), Figure 22) indicates that the differences are not significant however.

For TON, median concentrations in the derogated catchment were 1.61 mg/L in 2016/17, 2.04 mg/L in 2017/18 and 2.52 mg/L in 2018/19 compared to 1.29 mg/L, 1.32 mg/L and 2.08 mg/L, respectively, in the non-derogated catchment. In all years the derogated catchment had higher recorded concentrations than the non-derogated catchment but these were only significant in 2017-18. Further comparison is limited by the resolution of the data, which tends to miss short duration storm events and is biased toward low flows.

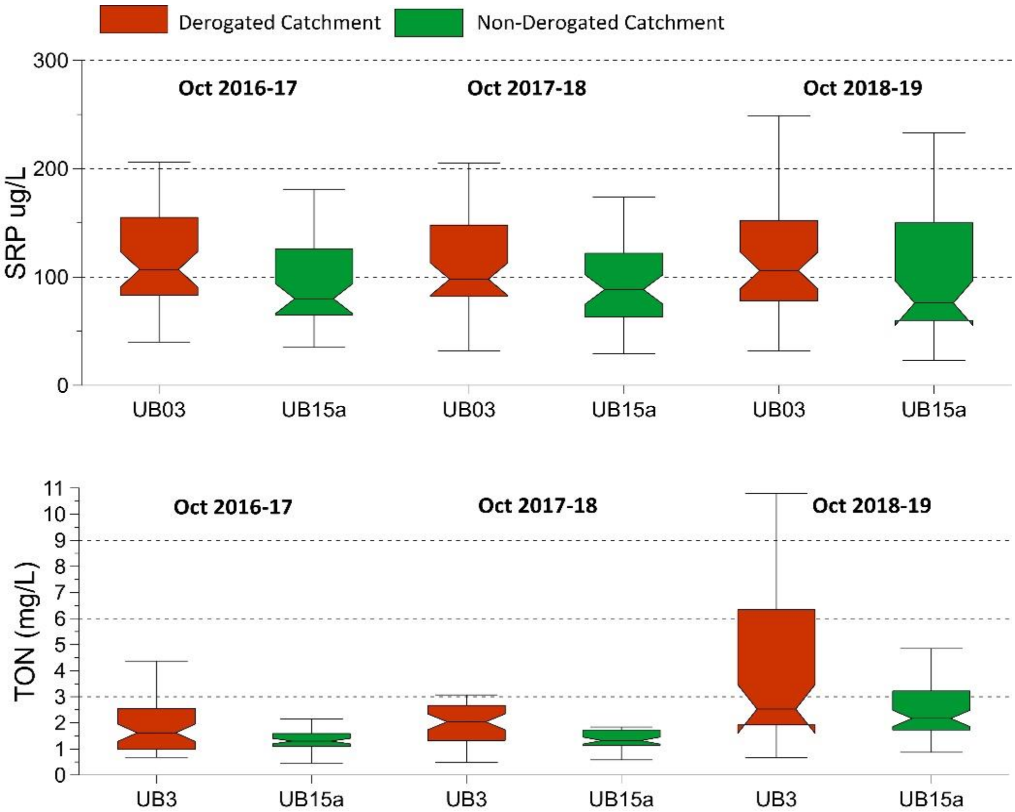


Figure 22: Box-plot comparison (showing median, 1st and 3rd quartiles and (interquartile range*1.5)) of fortnightly grab sampling data for SRP and TON in the derogated (UB03) and non-derogated (UB15a) sub-catchments for the hydrological years from 1st October. Notches display approximately the 95% confidence level in the median.

Time series from the higher resolution monitoring infrastructure provides greater insights into nutrient losses from the catchments, with Figure 23 showing the patterns of loss in 7-hourly data for Total P (TP) and Total Oxidised Nitrogen (TON) from October 2018 to February 2020. Base flow concentrations of TP in the Derogated catchment are consistently higher than in the non-derogated catchment, although differences are less clear during storm event pulses. For TON the difference in concentrations between catchment is more pronounced and were particularly notable during the autumn/winter 2018/19 nitrogen peaks associated with the summer drought.

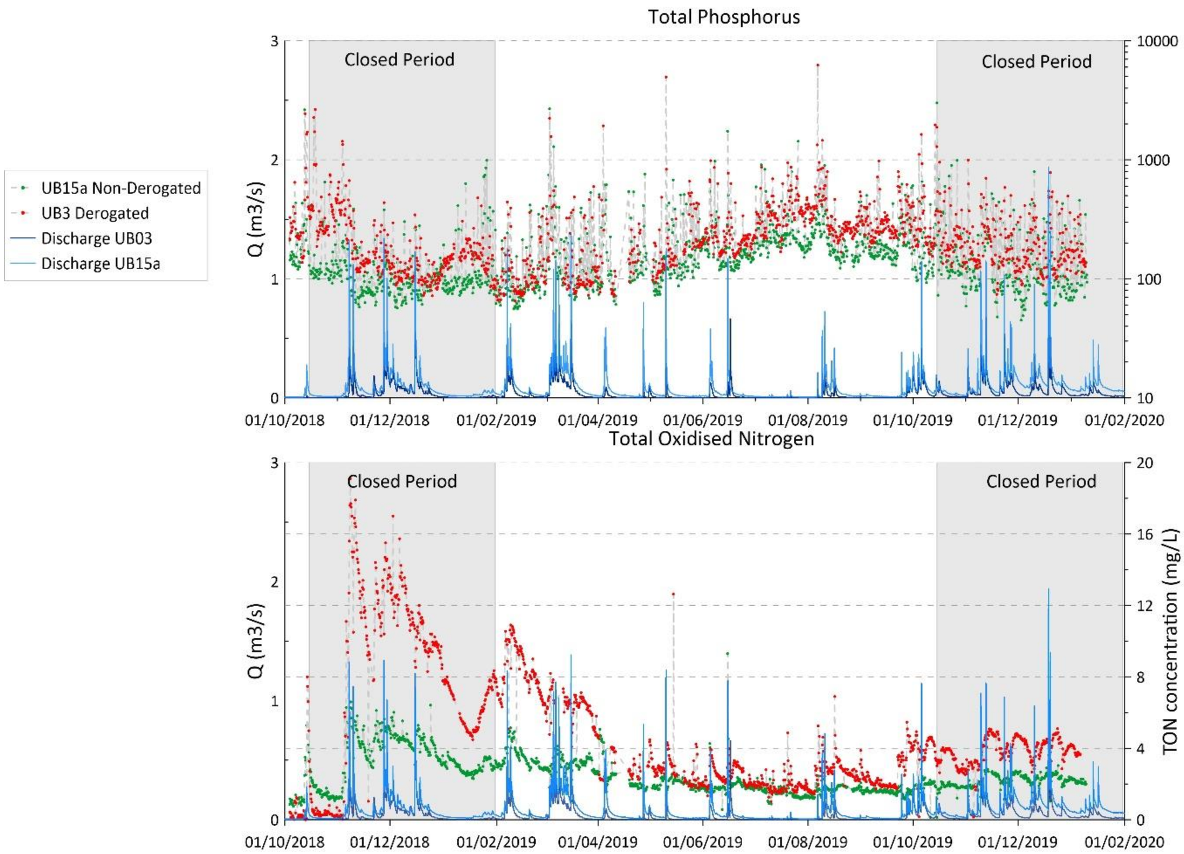


Figure 23: Comparative time series for Total Phosphorus and Total Oxidised Nitrogen (TON) for the Derogated (red) and non-Derogated (green) catchments from 7 hourly sampling data.

Rating curves to relate continuously monitored river water levels to discharge have been refined over the past 2 years and revised rating equations set in March 2020. These will be used to estimate the loads of nutrients lost from the catchment annually and when adjusted against the area of each catchment allow more accurate evaluation of the differences. Table 12 shows comparative loads across hydrological years, based on 7 hourly sampling and 15 minute discharge records since 1st October 2018. Although concentrations of TP are higher in UB03 (Figure 23), flows are lower and so the load for the derogated catchment (UB03) is less, at 0.86 kg/ha, than in the non-derogated catchment (UB15a) which lost 1.31kg/ha over the October 2018-19 period. From October 2019 to the end of the first half of the hydrological year (1st April 2020) the loads lost were broadly similar (1.03 kg/ha (UB3) and 1.10 kg/ha (UB15a), compared to the same period in the preceding year (01/10/2018- 01/04/2019: 0.53 kg/ha (UB3) and 0.76 kg/ha (UB15a)). This may be a legacy of the extremely dry conditions which persisted during summer 2018 in UB03, with soil moisture taking a long time to recover to field capacity in the autumn and winter months. As a result overland flow, which delivers phosphorus to the watercourses during rainfall events, was less likely to occur (infiltration dominated) and storm event P losses were thus lower in UB03 than in the wetter UB15a catchment. The impact on discharge in the catchments is consistent with this; UB3 having a total discharge over Oct 2018-19 of 0.96 x 10^6 m^3 (or 280 mm across the catchment area) compared to 2.18 x 10^6 m^3 (or 534 mm across the catchment area) in UB15a.

TON losses from the derogated catchment (UB3) were higher (23.42 kg/ha) compared to the non-derogated catchment (17.4 kg/ha) over the October 2018-19 period, with greatest differences in loading occurring during the autumn/winter period. The extremely dry soil conditions in UB3 during summer 2018 may also be implicated in this. Reduced denitrification in that catchment compared to the wetter non-derogated catchment may have contributed to increased TON loadings during winter/spring 2018/19, amplifying any differences in N loadings from agriculture in the catchment. Losses from both catchments are relatively consistent in autumn/winter 2019/20 for both P and N indicating that weather and landscape, rather than differences in agricultural intensity, are drivers of the observed differences.

Table 12: Load estimates for the Derogated and Non-Derogated catchments

Loss Estimates	Date Range	Catchment Area (km^2)	TON load (kg/ha)	TP load (kg/ha)
UB3 Derogated Catchment	01/10/2018 – 01/10/2019	3.42	23.42	0.86
	01/10/2019 – 01/04/2020		12.21	1.03
UB15a Non-Derogated Catchment	01/10/2018 – 01/10/2019	4.08	17.42	1.32
	01/10/2019 – 01/04/2020		10.77	1.10

Groundwater

Groundwater monitoring is a key requirement under Article 8 of the 2019 Derogation Decision for Northern Ireland, and is currently carried out by NIEA. Currently, NIEA are monitoring one borehole in the Colebrooke catchment and one in the Upper Bann catchment. To increase spatial resolution and focus on potential agricultural impacts on groundwater status, a baseline survey was undertaken in the autumn of 2016-spring 2017 to identify wells and springs suitable for sampling. As the survey found only 2 sources within the non-derogated catchment and 5 in the derogated catchment, the survey was extended to include another 6 sites just outside the catchment boundaries belonging to farmers who have land in the catchments. As the sample size is small and the hydrogeological context for all wells is similar (greywacke overlain by till) no differences can be inferred between derogated and non-derogated sites.

Seven groundwater sampling rounds have been completed to date, with 7-13 wells/boreholes sampled in each round (numbers have fluctuated due to access issues). The majority of households in the catchments rely on public water supply but the cost savings (£3000-4000 per year for dairy farms) make them attractive for larger farm businesses despite the difficulties in locating wells in these predominantly low productivity aquifer types (greywacke, granodiorite and granite with till overburden). Of the active wells and boreholes, operational groundwater supplies are boreholes with pumps to piped water supplies. Older hand dug wells and springs are no longer in regular use.

Sampling to date shows some diversity in nutrient concentrations among wells with a mean SRP concentration range of 9.8 – 164.3 ug/L and Total Oxidised Nitrogen (TON) concentration range of 0.04 – 6.2 mg/L. In the majority of wells TON concentrations during monitoring in December 2018 and March 2019 were higher than previously recorded, in common with the elevated surface water concentrations. There are issues with meeting the drinking water standards in terms for chloride and conductivity in one case, who is in contact with local authorities for assistance but in spite of upgrading the infrastructure around the well levels remain high.

Summary nutrient and key chemical parameters are shown in Table 13. In further sampling rounds effort will be made to include wells in the south of the catchments (in granodiorite and granite bedrock units) to increase the variety of aquifer types covered.

Table 13: Summary of chemical data for the groundwater sampling rounds in the Upper Bann derogated and non-derogated sub-catchments. 17 wells/boreholes were examined in total and sampled, dependent on access between 2-5 times, 5 in the derogated catchment; 2 in the non-derogated catchment and 10 in neighbouring catchments. Exceedances are highlighted in red.

Sites	**N**	**Conductivity** (µs/cm)	**pH**	**BOD** (mg/L)	**Chloride** (mg/L)	**Sulphate** (mg/L)	**SRP** (µg/L)	**TSP** (µg/L)	**TP** (µg/L)	**NO2** (µg/L)	**NH4** (µg/L)	**Nitrate** (mg/L)	**TON** (mg/L)
Exceedance limits		*800[2,3]*			*25[2]*	*250*				*50*	*500*	*37.5*	*37.5 ≠*
GUB1	6	544	7.82	1.0	24.5	8.5	15.7	18.2	42.5	1.5	29.7	0.01	0.04
GUB2	6	603	7.02	1.0	35.0	22.7	9.8	13.0	23.3	1.5	23.8	3.81	3.83
GUB3	6	351	7.83	1.0	17.7	9.3	31.7	34.0	38.0	0.5	44.3	0.16	0.21
GUB4	5	1291	6.53	0.7	458.7	20.0	16.6	20.4	31.2	3.2	136.6	2.40	2.54
GUB5	5	183	6.90	1.0	31.3	9.0	24.2	33.6	37.0	0.6	21.4	2.27	2.29
GUB6	7	268	6.80	1.3	14.4	11.1	43.4	50.7	74.1	0.7	48.3	6.16	6.21
GUB7	7	147	6.47	1.5	14.7	7.8	100.0	127.3	194.7	0.3	24.1	1.05	1.08
GUB8	7	250	7.44	0.8	14.7	20.8	164.3	170.1	177.1	1.3	210.6	2.12	2.33
GUB9	4	387	7.65	1.0	20.6	20.1	46.8	46.5	55.0	6.3	21.5	2.02	2.05
GUB10	3	376	7.72	0.3	14.5	9.5	17.7	23.3	24.0	1.3	13.7	0.11	0.13
GUB11	5	478	7.34	0.5	22.7	21.1	28.8	28.0	40.2	2.8	102.4	0.85	0.95
GUB12	5	490	6.88	1.0	32.7	30.0	46.6	52.2	73.0	1.4	24.0	3.15	3.18
GUB13	4	348	8.18	1.0	20.8	24.1	88.0	91.5	107.3	2.5	24.8	2.24	2.26
Median	**70**	370	7.29	1.0	20.1	16.4	25.0	30.0	41.0	1.0	24.5	1.9	1.9
Coefficient of Variation%		68	8	80	239	49	199	182	164	167	267		102
Mean concentration		**427**	**7.24**	**0.9**	**53.0**	**16.1**	**52.4**	**59.2**	**77.3**	**1.7**	**60.2**	**2.2**	**2.2**
±SD		292	0.61	0.8	126.3	7.9	104.6	107.9	126.8	2.8	161.1	2.2	2.2
Max		1612	8.34	3.0	851.2	34.6	552.0	556.0	613.0	18.0	1258.0	12.1	12.1
Min		98	5.74	0.0	9.7	4.3	4.0	4.0	10.0	0.0	0.0	0.0	0.0
Maximum Mean concentration		**1291**	**8**	**2**	**458.7**	**30.0**	**164.3**	**170.1**	**194.7**	**6.3**	**210.6**	**6.2**	**6.2**
Minimum Mean concentration		**147**	**6**	**0**	**14.4**	**7.8**	**9.8**	**13.0**	**23.3**	**0.3**	**13.7**	**0.01**	**0.0**

[1]± Standard deviation of 58 samples (n=58)

[2]To examine if groundwater abstraction is causing saline or other intrusions
[3]If value exceeds threshold value of 1875µS/cm, Groundwater chemical status is to be assessed to examine if the quality of groundwater that is abstracted for potable use is deteriorating, possibly resulting in a need for increased purification
[4]To examine the spatial extent of a groundwater body or group of bodies that are exceeding an EU Standard or threshold value

Further analysis of the results will relate the chemistry to land use, soil and geology and to continue at least annually thereafter.

References

Cassidy, R., Doody, D. G. & Watson, C. J. 2017. Impact of legacy soil phosphorus on losses in drainage and overland flow from grazed grassland soils. *Science of the Total Environment,* 575**,** 474-484.

Halliday, S. J., Wade, A. J., Skeffington, R. A., Neal, C., Reynolds, B., Rowland, P., Neal, M. & Norris, D. 2012. An analysis of long-term trends, seasonality and short-term dynamics in water quality data from Plynlimon, Wales. Science of The Total Environment, 434, 186-200.

Jordan, P. & Cassidy, R. 2011. Technical Note: Assessing a 24/7 solution for monitoring water quality loads in small river catchments. Hydrology and Earth System Sciences, 15, 3093-3100.

Neal, C., Reynolds, B., Rowland, P., Norris, D., Kirchner, J. W., Neal, M., Sleep, D., Lawlor, A., Woods, C., Thacker, S., Guyatt, H., Vincent, C., Hockenhull, K., Wickham, H., Harman, S. & Armstrong, L. 2012. High-frequency water quality time series in precipitation and streamflow: From fragmentary signals to scientific challenge. Science of the Total Environment, 434, 3-12.

Majdalani, S., Michel, E., Di-Pietro, L. & Angulo-Jaramillo, R. 2008. Effects of wetting and drying cycles on in situ soil particle mobilization. *European Journal of Soil Science,* 59**,** 147-155.

Mellander, P.-E., Jordan, P., Shore, M., Melland, A. R. & Shortle, G. 2015. Flow paths and phosphorus transfer pathways in two agricultural streams with contrasting flow controls. *Hydrological Processes,* 29**,** 3504-3518.

6. LAND USE AND AGRICULTURAL PRACTICE ON DEROGATED FARMS

6.1 Land use, cropping and agricultural practice on derogated farms

Agricultural holdings account for approximately 75% of Northern Ireland's land area, with 93% of the agricultural area being grassland. 90% of farms are classified as being mainly grazing livestock using EU farm classification typology.

Farm businesses operating under approved derogation in 2019 followed this pattern.

Farming activity, livestock and crops are detailed in Tables 14 and 15.

Table 14: Farming activity on farm businesses operating under derogation in 2019

Type	Frequency	Percent
POULTRY	7	1.6
DAIRY	390	88.8
CATTLE & SHEEP: LFA	8	1.8
CATTLE & SHEEP: LOWLAND	16	3.6
MIXED	18	4.1
Total	439	100.0

Notes: 1. Farm type - determined from DAERA Agricultural Census
2. Mixed farms – farms that have no dominant enterprise and do not fit into other categories
3. There were 2 farms where it was not possible to assign farm type so these farms have been excluded from the figures

Statistics in Table 15 are based on 424 farms submitting accounts – a full explanation to account for this figure is set out in section 8.2.3. Spring Wheat, Winter Wheat, Winter Barley and Spring Barley are the predominant crops, covering 62% of the total cropped area on all derogated farms. Crops are summarised in Table 15.

Table 15: Crops on farm businesses operating under derogation in 2019

Crop	Number of farm businesses 2019	Land area (ha)
Spring wheat	15	118.98
Winter wheat	37	362.82
Maize	12	165.38
Spring barley	15	77.74
Winter barley	9	55.83
Potatoes	3	4.03
Spring oats	3	16.74
Spring Triticale	1	4.05
Other	25	185.33
	Total land area:	**990.9**

7. MODELLING

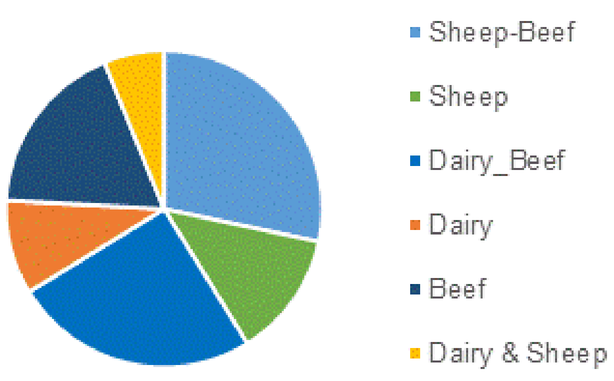
7.1 Preliminary results of model-based calculations of nitrate and phosphorus losses from derogated farms

FARMSCOPER nutrient modelling.

Results from a first application of the FARM Scale Optimisation of Pollutant Emission Reductions (FARMSCOPER) decision support tool (Gooday *et al*., 2014) to model P and N loads from the derogated and non-derogated catchments are available. The model allows farms to be specified individually and pollutant losses and potential efficacy of mitigation approaches to be assessed (Newell Price et al. (2011), with costs calculated if required.

The most recent published applications of the model (Gooday et al., 2014, Zhang et al., 2012) have involved large-scale applications using representative farm type data from censuses as the basis of modelling farms in each catchment. This application is at a smaller scale and for comparison between sub-catchments. Therefore, acquisition of specific farm level survey data for farms in both sub-catchments was necessary. The combined models for farms in each sub-catchment are then compared in an initial assessment of N and P losses, and evaluated against estimated nutrient loads from water quality monitoring at each sub-catchment outlet.

The catchment area of UB03 (derogated) is 381.3 ha with 83% of the area occupied by 18 farms. Eight of these farms returned survey data (52% catch. area) of which 4 are derogated (29% catch. area). The catchment area of UB15a (non-derogated) is 415 ha with 75% distributed among 26 farms, 11 of which returned data (39% catch. area). The areal coverage of different farm types within each catchment is given in Figure 24. A notable finding among farm types was that those with a component of dairying exhibit significantly greater export rates of P and NO_3 than those that do not (independent t-tests; mean diff = 0.87 kg P ha^{-1} yr^{-1}, $t=-4.86$, $p<0.001$; mean diff. = 24.3 kg N ha^{-1} yr^{-1}, $t= -2.99$, $p<0.01$).



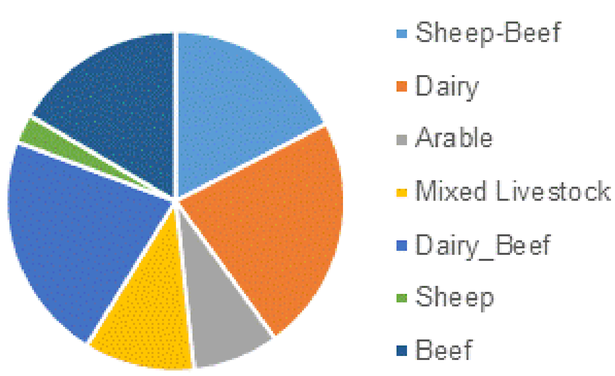


Figure 24: Areal coverage of predominant farm types in derogated (left) and non-derogated catchments (right).

For farms without survey data, the mean modelled export rates for farms of the same type (e.g. Beef, Sheep, Dairy, Dairy & Beef, Dairy & Sheep etc.) across both catchments was used to approximate the estimates of runoff and pollutant losses.

Summary outputs from (i) calculated farm P and N exports (from farm survey data), (ii) modelled concentrations from application of FARMSCOPER, and (iii) monitored water quality data at the outlets of both catchments, are presented in Table 16. Generally, concentrations of both N and P are higher in the derogated catchment, similar to monitored concentrations, although elevated. For derogated catchment UB03 the FARMSCOPER modelled mean drainage water P concentration of 326 µg/l is 39 µg/l greater than the mean monitored concentration (287 ug/L). The FARMSCOPER derived concentration can be lowered by adjusting to a more realistic runoff rate for the catchment (here 650mm), which then provides a closer match to the observed data. For nitrate agreement is less good than for P. The FARMSCOPER derived mean concentration (6.26 mg/L) exceeds the mean observed concentration (1.91 mg/L) by a factor of c.2.5. The situation is similar for catchment UB15; modelled nitrate export is again c. 2.5 times higher than the mean monitored concentration, and modelled mean P concentration is c.100 µg/l greater than the mean monitored concentration, although use of a higher runoff rate narrows the difference (Table 16).

Caveats in the comparison of modelled and measured concentrations include: (i) nutrient loads from human populations (wastewater) are not included in Farmscoper, (ii) monitoring data are presently limited to time-weighted concentrations whereas Farmscoper generates a flow-weighted concentration (iii) differences in the NI geoclimate from the regions defined for GB that are used in the model (iv) higher denitrification rates in Northern Ireland.

Table 16: Catchment P and NO_3 load and concentration estimates based on Farmscoper modelling, and summary concentration monitoring data of total phosphorus and NO_3 for each study catchment.

		Phosphorus		Nitrate	
	Catchment	*Derogated*	*Non-Derogated*	*Derogated*	*Non-Derogated*
Nutrient balances from Farm Survey Data Collection	Total farm load (kg)	590	501	11339	6512
	farmed area (ha)	318	310	318	310
	farmed area rate (kg ha^{-1} yr^{-1})	1.86	1.61	36	21
	other land loading (kg)	118	169	2271	2193
	Total catchment export (kg)	708	669	13610	8705
Farmscoper Modelled Concentrations	Mean conc. FS 570mm yr^{-1} (µg P , mg N /L)	**326**	**283**	**6.26**	**3.68**
	Mean conc. 650mm yr^{-1} (µg P / mg N L^{-1})	286	248	5.49	3.23
River water quality monitoring data	Monitored concentrations* n=28	µg P L^{-1}		mg N L^{-1}	
	Mean	**287**	**180**	**1.91**	**1.33**
	Median	201	124	1.69	1.29
	Range	67–1276	47–726	0.66–4.35	0.46–2.14
	SD	241	139	0.93	0.32

*fortnightly Sept. 2016 – Oct. 2017 incl.

Further work is required to assess whether adaptations to the model are practical to account for differences in rainfall and pathways for nutrients. This may require modifications to the models upon which FARMSCOPER is built including the "*Phosphorus and Sediment Yield Characterisation in Catchments" (PSYCHIC*), "*National Environment Agricultural Pollution"*

*(*NEAP-N) and “*MANure Nitrogen Evaluation Routine” (*MANNER) models and therefore would require considerable investment of time by the model developers, which needs to be assessed going forward. If resources allowed, an extension of modelling to additional Upper Bann and Colebrooke catchments, covering a broader range of agricultural intensity and pressures, would provide a more robust assessment of model performance.

Phosphorus Risk Modelling

Phosphorus (P) loss in run-off represents an economic loss to the farm, but also poses a threat to water quality. In catchments with high rainfall, clay soils and steep slopes - typical of many Northern Ireland catchments - overland flow is the primary pathway by which nutrients (particularly P) and sediment are transferred to surface waters. The identification of areas at high risk of P loss will enable farmers to target preventative measures in areas with greatest potential to reduce impacts on water quality. Risk is a combination of the presence of a contaminant that can be mobilised, i.e. a P source, and a pathway that can transfer it to water courses. Pathways can be modelled using high resolution LiDAR topographic data sets to generate HSA risk maps (following the method of Thomas et al. (2016)) which indicate areas at risk of contaminant mobilisation and transport in surface runoff pathways at field scale. A LiDAR survey of the Upper Bann catchment was commissioned in 2015. The river, stream and open ditch drainage networks were manually digitised and burned into the DTM to hydrologically correct it. The topographic wetness index (TWI; Beven and Kirkby (1979)) for the area was then calculated. To account for variations in soil hydraulic properties the TWI was developed into a Soil Topographic Index (STI) using the mean saturated hydraulic conductivity in the soil profile and soil depth for each soil series (1:50,000 General Soil Map of Northern Ireland, AFBI, 2009) in the catchment using historical data and expert judgement where required. Overland flow risk was based on ranked percentiles where the top 25% of STI values were determined as high risk HSAs. This decision was based on expert judgement (including HOST (Hydrology of Soil Types) classifications) and field observations of overland flow during storm events.

In terms of source risk, fields with soil phosphorus (P) in excess of the agronomic optimum for plant growth (>26 mg L-1 Olsen P) are considered to pose a particular risk to water quality, especially when those areas coincide with hydrologically sensitive areas (HSA) that focus surface runoff pathways during rainfall events. Combined soil testing under NAP E&I (16/4/03) and the EU EAA Soil Sampling and Analysis Scheme (SSAS) for 7,693 fields in 13 sub-catchments allowed the relationship between water quality, soil test P and hydrological connectivity to be explored in depth and has led to the development of a catchment carrying capacity framework (Cassidy et al., 2019) which identifies and prioritises areas (from field to farm to catchment scales) posing risks to water quality based on:

(1) Exceedance of the agronomic optimum soil test P and

(2) Identification of hydrologically sensitive areas (HSAs) posing a risk of nutrient transfer to watercourses.

These risk areas are targets for mitigation to reduce soil P and intercept runoff pathways to watercourses. The framework also, however, indicates soil P opportunities (low soil P and low HSA risk) for redistributing these risks (i.e. identifying safe locations where manure can be applied), adaptively and concurrently across scales from field, to farm to sub-catchment, possibly as part of developed advice provision and new schemes, in integrated nutrient management planning or trading.

Soil test P and HSA risk results

Within the entire sample set, results showed that 41% of fields exceeded the agronomic optimum for soil P across the sub-catchments (Figure 25), with a majority of those (28.9%) at Index 3. A larger proportion (45.7%) was below the agronomic optimum (<21 mg L-1 Olsen P), with 9.9% of fields at Index 0. A majority of farms exhibited an imbalance in soil fertility across fields; with a tendency to have high soil P in large fields close to the farm compared to those at the extremity of the holding.

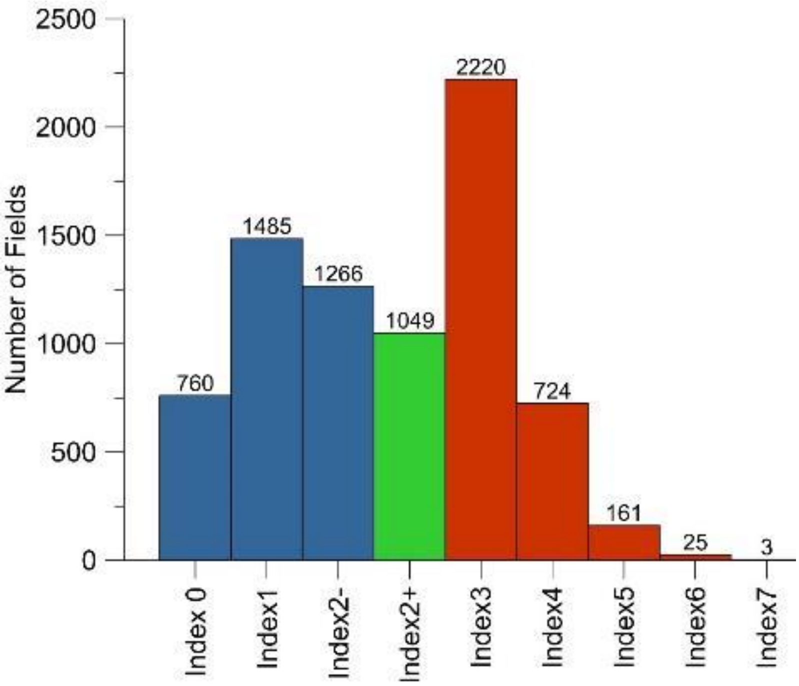


Figure 25: Distribution of soil P status across the sub-catchments of Upper Bann**.**

The proportion of the total area of all fields on individual farm holdings (n=531) exceeding the agronomic optimum soil test P was, on average, 36.8 % (Figure 25), while the proportion of total field area also classed as high HSA risk was 7.4 %. Across individual catchments there was a considerable spread in risk - between 0.1 to 66.4% of catchment areas were above the agronomic optimum, and between 0 and 22.9% of sub-catchment areas were above the agronomic optimum and also at high HSA risk.

Distributions of the spatial data (high HSA risk and soils above the agronomic optimum soil test P) were compared with river soluble reactive phosphorus (SRP) concentration measured fortnightly over one year (Oct 2016-17). When compared with the available water quality data for each sub-catchment (an annual median SRP concentration) (Figure 26), strong correlations were found between water quality at the outlet of each sub-catchment and (Figure 26 (a)) the area of a catchment above the agronomic optimum for soil test P (linear, r^2=0.92) and (Figure 26 (b)) between the area of a catchment above the agronomic optimum for soil test P and also at high risk of runoff (power-law, r^2=0.97). Compared with a target SRP river concentration threshold of 48 µg L-1 in a wider catchment, the results indicated the high soil P carrying capacity area of the sub-catchments was 15 % and that almost no HSA could carry high soil P (greater risk when >1.5 % of sub-catchment area). Above these levels and the models indicate that WFD water quality targets in the catchment will not be met.

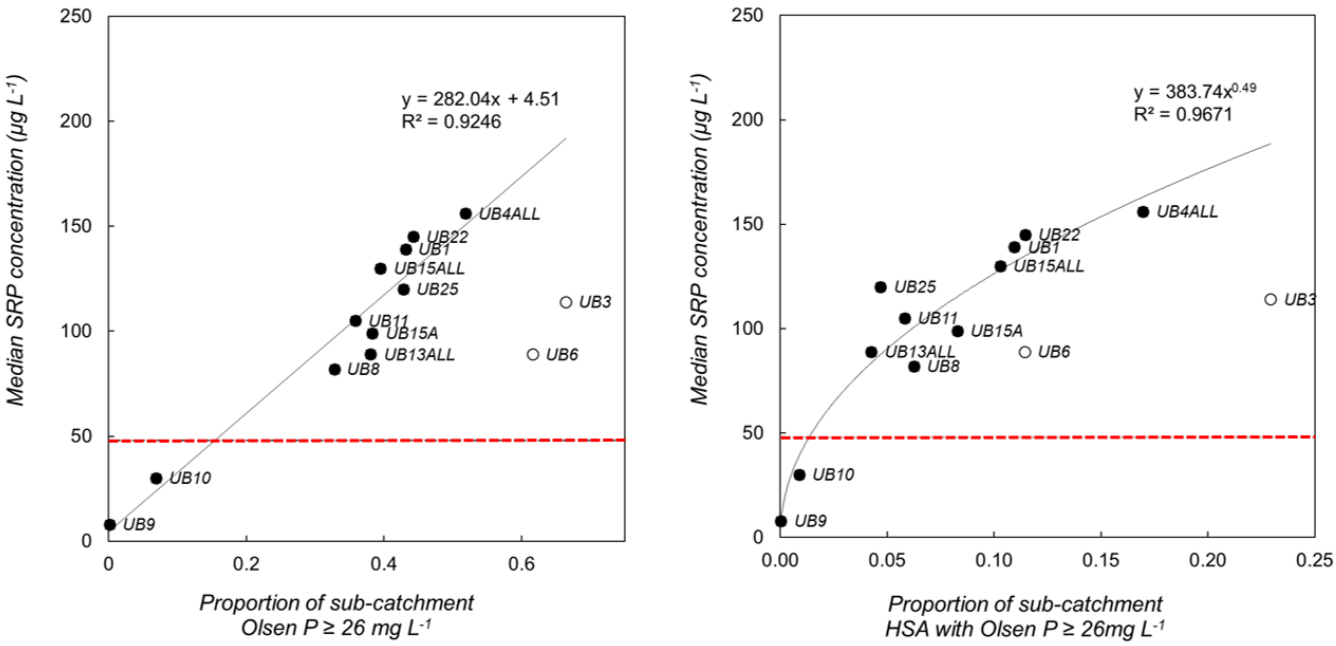


Figure 26: Relationships between (a) median annual SRP concentration at the sub-catchment outlet and the proportion of each sub-catchment in excess of the agronomic optimum (26 mg L^{-1} Olsen P) and (b) median annual SRP concentration at the sub-catchment outlet and the proportion of each sub-catchment in excess of the agronomic optimum (26 mg L^{-1} Olsen P) and with a high (25th percentile) HSA risk. The dashed red line indicates the site specific class boundary SRP concentration used to differentiate between moderate (>threshold) and good (<threshold) ecological status classification (48 ug L^{-1}) at the Upper Bann outlet. Open symbols are outliers to the power-law trend line identified using Cook's Distance.

Opportunity Mapping

The opportunities to redistribute soil P and HSA risk were analysed by identifying fields with below optimum soil P, where HSA risk was also minimal and the area was of a scale to allow nutrient application by tractor and slurry spreader (>0.2 ha and regular shape).

These ranged from 0.4 % to 13.8 % of sub-catchment areas in Upper Bann. Many holdings have no on-farm capacity for nutrient re-distribution (from 17% of farms in UB01 to 56 % of farms in UB03) which limits their options to either reducing stocking rates or moving nutrients off-farm.

Approaches to redistribute P were evaluated to include:

1. Redistribution of P within the farm to areas of opportunity
2. Redistribution of P among farms in the same sub-catchment
3. Redistribution of P among sub-catchments
4. Soil P drawdown and reduced farm P balance
5. Targeted interception measures

Options 1 to 4, above, consider the need for smart incentives for P use optimisation at the field, farm and sub-catchment level and this is important as P as a mineral fertiliser becomes more of a national and international resource issue for food security (e.g. Scholz et al. (2013)). Translation of targets defined using the carrying capacity framework into operational plans on farms and within sub-catchments is the next stage and will require a concentrated and coordinated effort among all the stakeholder groups involved, including both advice and scheme development. This is recognised as a challenge as the data required from farmers for options such as P redistribution and P drawdown in soils requires collection and organisation. The development of decision support tools, to include both maps (Figure 27) and graphical presentation of soil agronomic status, risk and opportunity for redistribution (following a format based on that in Figure 28) will be an intrinsic component of the communication process among stakeholders and in implementing the framework.

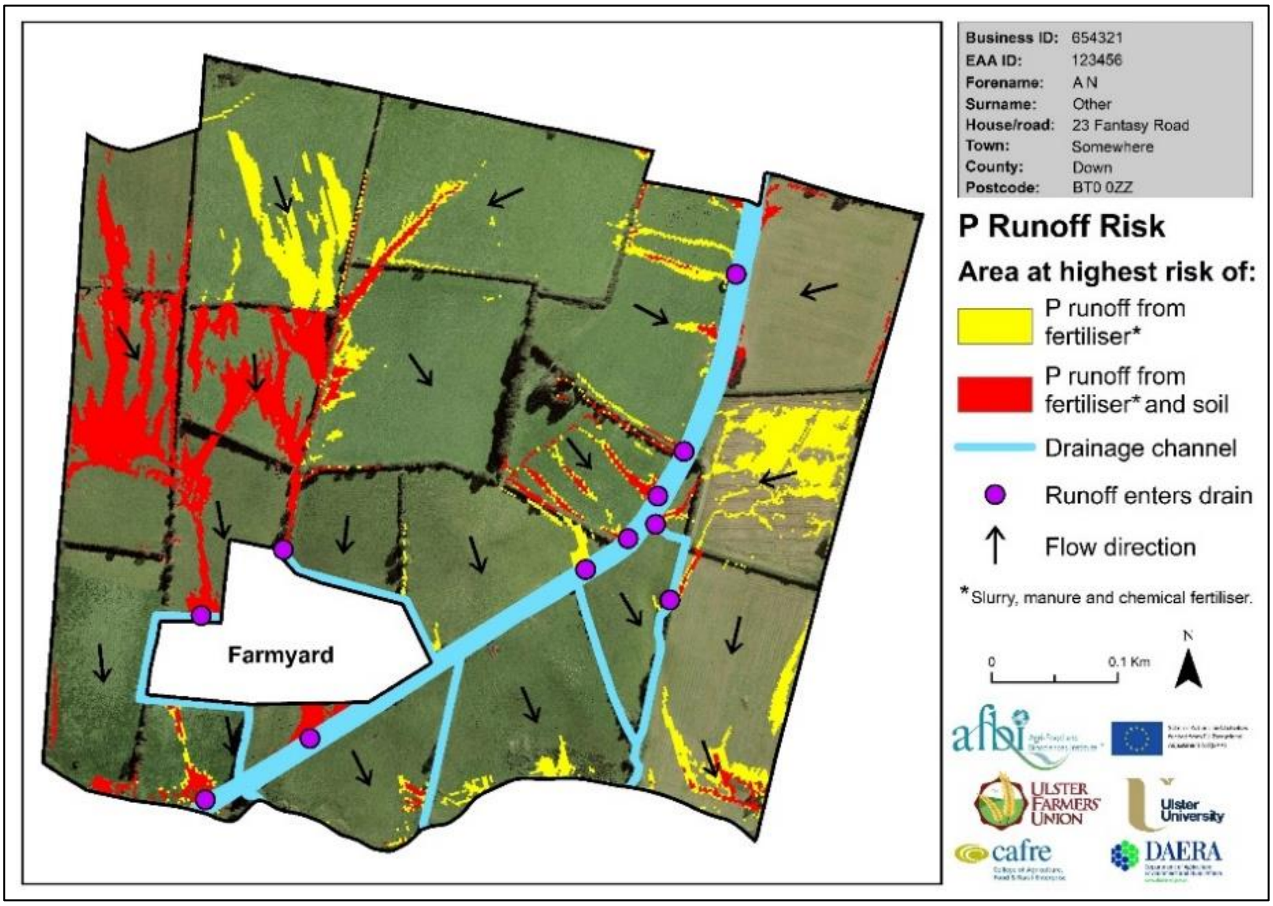


Figure 27: Exemplar risk map for P runoff risk (both surface applied nutrients and legacy soil P) produced as part of the EU EAA Soil Sampling and Analysis Scheme for the Upper Bann monitored sub-catchments.

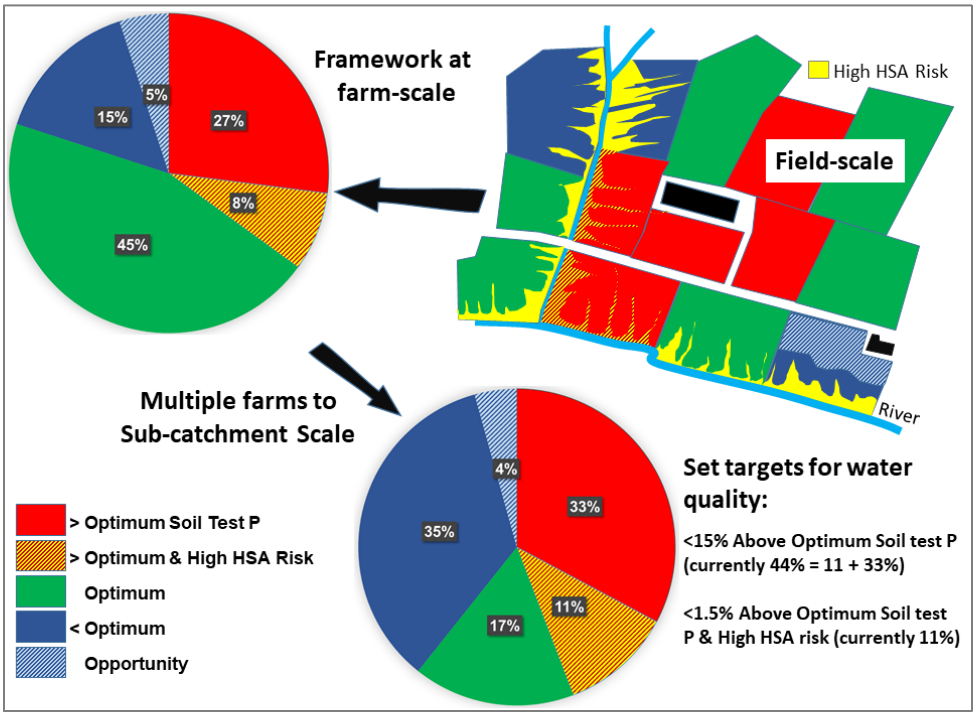


Figure 28: Schematic of the assessment framework applied at field, farm and sub-catchment scales to identify risk (high soil P and high HSA), opportunities (low soil P and no HSA) and management targets for nutrients relative to water quality. Note that the field schematic and farm scale pie chart summaries are illustrative examples (due to farm data confidentiality) but the sub-catchment summary is based on actual data for B2.

Conclusions

1. The catchment level assessment applied as part of this framework could feed into the development of a land use choices matrix integrating environmental, social, economic and cultural concerns, for use in other national or international river catchments.

2. Soil Olsen P and HSA runoff risk modelling were combined and used to identify priority areas for mitigation at field, farm and catchment scales.

3. Strong correlations were found between water quality at the outlet of each sub-catchment and (1) the area of a catchment above the agronomic optimum for soil test P (linear, r2=0.92) and (2) between the area of a catchment above the agronomic optimum for soil test P and also at high risk of runoff (power-law, r2=0.97).

4. Compared with a target SRP river concentration threshold of 48 µg L-1 in a wider catchment for good ecological status based on P, this study indicated that the carrying capacity for above agronomic optimum soil P was 15 % of the individual sub-catchments’ area.

5. The study also indicated that almost no HSAs (<1.5 %) could carry similarly high soil P concentrations i.e. that all these areas require mitigation.

   A carrying capacity framework was developed to indicate soil P opportunities for redistributing these risks, adaptively and concurrently across scales from field, to farm to sub-catchment possibly as part of developed advice provision and new schemes, in integrated nutrient management planning or trading.

References

Beven, K. J. & Kirkby, M. J. 1979. A physically based, variable contributing area model of basin hydrology / Un modèle à base physique de zone d'appel variable de l'hydrologie du bassin versant. Hydrological Sciences Bulletin, 24, 43-69.

Cassidy, R., I.A. Thomas, A. Higgins, J.S. Bailey, P. Jordan (2019) A carrying capacity framework for soil phosphorus and hydrological sensitivity from farm to catchment scales, Science of the Total Environment, https://doi.org/10.1016/j.scitotenv.2019.05.453

Gooday, R. D., Anthony, S. G., Chadwick, D. R., Newell-Price, P., Harris, D., Duethmann, D & Winter, M. (2014). Modelling the cost-effectiveness of mitigation methods for multiple pollutants at farm scale. *Science of the Total Environment*, **468**: 1198-1209

Newell Price, J. P., Harris, D., Taylor, M., Williams, J. R., Anthony, S. G., Duethmann, D., & Misselbrook, T. H. (2011). An inventory of mitigation methods and guide to their effects on diffuse water pollution, greenhouse gas emissions and ammonia emissions from agriculture. *Report prepared as part of Defra Project WQ0106, ADAS and Rothamsted Research North Wyke.*

Scholz, R. W., Ulrich, A. E., Eilittä, M. & Roy, A. 2013. Sustainable use of phosphorus: A finite resource. Science of the Total Environment, 461-462, 799-803.

Thomas, I. A., et al. (2016). A sub-field scale critical source area index for legacy phosphorus management using high resolution data. Agriculture, Ecosystems & Environment 233: 238-252. Zhang, Y., Collins, A. L. & Gooday, R. D. 2012. Application of the FARMSCOPER tool for assessing agricultural diffuse pollution mitigation methods across the Hampshire Avon Demonstration Test Catchment, UK. *Environmental Science & Policy,* 24**,** 120-13.

8. COMPLIANCE WITH THE DEROGATION CONTROLS FOR 2019

8.1 Derogation controls in Northern Ireland

NIEA, on behalf of DAERA, is the competent authority for enforcement of the NAP legislation in Northern Ireland. In accordance with Article 4 of the 2019 Decision, the 2019 NAP Regulations require farmers in Northern Ireland, who wish to benefit from derogation, to submit an annual application to the NIEA by 1 March for that calendar year. NIEA have 28 days from receipt to make a decision on whether to grant or refuse the application.

The Regulations also require farmers in Northern Ireland to prepare and keep a fertilisation plan for the calendar year in accordance with Article 6 of the 2019 Decision. This must be available on derogated farms no later than 1 March of that calendar year.

Additionally, in accordance with Article 6 of the 2019 Decision, the Regulations require farmers in Northern Ireland to submit fertilisation accounts to NIEA for the previous calendar year by 1 March of the following year. Therefore, applications for derogation must be accompanied by the fertilisation account for the previous year, where relevant.

8.2 Compliance with the derogation controls

Compliance with the Decisions is assessed in three key ways:-

1. administrative checks on derogation applications for the current calendar year;
2. on-farm inspections of records from previous years, current fertilisation plans, farm facilities and fields; and
3. administrative checks of fertilisation accounts for the previous calendar year.

8.2.1 Administrative checks of 2019 derogation applications

In 2019, there were 460 approved derogation applications however 19 subsequently withdrew leaving 441 farms.

8.2.2 2019 On-farm inspections

In accordance with the 2019 Decision, at least 5% of derogated farms are selected for on-farm inspections. In 2019, a total of 460 farmers applied for a derogation from NIEA with 19 subsequently withdrew their applications before the cut-off point. 31 of the remaining farms (5%) were selected for on-farm inspections. During inspection derogated farms are assessed against all of the NAP and Derogation requirements.

There were 3 farms with non-compliance detected from the 31 inspected. Two had water pollution incidents: one from their on-farm storage facilities, and one had insufficient storage; and the other farm had undertaken closed season spreading.

8.2.3 Administrative checks of fertilisation accounts for 2019

In 2019, a total of 460 farmers applied for a derogation. Nineteen of these farm businesses withdrew in-year leaving 441

The online system previously introduced for the submission of fertilisation accounts continued to be the main method used for the calendar year 2019. Of the 441 farms operating under derogation in 2019, 424 farm businesses submitted their fertilisation accounts online.

Table 17 shows the finalised results of administrative checks on the 441 fertilisation accounts submitted for the calendar year 2019. Compliance with the rules has decreased compared to 2018. Targeted training and guidance will assist increasing compliance levels, particularly with newly derogated farms.

A total of 15 non-compliances were detected in the accounts of 13 farm businesses. Non-compliance was mostly aligned to P balance, and nitrogen loading. NIEA continues to engage with colleagues in the Department and stakeholder representatives regarding these non-compliances. Reminders were issued by NIEA to agents and farmers in relation to accounts which resulted in no non-compliances for late accounts this year.

Table 17: Compliance of fertilisation accounts for 2019

Measure Description	Average (min-max)	Number of Breaches
80 % grassland	98.05 (80.22 – 100)	0/441
Total grazing livestock N (up to 250 kg N/ha/year)	206.35 (54.04 – 288.17)	4/441
Total livestock manure N loading (170 kg N/ha/year non-grazing + 250 kg N/ha/year grazing)	206.26 (54.04 – 286.57)	3/441
Total chemical N fertiliser usage on grassland (not to exceed 272 or 222 kg N/ha/year for dairy or other farms respectively)	171.05(0 – 270.41)	0/441
Total chemical N fertiliser usage on land other than grassland (not to exceed crop requirement)	149.85 (0 -220)	0/94*
Phosphorus balance up to 10 kg P/ha/year	4.7 (-41.71.98 – 24.83)	11/441
No, partial or late records	N/A	0/441

*Statistics based on farms which had land other than grassland and which used chemical N fertiliser.

Total grazing livestock N (up to 250 Kg/N/ha/year) & Total livestock manure N loading (170 Kg/N/ha/year non-grazing + 250 Kg/N/ha/year grazing) are classed as the same Breach on the Cross Compliance CC2 Form. Therefore there are 4 breaches in total.

Table 18: Predicted and observed statistical values (verified for land area) for farm businesses which operated under derogation in 2019.

Average (min-max)	**Predicted from Applications 2019**	**Fertilisation accounts 2019**
Grassland area (%)	98.38 (80.3 - 100)	98.05 (80.22 – 100)
Farm size (ha)	86.05 (6.83-358.96)	87.28 (6.72 - 362.49)
Total livestock manure N loading (kg N/ha/year)	209.8 (-33.11 - 249.57)	206.26 (54.04 – 286.57)
Grazing livestock manure N loading (kg N/ha/year)	N/A	206.35 (54.04 – 288.17)
Chemical N fertiliser usage (kg N/ha/year)	N/A	14982.7 (0 – 68760)
Phosphorus (P) balance (kg P/ha/year)	N/A	4.7 (-41.71 – 24.83)

Table 18 shows statistics for observed values for the 441 farm businesses that submitted fertilisation accounts for the calendar year 2019. The values are calculated (using land areas verified through cross-checks with other data sources) from information supplied in the fertilisation accounts. Observed values are compared, where possible, to predicted values from the initial 460 approved derogations. Fertilisation accounts which have produced outlying values are likely to be examined further and the farm business may be more likely to be subject to an on-farm inspection (due to a higher environmental risk rating).

9. GUIDANCE AND TRAINING TO SUPPORT THE DEROGATION

9.1 Nitrates derogation guidance

In October 2019, following approval of the Northern Ireland derogation for 2019-2022 by the European Commission, a letter was sent to all farm businesses that had applied for a Nitrates Derogation for 2019, informing them that all previous conditions for derogated farms would be carried forward. New additional requirements and subsequent changes to the content of the fertilisation account were also highlighted. Work commenced to update the Nitrates Derogation Guidance, Fertilisation Plan and Fertilisation Account. In January 2020, these farm businesses were sent a letter reminding them to submit their Fertilisation Account using the new online system. The deadline for submission of Fertilisation Accounts was extended to 31 March 2020. In addition they were also reminded to apply using the online system again by 31 March 2020, if they wished to continue to operate under a derogation for 2020.

As a result of the COVID-19 pandemic the deadline for application for derogation for 2020 and the submission of 2019 fertilisation accounts were further extended to the 30 April 2020.

9.2 Nitrates derogation training

In 2019/20 the College of Agriculture, Food and Rural Enterprise (CAFRE) within DAERA continued to provide a wide range of support on the Nitrates Derogation for farmers in Northern Ireland.

Farmers operating under derogation in 2019 were supported by CAFRE Dairy Development Advisers on a one to one basis as and when requested.

9.3 Other training and support associated with Nutrients Action Programme

Other training related to the NAP that took place in 2019-2020 included:

- An update for CAFRE Development Advisers on NAP 2019-2022 changes, additional requirements for Nitrates Derogation, fertilisation account and derogation application process.
- CAFRE Development Advisers continued to deliver nitrates and nutrient management to almost 3000 farmers in Business Development Groups.
- Through the Family Farm Key Skills (FFKS) initiative and working in conjunction with Ai Services, CAFRE provided Nutrient Management Planning & Understanding your Soil Analysis training to farm business within the Strule & Colebrook Catchments targeted under the European Area Aid Soil Sampling and Analysis Scheme.

In addition, CAFRE Advisers and the Agri-Environment Team successfully dealt with numerous calls from farmers, Advisers and Consultants on nitrates related issues including the closed period, manure exports (online record submission) and nitrates derogation (online record submission).

9.4 CAFRE Nutrient Calculators

As described in the previous report, CAFRE has lead responsibility for the development and maintenance of a suite of five online calculators designed to help farmers to manage their farms to comply with various aspects of the NAP Regulations. Updates of figures used in the calculators were carried out to reflect the NAP 2019-22 changes. The calculators are available on the DAERA web-site at: www.daera-ni.gov.uk. The calculators continue to be well used and Table 19 shows the number of unique users for each of these online calculators at March 2020. The total number of users continued to increase, up 8% in 2019/20.

Table 19: User numbers for online calculators

Calculator	Number of users at March 2020
Livestock Manure Nitrogen Loading	4852
N Max for Grassland	1397
Crop Nutrient Recommendation	1212
Phosphorus Balance	1089
Livestock Manure Storage	2094

9.5 Other communication methods

In 2018-2019 DAERA issued technical information in the form of a number of press articles and management notes through various channels including the agricultural press, the DAERA website and the Farm Advisory System Newsletter to: update farmers on water quality and nutrient issues, and promote the nitrates derogation, nutrient management planning and the CAFRE Nutrient Calculators. These articles are also published on DAERA's website along with frequently asked questions, NAP summary and guidance booklets, derogation guidance booklet, and booklets for derogation fertilisation plans.

DAERA continues to highlight the NAP Regulations, including nitrates derogation, at a variety of agricultural shows, events and meetings. For example, in meetings held by the CAFRE Dairy Development Advisers in January and February 2020, they reminded farmers about the need to meet the 31 March 2020 deadline for submission of their fertilisation account and next year's derogation application.

10. RESEARCH PROJECTS (Evidence and Innovation (E&I))

In order to underpin the implementation of the Directive and the action programme measures in Northern Ireland, DAERA commissioned AFBI to carry out a range of research projects during the period 2008-2012. Some of the research was undertaken in accordance with Articles 8.2-8.6 of the 2007 Derogation Decision for Northern Ireland, granting derogation for intensive grassland systems, and is still on-going. Further research in support of the 2019-2022 derogation and NAP has also been commissioned by DAERA. A summary of key findings from on-going research and details of new Evidence and Innovation (E&I) research projects, are provided here.

10.1 E&I Project 0618 - Monitoring the effectiveness of the Nitrates Action Programme for Northern Ireland (On-going Research)

Under Article 8.6 of the 2007 Derogation Decision, and as part of monitoring the effectiveness of the NAP for Northern Ireland, a representative soil sampling scheme (RSSS) has been operated by AFBI since 2004, to identify the impact of the NAP on soil fertility in Northern Ireland, especially on soil Olsen-P. In the RSSS, 500 grassland fields across Northern Ireland are sampled, 100 per year, on a five-year rolling basis. The fields selected were located on intensively stocked, but non-derogated farms operating at near to the 170 kg N/ha manure loading limit. Alongside this RSSS a 5km grid soil sampling survey across all land cover classes was initiated in 2004 and repeated every 10 years.

The overall objective of this project is to monitor soil quality across Northern Ireland in both the intensive sector (RSSS monitoring) and the general agri-environmental landscape (5km monitoring), and was designed to provide soil data to support DAERA's evidence-based responses to EU Directives, particularly the Nitrates Directive.

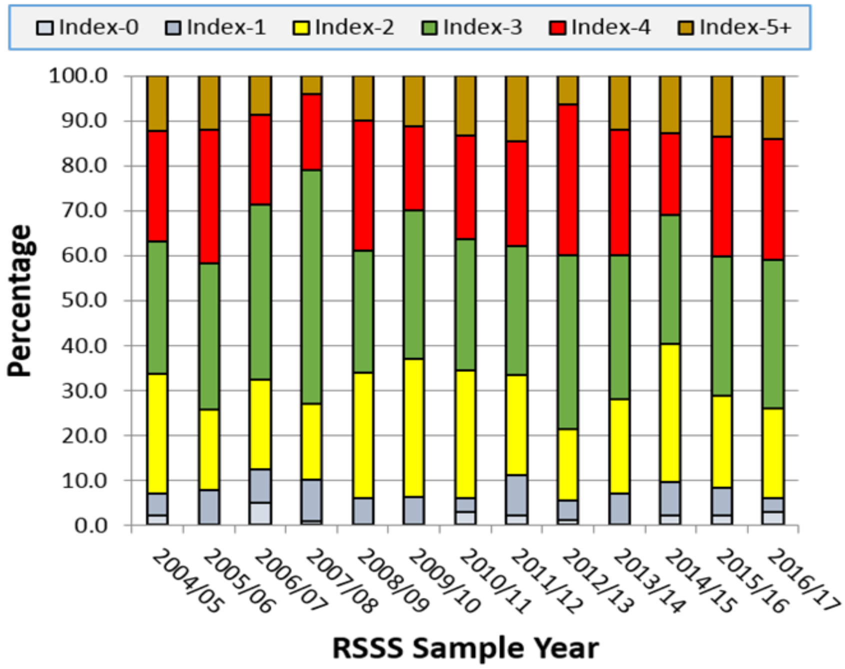


Figure 29. Olsen P-Index in the top 75mm of the soil profile for years 2004/05 to 2016/17

The RSSS 2016/17 sampling campaign was completed on schedule in March 2017. As the headline indicator of soil nutrient status, Olsen-P was used to illustrate the results of the project in 2016/17. Figure 29 places the RSSS Olsen-P results from 2015/16 and 2016/17 within the context of the analysis to date. Once again the general trend observed in the sampled intensive grassland sector is evident within these 2 years; soils with sub-optimal levels of soil P *(< Index 2*) have a frequency of less than 10% and soils oversupplied in P (*Index 3 and above*) account for approximately 70% of the samples.

Paired two-tailed t-tests between the RSSS 2016/17 and RSSS 2011/12 Olsen-P data sets (*with average values of 44.77 mgl^{-1} and 42.53 mgl^{-1} respectively)* suggest that although an increasing trend is observed, it is not statistically significant. Figure 30 illustrates the relationship. Interestingly there was a small but statistically significant increase in Olsen P between RSSS 2006/07 (the initial sampling cycle) and RSSS 2016/17 with average values of 40.02 mgl^{-1} in 2005/07, and 44.77 mgl^{-1} in 2016/17. This would imply that despite possible re-distribution of manure nutrients within some farms, overall there has been little reduction in P inputs to these relatively intensive farms.

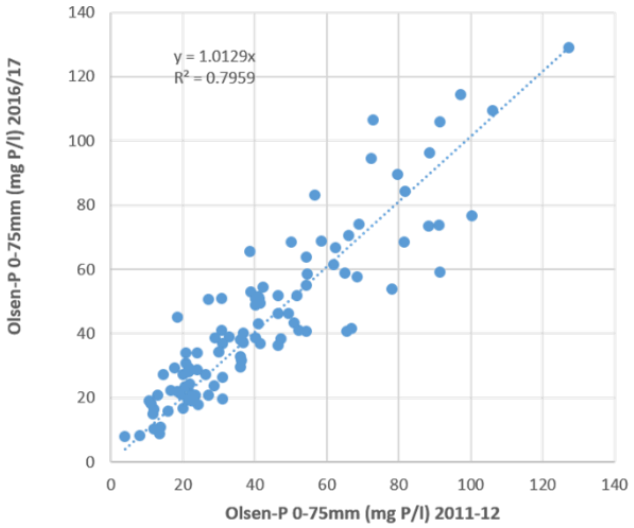


Figure 30. Linear regression of RSSS Olsen-P data for winter 2016/17 compared to matching data for winter 2011/12. The fitted line, forced through the origin, is shown along with the slope and R^2 value (df=99).

The results from the 5km regular grid survey, which covers land under a wide range of farming intensities from low to high, unlike the RSSS (*which focuses on the intensive sector*), indicate that there has been a small increase in soil P in lowest Olsen-P ranges Index 0 and 1, but a decrease in soil P in soils initially at Olsen P Index 4 and above. This is possibly because of efforts by land owners to distribute manure-P away from high P index fields (*nearest to farmyards*) where historically it had been primarily applied, to land receiving little organic manure-P in the past (Table 20).

Table 20: 5km grid survey – comparing Olsen-P levels in 2 periods

	Olsen P 0-7.5cm		
5km survey	***2004/5***	***2014/15***	
All Samples	**27**	**27**	NS
Index 0	6.4	9.7	P= .000
Index 1	13	17.9	P= .001
Index 2	20.6	21.1	NS
Index 3	34.5	33.2	NS
Index 4+	65.1	55.4	P= .000

Comparing the mean Olsen-P concentrations (in the 0-75 mm soil layer and the complete A horizon) across all sites in the first five-year period (2004-2008) with those in the second five-year period (2009-2013), revealed small but significant ($P \leq 0.02$) increases in soil Olsen P with time, in both the 0-75 mm soil layer (41.69 mg Olsen-P l^{-1} for 2009-2013 compared to 40.19 mg Olsen-P l^{-1} for 2004-2008) and the A horizon (39.19 mg Olsen-P l^{-1} for 2009-2013 compared to 35.06 mg Olsen-P l^{-1} for 2004-2008).

Although statistically significant, the increases are small in real terms and likely to be of minimal environmental significance. Analysis of the latest paired RSSS samples (100 sites, sampled in 2014-2015, compared with the same sites sampled in 2009-2010), indicate no significant change in soil Olsen-P status.

Subject to securing of funding, continuation of the sampling programme will allow further monitoring of soil P concentrations so any emerging trends can be identified and mitigation action taken if necessary. The RSSS, however, has been suspended until 2019/20.

10.2 E&I Project 9420 – UK Environmental Change Network: Freshwater

Key Findings from Long Term Ecological Research (LTER) project:

- Despite the lack of an overall trend in P catchment inputs Lough Neagh water concentrations remain high, likely driven by sediment P.
- River catchment inputs of N to Lough Neagh have decreased since the mid 1990's and were closely correlated with lake N concentrations.
- Dissolved organic N concentration (DIN) remains relatively low in Lough Neagh compared to values in the mid-1990s.
- The major drivers of primary production have changed across the time series. DIN has become more important in relation to phytoplankton biomass in recent years in Lough Neagh.
- The mass of P released from Lough Neagh sediment has increased since the mid-1990s.

Lough Neagh water concentrations of TP have increased since the 1990s and were not correlated with total catchment inputs (this was presented in last year's report). In contrast external inputs and lake concentrations of TON were highly correlated (R = 0.88) (also presented in last year's report).

The mass of nutrients from each of the major Lough Neagh subcatchments has been updated and is presented in this year's summary. The mass was calculated using the flow weighted mean concentrations of river nutrients. This takes into account the influence of flow on the loading from the river catchments. In most cases the river water nutrient loading time series covered the period from 1984-2018, however some rivers had different time series depending on the availability of river flow. Trends were examined across the entire period available and in some cases before and after 1995 to uncover any recent changes in trends.

Catchment loading to the lake was calculated for different fractions of N and P. Presented are TP, total soluble P (TSP), soluble reactive P (SRP) and particulate P (PP). Soluble reactive P is phosphorus that is available to organisms (bioavailable). Total P includes all P. We did not have data for total N for the whole timeseries, however dissolved inorganic N (DIN) provided indication of total N. Dissolved inorganic nitrogen was the sum of total oxidised N (TON) and ammonium.

Also presented are the trends in major water chemistry parameters for Lough Neagh: TP, DIN and chlorophyll-a for 1974 to 2016. Nutrient mass balance for the lake is presented for summer 1984–2014.

Data in the LTER Project for freshwater in Northern Ireland is continually quality assured and changes may be made to the data in future.

River inputs of N and P to Lough Neagh

Summary of total P (TP) loading to the Lough from each river catchment and TP outflow

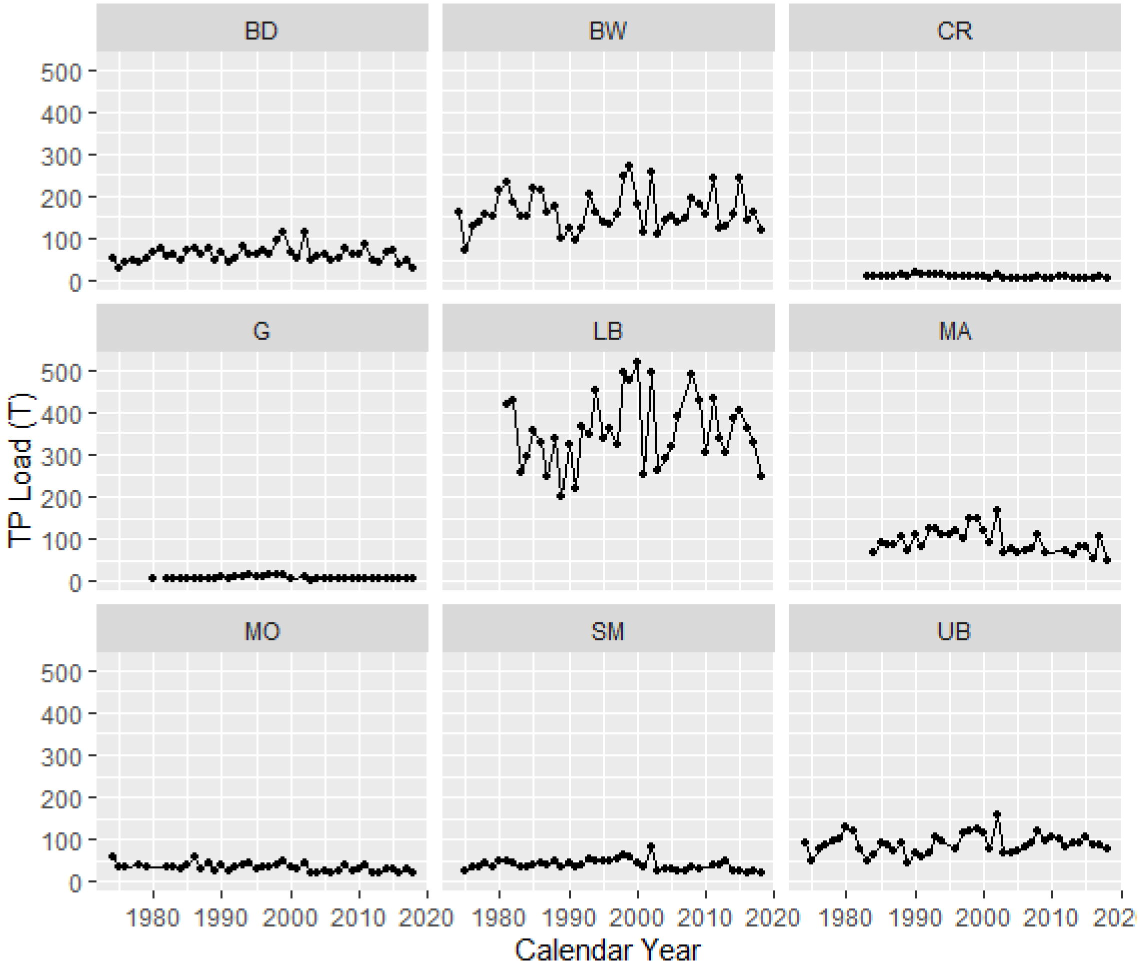


Figure 31: TP loading (T) of inflows and outflow of Lough Neagh 1974 – 2018. BD – Ballinderry; BW – Blackwater, CR – Crumlin, G – Glenavy, MA – Maine, MO – Moyola, SM – Six Mile, UB – Upper Bann. LB – Lower Bann (outflow).

Figure 31 shows the relative contribution of each catchment to Lough Neagh TP loading. The Blackwater was the main contributor over time of TP loading to the Lough. As the Lower Bann is the outflow it can be seen as representative of the lake.

Summary of soluble reactive P (SRP) loading to the Lough from each river catchment and SRP outflow

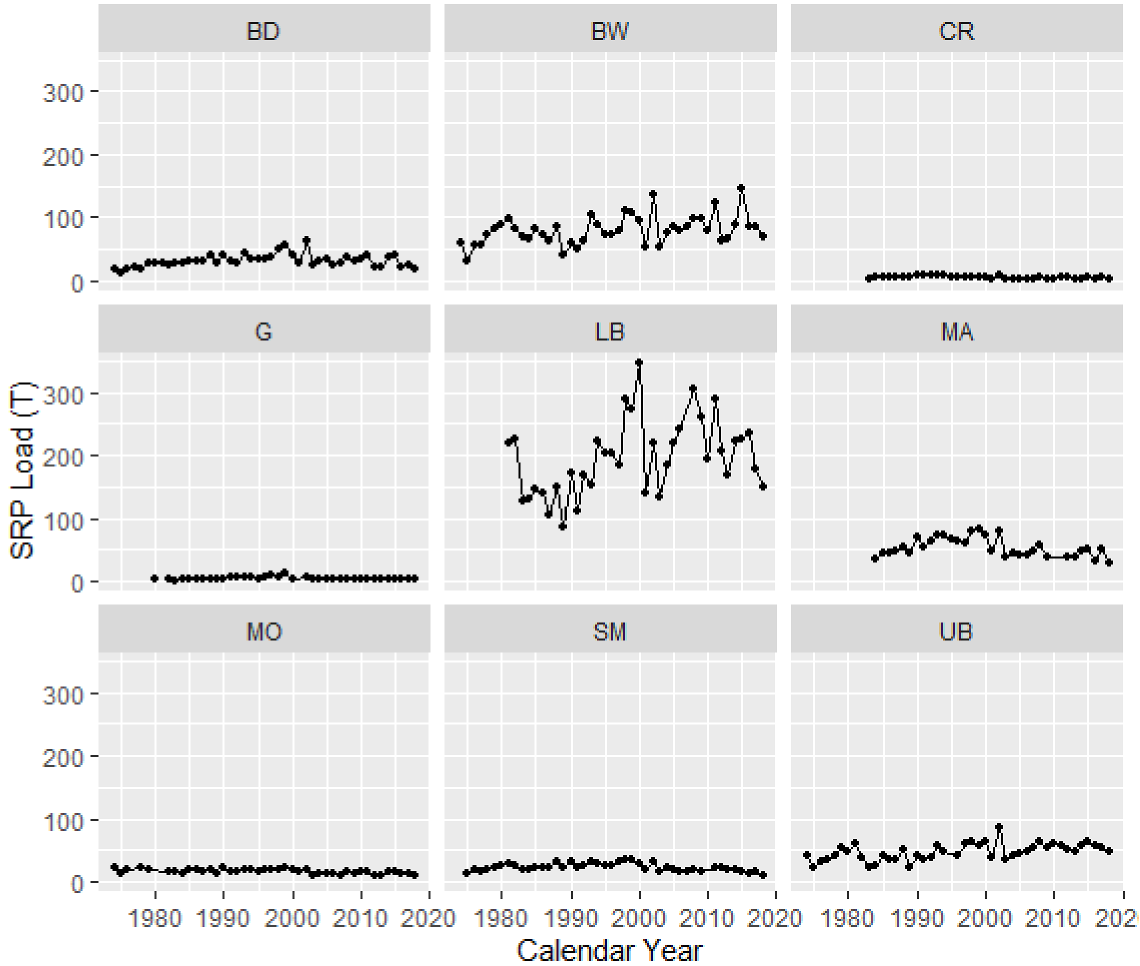


Figure 32: SRP loading (T) of inflows and outflow of Lough Neagh 1974 – 2018. BD – Ballinderry; BW – Blackwater, CR – Crumlin, G – Glenavy, MA – Maine, MO – Moyola, SM – Six Mile, UB – Upper Bann. LB – Lower Bann (outflow).

Figure 32: shows the relative contribution of each catchment to Lough Neagh soluble reactive P (SRP) loading. The Blackwater was the main contributor over time of SRP loading to the Lough. The SRP is the fraction of P that is most available to organisms.

Summary of particulate P (PP) loading to the Lough from each river catchment and SRP outflow

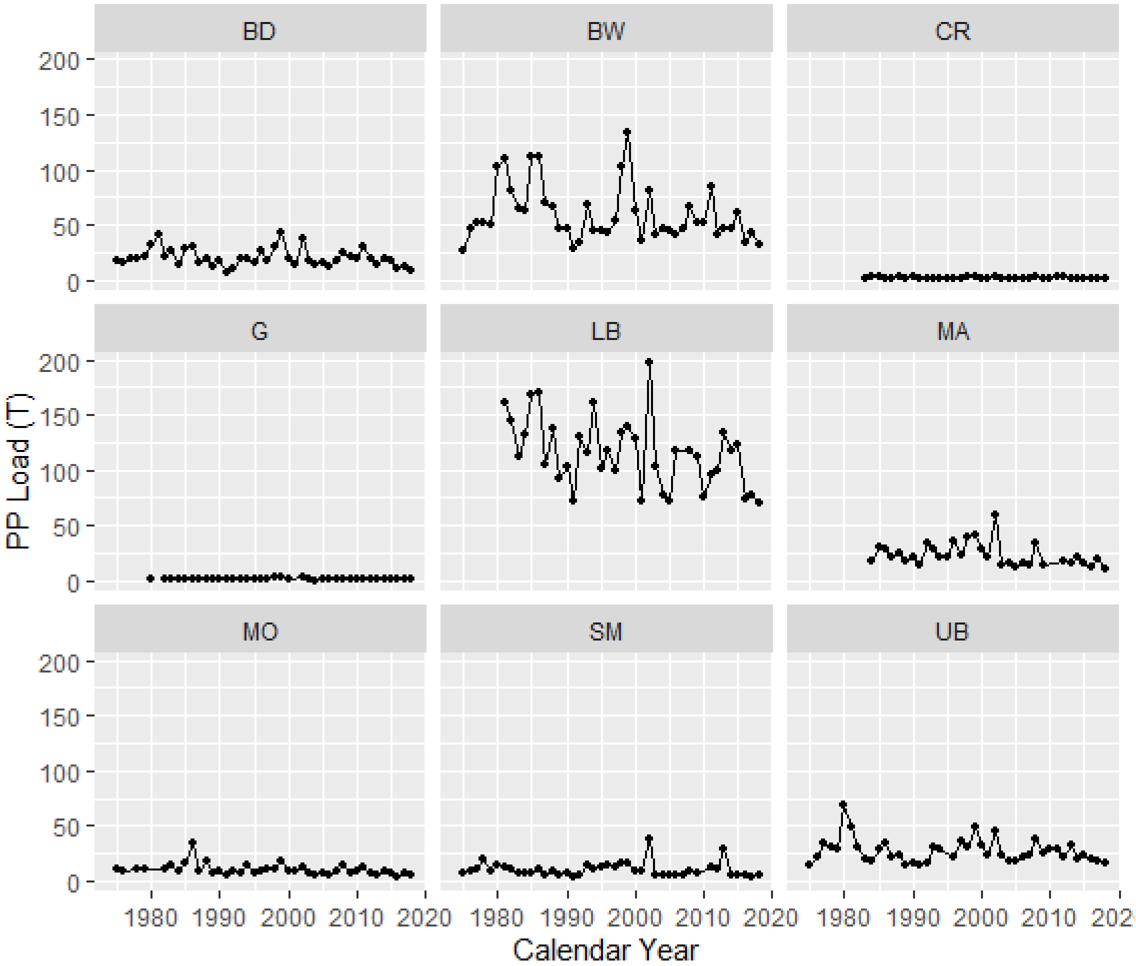


Figure 33: PP loading (T) of inflows and outflow of Lough Neagh 1974 – 2018. BD – Ballinderry; BW – Blackwater, CR – Crumlin, G – Glenavy, MA – Maine, MO – Moyola, SM – Six Mile, UB – Upper Bann. LB – Lower Bann (outflow).

Figure 33: shows the relative contribution of each catchment to Lough Neagh particulate P (PP) loading. The Blackwater was also the main contributor over time of PP loading to the Lough.

Summary of dissolved inorganic nitrogen (DIN) loading to the Lough from each river catchment and DIN outflow

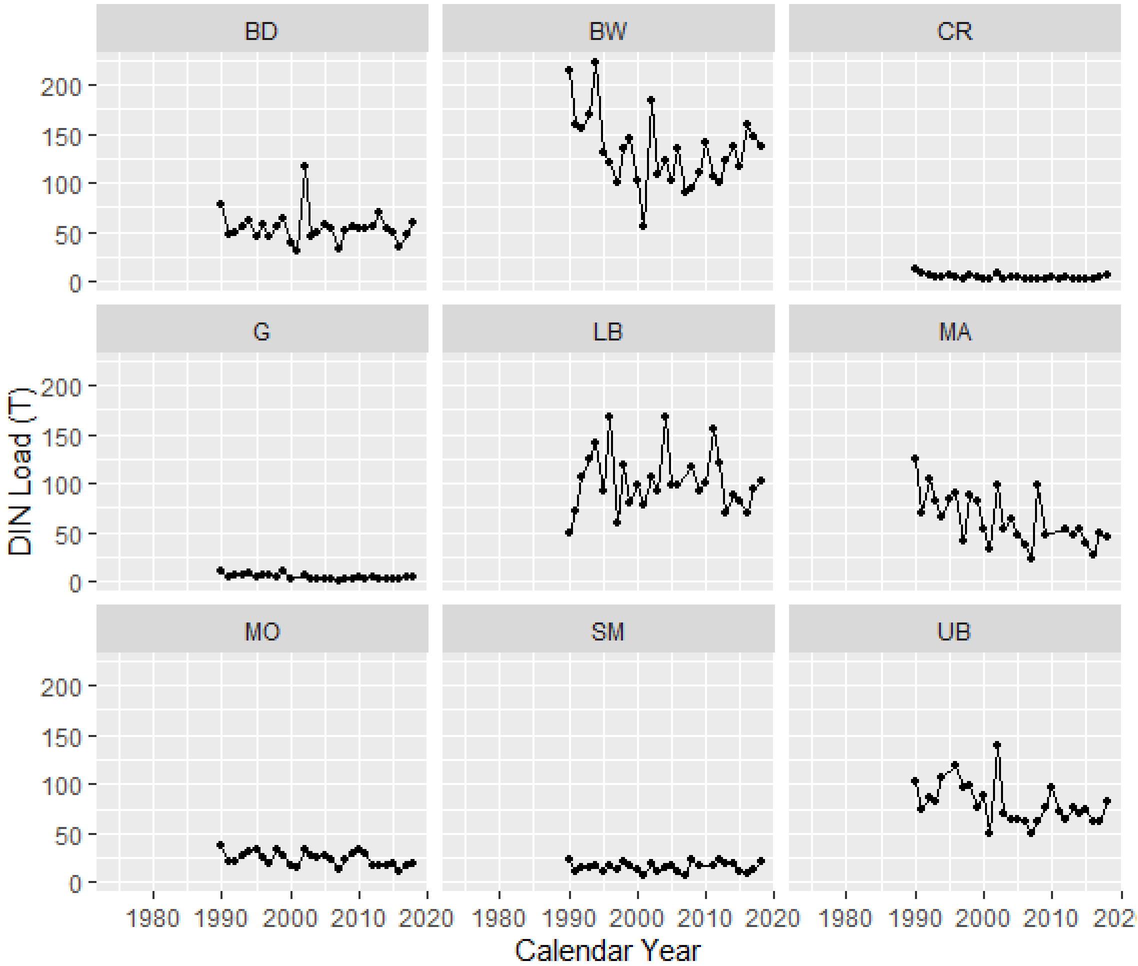


Figure 34: DIN loading (T) of inflows and outflow of Lough Neagh 1990 – 2018. BD – Ballinderry; BW – Blackwater, CR – Crumlin, G – Glenavy, MA – Maine, MO – Moyola, SM – Six Mile, UB – Upper Bann. LB – Lower Bann (outflow).

Figure 34 shows the relative contribution of each catchment to Lough Neagh DIN loading. The Blackwater was also the main contributor over time of DIN loading to the Lough.

Lough Neagh sub-catchment P and N loading and trends

The **Ballinderry** catchment covers an area of 434 km^2 which accounts for 6.5 % of the total study area. In the Ballinderry River catchment, 1974-2018, the only significant trend for P fractions was SRP which showed a significant increase (P < 0.01) across the entire time series available. The time series was also examined pre and post 1995 for trends. Before 1995 there was a significant increase in all P fractions (TP P < 0.05, TSP and SRP P < 0.001) except PP. After 1995 however we observed a reversed trend, with a decrease in TP, PP and SRP loading (all P < 0.05). There was no significant increase or decreasing trend in DIN 1990-2018. Before 1995 there were too few observations to carry out any statistical tests. After 1995 there was no trend.

The **Blackwater** catchment (nutrient monitoring area) covers an area of 1,390 km^2 which accounts for 21 % of the total study area. Across all years available (1974-2018) TSP and SRP showed increasing trends, *P* < 0.05 and 0.01 respectively. Before and after 1995 no changes were evident. No significant trends were evident for the entire time series available for DIN loading, 1990-2018. Not enough observations were available to detect patterns before 1995. After 1995 no differences were detected.

The **Crumlin** catchment (nutrient monitoring) is small and covers an area of 65 km^2 which accounts for approximately 1 % of the total study area. River nutrient loading from TP, TSP and SRP all decreased across all the years available for analysis, 1983-2018, (*P* < 0.001, 0.01 and 0.05 respectively). Before 1995 only SRP showed any trend, increasing (*P* < 0.01). After 1995 there was no significant trend. There was a decrease across the entire time series (1990-2018) for DIN loads (*P* < 0.05). There were too few observations before 1995 to draw any conclusions regarding trends and after this period there was no significant change.

The **Glenavy** catchment (nutrient monitoring) covers an area of 45 km^2 which accounts for approximately 1 % of the total study area. Across the entire time period available, 1980-2018, was no trend in P fractions. Before 1995 there was a highly significant increase in the mass of TP, TSP and SRP exported from the catchment (*P* < 0.001). Particulate P showed no trend. After 1995 only SRP showed any trend, decreasing (*P* < 0.05). DIN loading for the time period 1990-2018 showed a significant decrease (*P* < 0.01). Before 1995 there were too few observations to draw any conclusions and no trend was apparent after this period.

The **Maine** catchment (nutrient monitoring) covers an area of 704 km^2 which accounts for 11 % of the total study area. The majority of the P fractions showed no change across the entire time period (1984-2018), however, PP loading decreased (*P* < 0.05). Before 1995 the mass of TP, TSP and SRP increased (*P* < 0.05 and *P* < 0.01 respectively). Alter 1995 all P fractions decreased, TSP and SRP had the most significant changes (*P* < 0.01). Across all years (1990-2018) there was a significant decrease in DIN load (P < 0.01). Before 1995 there were too few observations and after 1995 there were no changes in DIN loading trends.

The **Moyola** catchment (nutrient monitoring) covers an area of 308 km^2 which accounts for 5 % of the total study area. Across the time period 1974-2018 the mass of all four fractions of P decreased, with TP, SRP and PP having the most significant change (*P* < 0.001). No changes were apparent before 1995, all fractions decreased after 1995 (*P* < 0.05). The mass of DIN exported from the Moyola catchment decreased from 1990 to 2018 (*P* < 0.05). Before 1995 there were too few observations and no trend was apparent in loading after 1995.

The **Six-Mile** catchment covers an area of 301 km^2 which accounts for 5 % of the total study area. Across the entire time period available (1975-2018) there was no significant trend in the mass of P fraction loading. Before 1995 TP and the soluble fractions of P (TSP and SRP) demonstrated significant increases in loading (*P* < 0.05, *P* < 0.01, *P* < 0.001) respectively. After 1995 all fractions of P decreased highly significantly; TP and PP both *P*

< 0.01; SRP and TSP, $P < 0.001$. No significant trend in DIN loading was apparent across the entire time series (1990-2018). Before 1995 there were too few observations, after 1995 there was no significant trend in loading.

Figure 34 shows the relative contribution of each catchment to Lough Neagh dissolved inorganic N (DIN) loading. As the Lower Bann is the outflow it can be seen as representative of the lake. The Blackwater catchment is the main contributor over time of DIN loading to the Lough. DIN is bioavailable N.

Lough Neagh nutrient trends

The concentration of TP has increased in the lake, a large change occurred in the mid-1990s and concentrations have generally stayed relatively high since then (Figure 35).

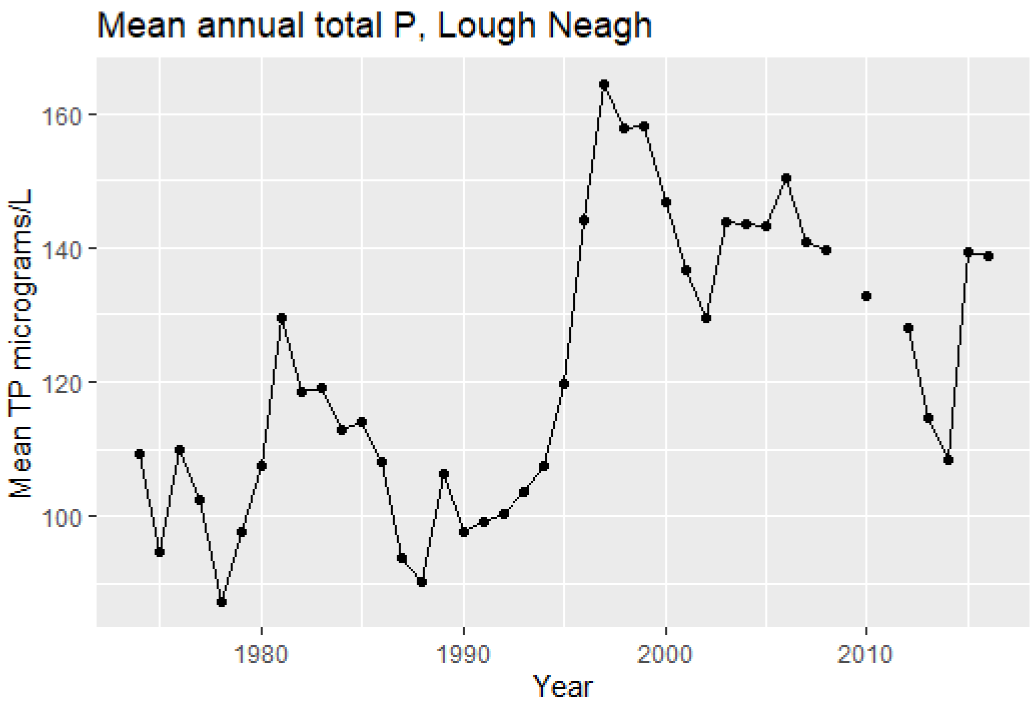


Figure 35: Trends in lake water total P (TP µg/L) in Lough Neagh 1974-2016.

In contrast DIN concentrations in the lake have decreased since the mid-1990s and remained relatively low (Figure 36).

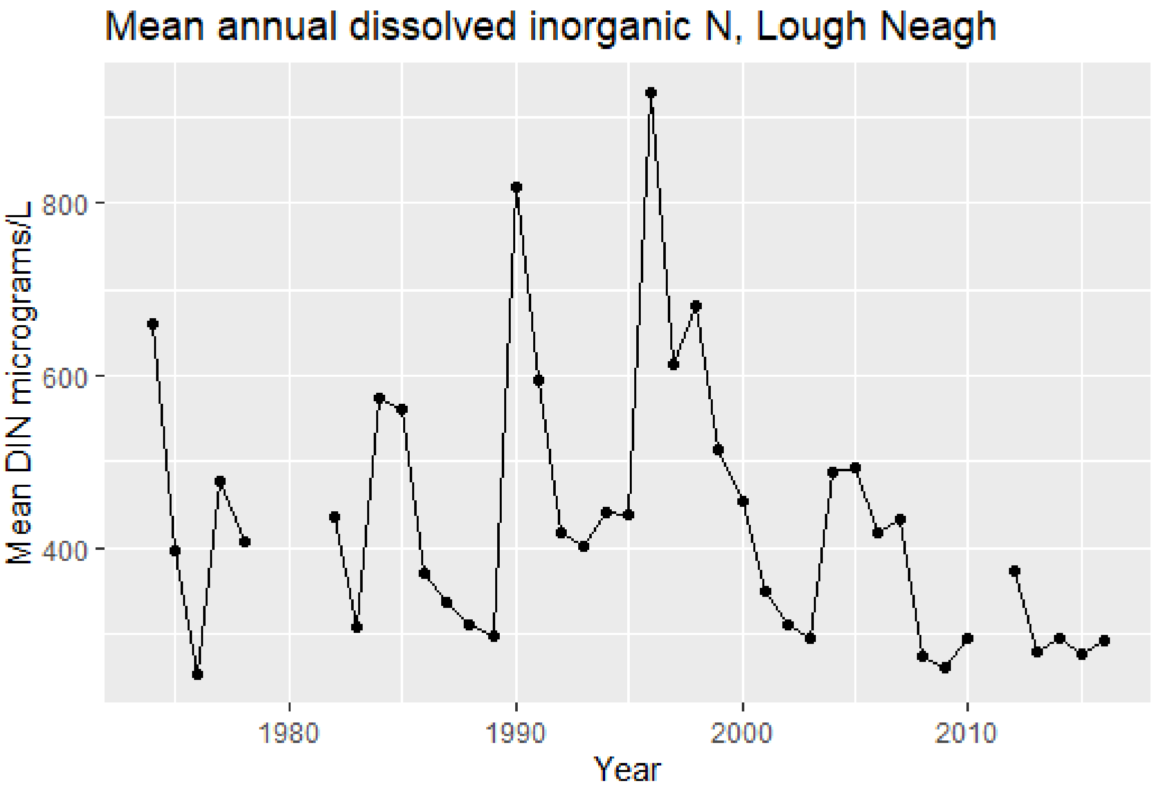


Figure 36: Trends in lake water dissolved inorganic N (DIN µg/L) in Lough Neagh 1974-2016.

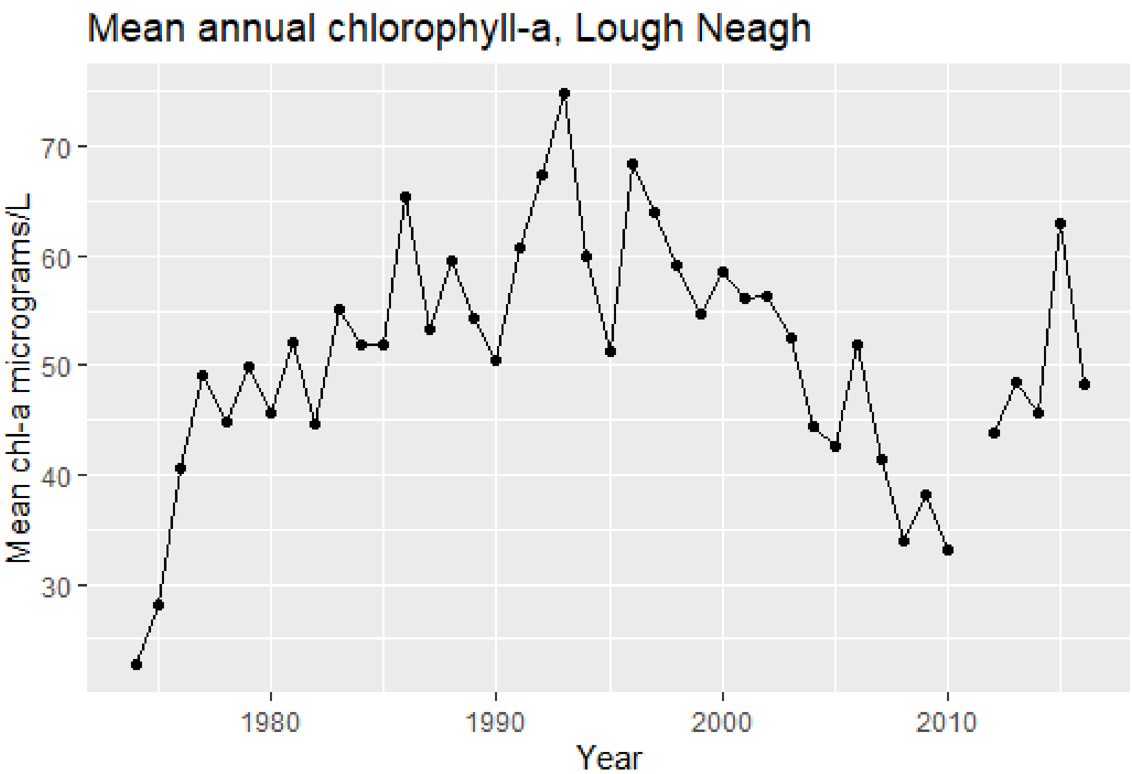


Figure 37: Trends in lake water chlorophyll-a (chl-a µg/L) in Lough Neagh 1974-2016.

Chlorophyll-*a* has also generally decreased since the mid-1990s despite high levels of available P (SRP) (Figure 37). After the peak chl-*a* concentration in 1993, dissolved inorganic N (DIN) became a more important predictor of chl-*a*, accounting for half of the 42 % variance explained by a hierarchical partition model (1994-2012). This, together with decreasing DIN:TP ratios, suggests that N control of chl-*a* has become more important in the lake recently. The changing ratio was brought about by increasing P and decreasing N water concentrations in the lake water.

Lough Neagh mass balance of nutrients

The mass balance of nutrients provides information on the amount of nutrients stored by or released from the lake. When the lake sediment releases bioavailable P to the water column it is termed "internal loading". The characteristics of internal loading of P have changed since the mid-1990s, with a larger mass of P released from the sediments each summer (Figure 38). From 1984 to 1994 the mean summer release of TP was 107 T, from 1996 to 2014 the mean TP release increased to 249 T. The timing of the release of sediment P has also changed with the release happening earlier in the year as we move through the time series to more recent times.

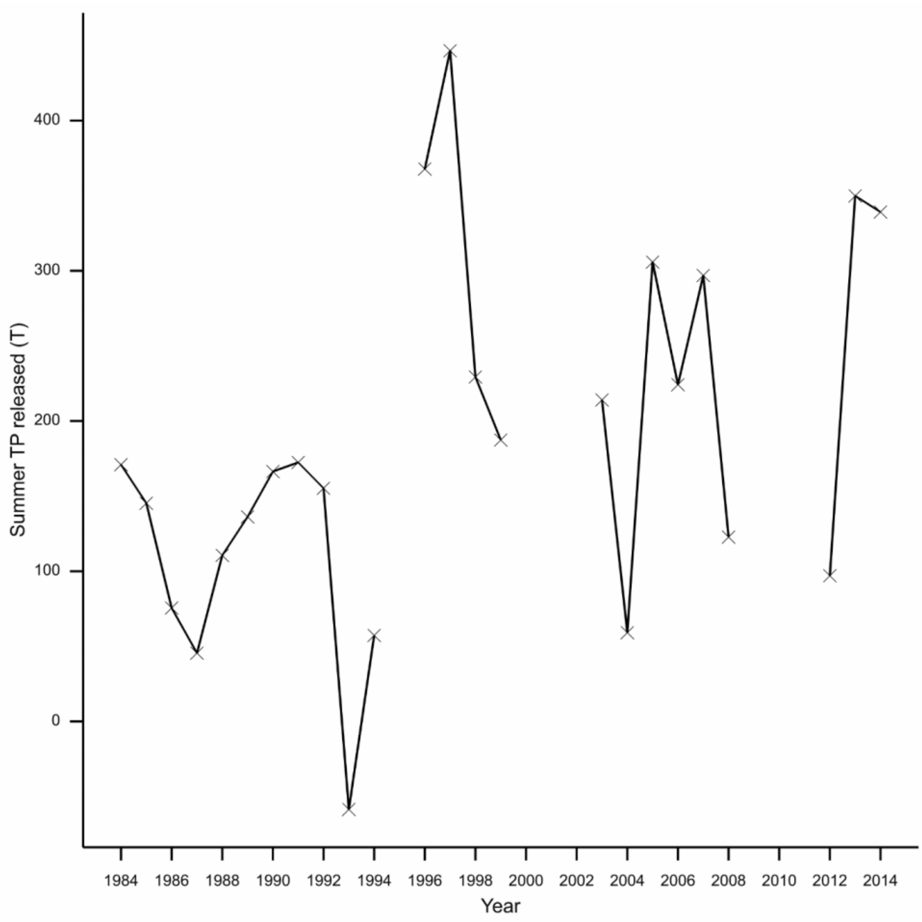


Figure 38: Net release of total P (T) for summer (June, July, August) 1984–2014.

10.3 E&I Project 16/4/03 – Monitoring, modelling and mitigation of N and P losses from land to water under derogated and non-derogated conditions in the Colebrooke and Upper Bann Catchments (New Research)

The Overall Objectives of the monitoring/modelling programme are to provide scientific evidence to:-

1. Meet the additional/amplified monitoring and reporting requirements of Articles 8 and 10 of the 2015 Derogation Decision.
2. Identify and validate strategies which minimise N and P losses to water, optimise farm productivity and reduce variable costs on ruminant livestock farms.

These Objectives are being addressed in four Work Packages (**WPs**):-

WP 1 – Surface Water Sampling

In compliance with Article 8(3) of the 2015 Derogation Decision, monitoring of chemical water quality (nutrients) in streams in 12 sub-catchments in each of the Upper Bann (UB) and Colebrooke (CB) catchments, which ceased in 2014, will be reinstated at bi-monthly frequencies. This will be supplemented with targeted storm flow event sampling to improve resolution and precision of sampling - c.f. the 2009-2014 monitoring program.

Bi-monthly and seasonal hydrological event sampling for nutrients will also be conducted in two contrasting sub-catchments (*one containing* a significant proportion of derogated *farmland and the other containing only non-derogated farmland*) within the UB catchment. Samples will also be analysed for a range of tracers to help determine the contributions of rural septic tank outflows to P loads.

WP 2 – Groundwater Sampling

Groundwater monitoring is a requirement under Article 8 of the 2015 Derogation Decision. A baseline survey within the pair of UB sub-catchments (*but if necessary extended to include other sub-catchments*) will be undertaken to identify, in consultation with landowners, wells and springs suitable for sampling. Potential threats to water quality in each well/spring will be noted during sampling. After an initial monitoring round, a sub-set of wells/springs will be identified and monitored annually.

WP 3 – Soil Sampling, Farm Data Collection & Nutrient Management Advice

To provide relevant soil P data for modelling P losses from derogated and non-derogated farms on the main soil types of Northern Ireland in accordance with Article 8(2) 2015 Derogation Decision, and information on farming practices etc. on derogated and non-derogated farms in accordance with Article 8(4) of the 2015 Derogation Decision, a sampling/data collection scheme will be conducted.

In the pair of UB sub-catchments (*one sub-catchment with a significant proportion of derogated farmland and the other with no derogated farmland*), all fields (*713 in total*) will be soil tested for Olsen-P and Calcium Chloride extractable P ($CaCl_2$-P). A 1 m resolution Light Detection And Ranging (LiDAR) digital terrain model (DTM) will be applied to help identify potential Critical Source Areas (CSAs) for P loss within sub-catchments.

For whole farms in each sub-catchment (*33 in total – 10 dairy and 23 Beef & Sheep*), information and data on nutrient imports and exports will be collected to calculate annual farm N and P surpluses, and concentrate feeds, silages and manures analysed.

For the 10 dairy farms, in addition to this, records of fertiliser and manure application to fields will be maintained annually, and twice yearly, samples of manures and concentrate feeds will be collected and analysed for N and P to help quantify nutrient cycling and flows within these farming systems.

WP 4 – Modelling of Nutrient Losses from Farmland

In compliance with Article 10 of 2015 Derogation Decision, nitrate and P losses from derogated farmland will be modelled using soil-P and farm-gate N and P balance information together with the results of chemical water quality monitoring in the sub-catchments using a 'source-pathway-receptor' approach.

An export coefficient modelling approach will be applied to estimate diffuse losses of P and N from individual farms. Soil P distribution data for farms of different intensities will then be used together with known breakdowns of farming intensities within catchments to scale-up model estimates of P loss to catchment scale.

Catchment scale modelling will be attempted to model N and P contributions from multiple sources and to assess the overall impact of derogation on nutrient losses from farmland across Northern Ireland.

Results

The results to date for WP1 and WP2 are reported in Section 5 of this report, those for WP4 in Section 7 and some for WP3 are reported in Section 4. Additional results under WP3 are reported below:

Evidence of P mismanagement on different farm types and land classes - based on soil P data collected in the EAA-Soil Sampling and Analysis Scheme

In 2017 AFBI launched a free Soil Sampling and Analysis Scheme open to all livestock farmers in Northern Ireland, and funded under the EU's Exceptional Adjustment Aid (EAA) Package. The impetus for the scheme was the general lack of soil testing on grassland farms across Northern Ireland, and the need to improve, in particular, phosphorus (P) use efficiency in order to enhance farm profitability and simultaneously meet targets for water quality improvement set under the Nitrates Directive and the Water Framework Directive. This was the largest and most intensive soil testing scheme ever carried out in Northern Ireland, and probably the largest and most detailed ever carried out in the British Isles. The results of this scheme in relation to soil P are particularly relevant to this project and therefore reported here.

The scheme had two components:

- **An 'Open Scheme'** – open to all livestock farmers across Northern Ireland, with successful applications from 520 farms covering 12,000 fields (Figure 39).
- **A 'Catchment Scheme'** – targeted at livestock farmers within 11 sub-catchments of the Upper Bann catchment, with successful applications from 519 farms covering 8,000 fields (Figure 39)

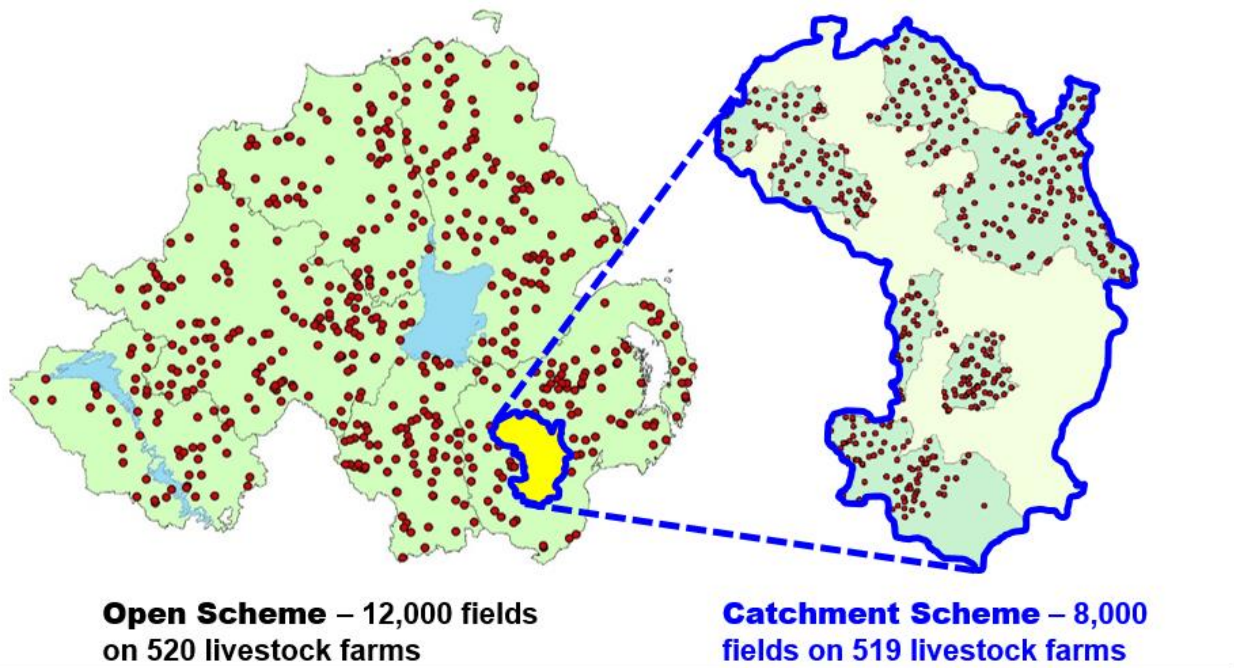


Figure 39. Map of Northern Ireland showing the location of 520 farms in the Open Scheme and 519 farms in the Catchment Scheme in 11 Sub-catchments of the Upper Bann River Catchment

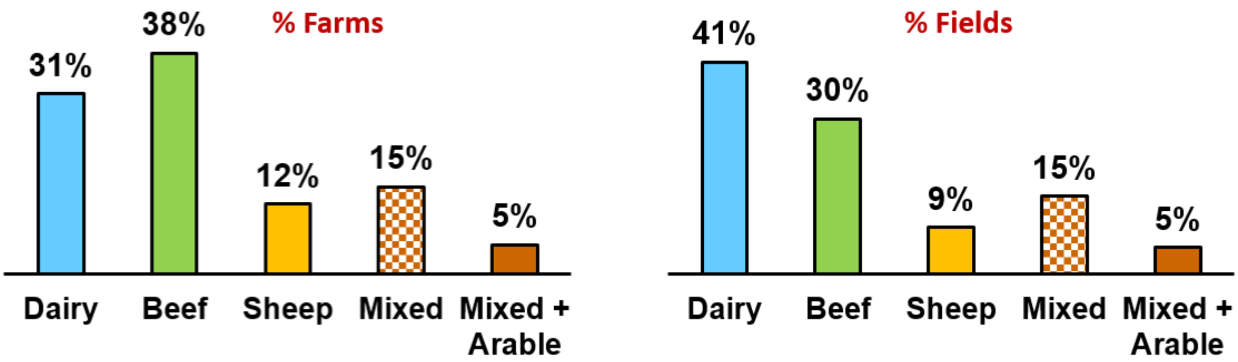


Figure 40. Percentages of farms and fields in different farm types in the EAA Open Scheme

The Open Scheme has provided, previously unavailable breakdowns of soil P, K and pH statuses for the three main grass-based livestock sectors (*based on sole or predominant type of livestock present on farms*) across the three main classes of farmland – as defined below.

- **Lowland** (non-LFA) – this is all land which is classified as neither disadvantaged nor severely disadvantaged under the directives above.
- **Disadvantaged** (DA) – subset of LFA – these are areas where agricultural production is constrained by socio-economic and environmental factors (*shorter growing season, higher rainfall, steep slopes, low productivity, depopulation etc.*) compared to non-LFA.
- **Severely Disadvantaged** (SDA) subset of LFA – these areas are generally 'upland' in nature and the constraints to agriculture identified above operate at a more challenging level than in DA area.

The breakdowns of soil P indices across farm types and land classes are given in Figure 41. As expected, the national dairy grassland platform was found to have the most acute P over-supply problem, with around 50% of fields at soil P indices greater than 2 on all classes of farmland. But the beef and sheep platforms had also significant P over-supply problems (*on an area 3 times larger than used for dairy*), with around 40% of fields oversupplied in both Lowland and Disadvantaged areas, and > 30% over-supplied in Severely Disadvantaged areas. Clearly, in the absence of soil testing, there has been poor distribution of P on all 3 types of livestock farm and on all classes of farmland. But this alone cannot be responsible for the significant percentages of fields farmed for beef and sheep which are at Index 3 and above. Clearly, unwarranted use is being made of chemical P fertilisers by beef and sheep farmers – and indeed anecdotal evidence indicates that many such farmers continue to use fertilisers such as 20:10:10. In a number of farms in Upper Bann where full farm P balance data were available, on average it was found that 20% of total P inputs to land as fertiliser and manure, were in the form of chemical P, compared with just 2% on intensive dairy farms

As regards farmland under-supplied with P, less than 20% of fields on dairy farms are at sub-optimal P status (Index 0 and 1), whereas close to 30% of fields on beef and sheep enterprises were potentially sub-optimal in P. With soil tests now available for the 1039 farms participating in the EAA Schemes, and with the likelihood of more farmers investing in soil testing going forward, there is a risk that fertiliser P may be imported to correct ‘apparent’ P deficits on some 40% of farmed grassland across all livestock enterprises, whether there is a need for it or not. Clearly, in situations where grassland is farmed intensively, and where manure-P resources on farm are insufficient to meet RB209 crop P requirements for intensively managed swards, there would be justification for importing chemical or manure-P to address the P insufficiency. However, on many beef and sheep farms, swards are managed extensively, and therefore use of RB209 recommendations, which are designed for intensively managed swards, could result in excessive and unnecessary applications of P on land where P may not actually be limiting to grass production. In other words, the EAA Scheme and the subsequent 2018/19 Colebrook and Strule Soil Testing and Training Scheme (CSSTTI), could produce a ‘perverse’ out-come: namely an undesirable and unnecessary increase in chemical or imported manure-P usage on land close to streams and rivers where water quality may currently be good. Accordingly, a provisional set of P recommendations for grassland that is managed extensively is now in place.

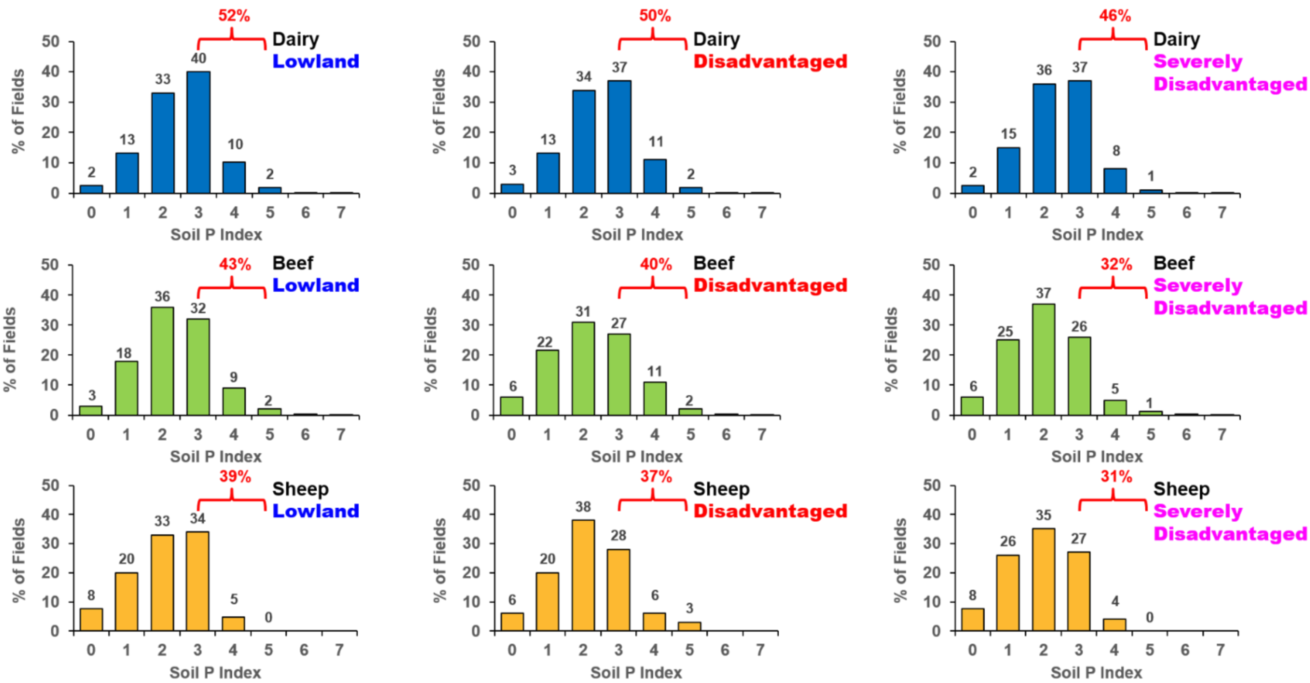


Figure 41. Percentages of fields on different types of livestock farms and on different classes of agricultural land with different soil P indices (Open Scheme).

Conclusions

1. The EAA and CSSTTI soil testing schemes, have been beneficial in that they appear to have finally stimulated an interest in soil testing by grassland farmers by making them more aware of the inherent benefits and dividends that knowledge of soil nutrient status can deliver.

2. The schemes, however, have provided definite evidence of P mismanagement and over-use, not just within the intensive dairy sector (*as had been expected*), but surprisingly in the beef and sheep sectors, and even more disturbingly, on the most severely disadvantaged farmland – where more pristine aquatic environments would be expected.
3. The increased interest in soil testing by grassland farmers, could give rise to a 'perverse' outcome at least on extensively managed grassland platforms, should current RB209 P recommendations be applied to such land, since the recommendations were not designed for extensive grassland systems. Instead, new P recommendations are needed for such systems, to prevent the unwarranted and undesirable use of chemical or imported manure P on land where P may not to be limiting to grass production, even when soil P levels are relatively low (P Index 2^- or I).

Revised P recommendations for extensively managed grassland

Intuitively, it would be expected that grassland managed 'extensively' with relatively low N inputs, should have lower P requirements and a lower target soil P test level than grassland managed 'intensively' with high N inputs driving high levels of grass production and P removal. Currently, if extensively managed grassland is soil tested and found to have P indices < 2^+, farmers are permitted to fertilise their swards using P recommendations given in NAP guidelines, which were designed for intensively managed swards, thus potentially wasting money and increasing the risk of P loss to water. To avoid this 'perverse' outcome, new bespoke P recommendations are required for low-input extensively managed grassland systems.

The first step in developing new recommendations was to define what is meant by 'extensively managed grassland'. In the current project farm nutrient input/output and herbage nutrient and potential DM yield data (*based on total available N applied to grassland and predicted using yield models developed from field trials at AFBI Hillsborough*) were available for a range of low and high input ruminant (*dairy, beef, beef and sheep, and sheep*) systems in Upper Bann, and these data were scrutinized to see if grassland management could be defined as 'extensive' or 'intensive' based on amounts of N applied (*fertiliser & manure-N*). As shown in Table 21, it was possible to categorise farms based on N fertiliser usage.

Table 21: Farm areas, manure and fertiliser N loadings, predicted annual DM yields, herbage protein and DRIS N and DRIS P indices at 1st cut, and soil P Indices.

Farm Type/ID	Area (ha)	Man' N (kg/ha)	Fert' N (kg/ha)	DM yield (t/ha)	Herbage protein %	Herb' DRIS N	Herb' DRIS P	Soil P Index
'Extensively Managed Grassland Farms' (< 60 kg Fertiliser N/ha/year)								
Beef/1	32	55	14	4.3	8.5	-4	17	2^-
Beef/31	13	88	58	6.0	-	-	-	3
B&S/17	7	56	41	4.7	-	-	-	3
B&S/14	16	87	25	4.8	11.8	6	31	4
B&S/23	32	113	4	4.8	9.9	3	26	3

B&S/30	51	71	19	4.5	11.8	8	21	3
Sheep/24	6	33	48	5.2	-	-	-	1
Dairy/16	28	167	54	9.9	10.7	0	14	3
MEAN	**23**	**83**	**33**	**5.5**	**10.7**	**3**	**22**	**3**
'Intensively Managed Grassland Farms' (> 60 kg Fertiliser N/ha/year)								
B&S/16	43	100	95	10.6	12.5	18	21	3
Dairy/35	62	143	336	14.1	17.4	26	20	3
Dairy/11	49	202	124	11.9	12.1	7	13	2^+
Dairy/25	88	237	215	13.5	15.9	14	10	3
Dairy/9	74	241	116	11.9	15.6	13	18	3
Dairy/27	32	216	158	12.6	14.7	8	19	4
Dairy/5	64	196	142	12.2	12.6	10	25	4
MEAN	**59**	**191**	**169**	**12.4**	**14.4**	**14**	**18**	**3**

A chemical N rate of 60 kg N/ha/year was selected as an appropriate cut-off point to differentiate between grassland farms managed 'extensively' and receiving < 60 kg fertiliser N/ha/year, from those managed 'intensively' and receiving ≥ 60 kg fertiliser N/ha/year. The rationale for choosing this rate of fertiliser N application as the break-point between these two grassland management categories is based on the nutrient status of herbage. Herbage protein and DRIS N and P data for 1st cut silage crops were available (*on most farms*) to indicated the degree to which N and P may have been limiting to sward DM production - DRIS indices less than 5 being indicative of nutrient shortage, those above 15 being indicative of nutrient surplus, and indices between 6 and 15 indicative of optimum nutrient supply (Bailey *et al*., 2014).

On average, for the 7 farms categorised as 'intensively managed', herbage protein concentrations and DRIS indices clearly show that both N and P were present in optimal supply (*with average sward protein content of 14.4%, and average DRIS N and DRIS P of 14 and 18, respectively*) and capable of supporting high levels of DM production (*average potential DM yield of 12.4 t DM/ha*). In contrast, for the 8 farms categorised as 'extensively managed', herbage protein concentrations and DRIS indices demonstrate that N was in sub-optimal supply (*with average protein concentration at 10.7% and average herbage DRIS N at just 3*), but that P was abundantly available (*with average herbage DRIS P at 22*). In other words, for this group of farms, N and not P, was limiting to grass production and had been responsible for in low DM yield potential (*mean potential DM yield just 5.5 t DM/ha*). Accordingly, there appears to be little justification for applying rates of P designed to support DM production in excess of 10 t DM/ha/year, and maintain soil P at Index 2^+, when such levels of production will never be achieved with the amounts of N applied to these 'extensively managed' grassland farms.

Having defined 'extensively managed grassland farms', the next step was to set a target soil P index which the new P recommendations would be designed to achieve. This was more difficult, as the soil P indices on the 'extensively managed farms', were nearly all > 2^-. Moreover, the only farm where the P index was < 2^-, was a sheep farm where silage was not produced and hence herbage nutrient data were not available. Taking a precautionary approach, therefore, it was decided that a soil P index of 2^- (16-20 mg

Olsen-P/1) would be an appropriate 'provisional' target index for extensively managed grassland. This target may be revised once results of a new series of field phosphorus trials on low input beef and sheep farms in the Blackwater Catchment become available over the next two years in the Catchment Care Project.

Having decided on a target P index of 2^- for extensively managed grassland farms, the final step was to decide on the amounts of P that needed to be applied to different grassland crops on soils of different soil P indices, to bring soil P status into this target index 2^- range. Results from the afore-mentioned field experiments would have been very useful for this purpose, but in their absence, provisional recommendations were assigned simply by taking the existing P recommendations for intensively managed grassland (*which are designed to bring soil P status into the index 2^+ range*), and move them one P index category to the left to provide the recommendations shown in Table 22.

It was recognised, though, that even on extensively managed farms where average fertiliser N usage is less than 60 kg N/ha/year, fields set aside for silage or hay, may sometimes receive more than 80 kg N/ha (*albeit average N application across the farmed grassland platform remains < 60 kg N/ha/year*). To ensure that crops fertilised with > 80 kg fertiliser N/ha do not suffer from P insufficiency, P recommendations for 'intensively managed' grassland will be permitted for such crops, as shown in Table 22a.

Table 22. Maximum phosphate fertiliser application limits (kg P_2O_5 per ha) for extensively managed grassland receiving < 60 kg chemical fertiliser N/ha/year

	Soil P Index					
	0	**1**	**2-**	**2+**	**3**	**4**
Grass establishment	80	65	50	30	0	0
Grazed grass (whole season)	50	35	20	0	0	0
First cut silage†	70	55	40	0	0	0
Hay†	55	43	30	0	0	0

† *Subject to Table 22a*

Table 22a: If extensively managed grassland, but silage or hay crops receive > 80 kg chemical fertiliser N/ha/year, the following maximum phosphate fertiliser application limits apply

	Soil P Index					
	0	**1**	**2-**	**2+**	**3**	**4**
First cut silage	100	70	55	40	0	0
Hay	80	55	43	30	0	0

Conclusions

1. Grassland farms receiving (*on average*) ≥ 60 kg N/ha/year as fertiliser N are categorized as 'intensively managed', whereas grassland farms receiving < 60 kg N/ha/year as fertiliser N are categorized as 'extensively managed' and on such farms grass production is limited by N deficiency rather than inadequate P supply.
2. It is proposed that for grassland managed extensively (< 60 kg fertiliser N/ha/year or < 120 kg organic N/ha/year) and supporting grazing and one cut of silage or hay

per season, the target soil P index should be 2-, and (provisional) P recommendations are simply those for ‘intensively managed’ grassland crops shifted one P index category to the left (note, field trials are on-going to validate and if necessary refine these provisional recommendations). But to ensure that silage and hay crops which may receive > 80 kg fertiliser N/ha (albeit < 60 kg fertiliser N/ha/year is applied on average across the complete farm grassland platform) do not become P - limited, P recommendations for ‘intensively managed’ grassland crops will be permitted on such crops.

Optimum farm gate P balances for different types of ruminant enterprise

Annual farm-gate P surpluses or balances provide a ‘snap-shot’ in time of the differences between total P inputs and total P outputs at whole farm scale. In due course these surpluses pass into animal excreta and from there into soil, with a small proportion deposited on laneways etc. However, linking soil P status to annual assessments of farm P surplus is problematic because the former is the product not just of one year, but of many years of differential P management, and is dependent on the length of time a particular P surplus has been maintained, or the extent to which it has increased or decreased over time. Nevertheless, because farm P surplus is being regulated under Article 5(8) of the European Commission (EC) Decision 2007/863/EC, with an enforced maximum limit of 10 kg P ha^{-1} yr^{-1} for derogated farms, and because this limit might need to be extended to all grassland farms to help counteract a recent deterioration in water quality, it is important to assess the impact that such restrictions might have on soil P status and hence on grass-based ruminant production in Northern Ireland. To this end relationship(s) between farm P surplus and soil Olsen-P status on different types of ruminant livestock enterprises were evaluated, to determine what level(s) of P surplus are needed to maintain Olsen-P within the target Index 2^+ range (21-25 mg P/l) for intensively managed grassland, and within the target Index 2^- range (16-20 mg P/l) for extensively managed grassland.

Suitable farm-gate P balance and soil P test data were available for only a small selection of dairy, beef, and beef & sheep (B&S) farms in Upper Bann (UB), since a number of farms in the catchment (*for which data were also available*) had made recent changes to P management which had altered their farm P surpluses appreciably such that soil P status, which takes several years to respond to changes in P surpluses, could no longer be considered as being reflective of current P management. To compensate for the small number of suitable farms in UB, data from dairy farms in the 2009-2012 Dairyman INTERREG Project were also included.

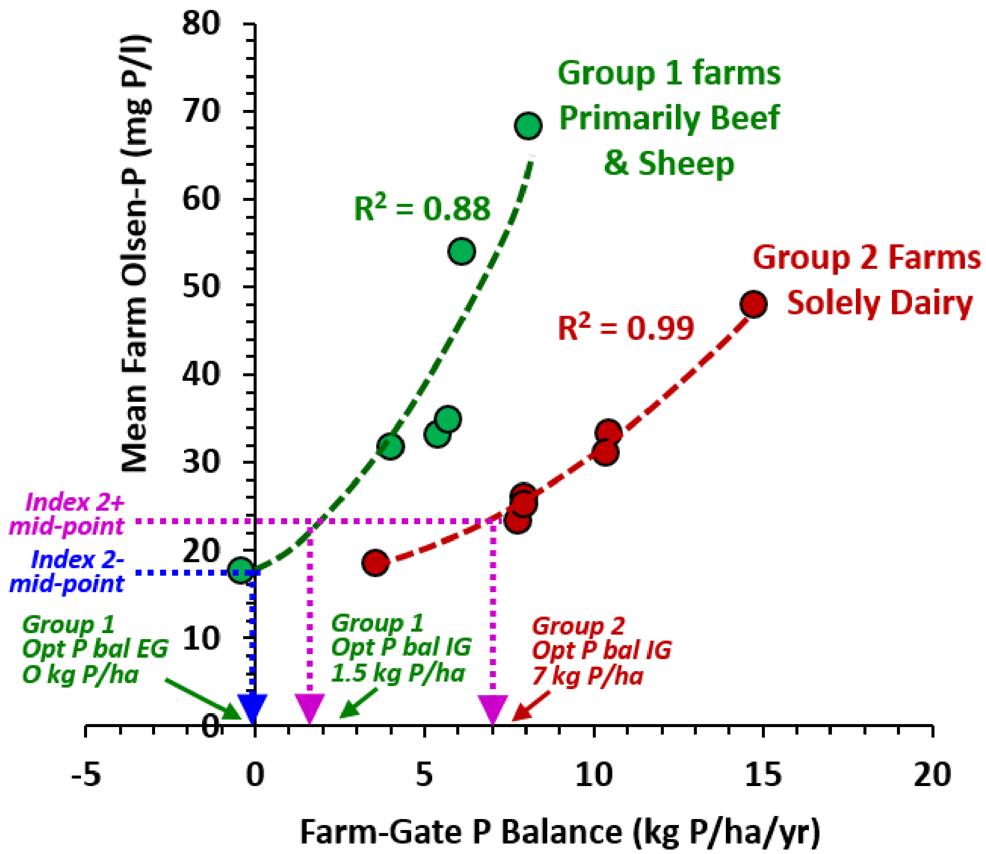


Figure 42: Mean farm soil Olsen-P versus farm-gate P balance with fitted regression curves distinguishing two groups of farms, and optimum P balances to achieve the mid-point of the soil Olsen-P index 2^- and 2^+ ranges

When mean farm soil Olsen-P was plotted against farm-gate P balance, two distinct relationships emerged distinguishing two groups of farms (Figure 42). **Group 2** was comprised entirely of intensive dairy farms, whereas **Group 1** consisted of extensive beef and B&S farms, plus an extensive dairy farm and an intensive dairy farm (Table 23). From the relationships in Figure 42, three farm-gate P balance optima were identified. Optima of 7 kg P/ha/year and 1.5 kg P/ha/year, were necessary to maintain soil Olsen P at the centre of the Index 2^+ range (*the target for intensively managed grassland - **IG***) on **Group 2** and **Group 1** farms, respectively, whereas an optimum of 0 kg P/ha/year was sufficient to maintain Olsen-P at the centre of the Index 2^- range (*the target for extensively managed grassland -**EG***) on **Group 1** farms.

The question then is: what is the difference between these two groups of farms that has resulted in soil Olsen-P responding differently to changes in farm-gate P surplus? Table 23 indicates that **Group 2** farms (*solely dairy*) have generally higher rates of fertiliser N usage and manure-N loading (stocking densities) than **Group 1** farms (*primarily beef and B&S*), but with one exception: UB 27, an intensive dairy farm in **Group 1** with N fertiliser usage and a manure N loading equivalent to those of farms in **Group 2**. Hence, these variables are not the key factors discriminating between the 2 groups of farms (Figure 42).

Table 23. Soil and characteristic nutrient (P and N) data two distinct groupings of farms

Farm Type	ID	Area (ha)	Soil P (mg P/l)	P Balance (kg/ha)	P inputs (kg/ha)	Fert P (kg P/ha)	Man N (kg N/ha)	Fert N (kg N/ha)
Ext' Beef	UB 1	32	18	-0.4	0.6	0.0	55	14
Ext, Beef	UB 31	13	33	5.4	12.7	5.1	88	58
Ext' B&S	UB 30	51	32	4.0	5.4	1.0	55	19
Ext' B&S	UB 14	16	68	8.1	11.6	5.4	87	25
Ext' Dairy	UB 16	28	35	5.7	11.9	1.0	164	54
Int' Dairy	UB 27	32	54	6.1	31.8	0.6	216	158
MEAN		**29**	**40**	**4.8**	**12.3**	**2.2**	**111**	**55**
Int' Dairy	UB 11	49	23	7.7	20.0	0.0	202	124
Int' Dairy	UB 5	64	48	14.7	26.1	2.2	196	142
Int' Dairy	DM03	94	19	3.6	18.3	0.0	170	192
Int' Dairy	DM02	74	33	10.4	21.7	3.1	168	147
Int' Dairy	DM05	72	26	7.9	23.8	0.0	165	198
Int' Dairy	DM08	255	31	10.3	28.4	1.9	164	146
Int' Dairy	DM12	129	25	7.9	19.6	0.8	156	157
MEAN		**105**	**29**	**9.0**	**22.6**	**1.1**	**174**	**158**

To help shed light on the matter, P fluxes through ruminant livestock enterprises were examined to see if the difference between farm groups could be linked to one or more of the variables involved in the farm P cycle (Figure 43). Figure 43 indicates the key P inputs and outputs on ruminant farms and the relevant pools and sinks where P is temporarily stored. The blue boxes contain variables that were available, whereas boxes of other colours contain variables that need to be quantified. To aid our investigation, best estimates of these latter variables were therefore sought.

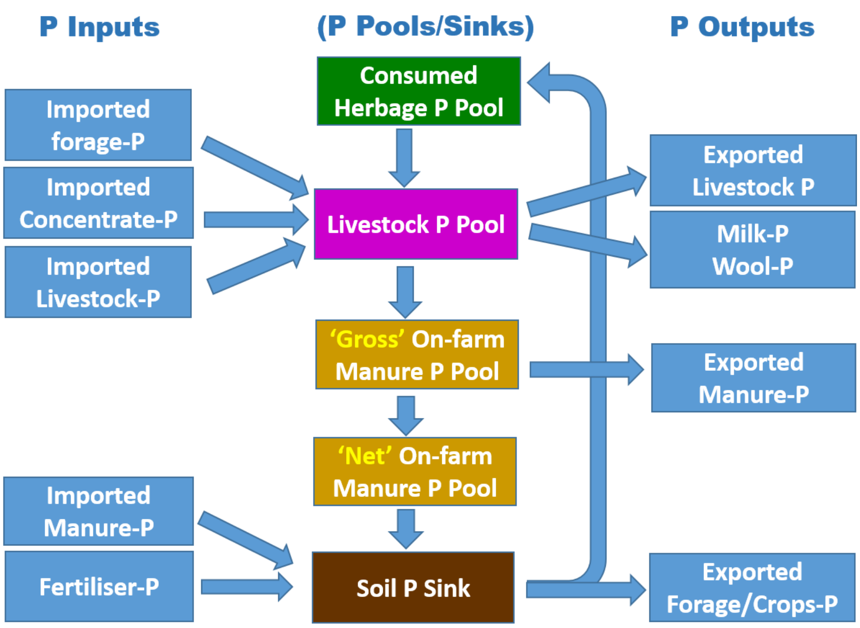


Figure 43. Farm P cycle: inputs, outputs, pools and sinks of P within ruminant farms

Figure 44a shows how Farm-gate P balance is calculated. However, as our primary concern is with the impacts of P surpluses on soil P, a more meaningful Soil Surface (SS) P balance may be calculated as indicated in Figure 44b. Unlike the Farm-gate P balance calculation, where the values of all variables are available (Figure 44a), the SS P balance calculation (Figure 44b) contains two key variables that are more difficult to evaluate –'net on-farm manure-P' and 'consumed herbage-P'. *('net on-farm manure-P' being the actual amount of on-farm produced manure applied to land; and 'consumed herbage-P' being the net amount of herbage removed from the field, excluding that in herbage trampled and left by grazing animals, and that in forage rejected by over-wintered livestock and returned to land via manure application*).

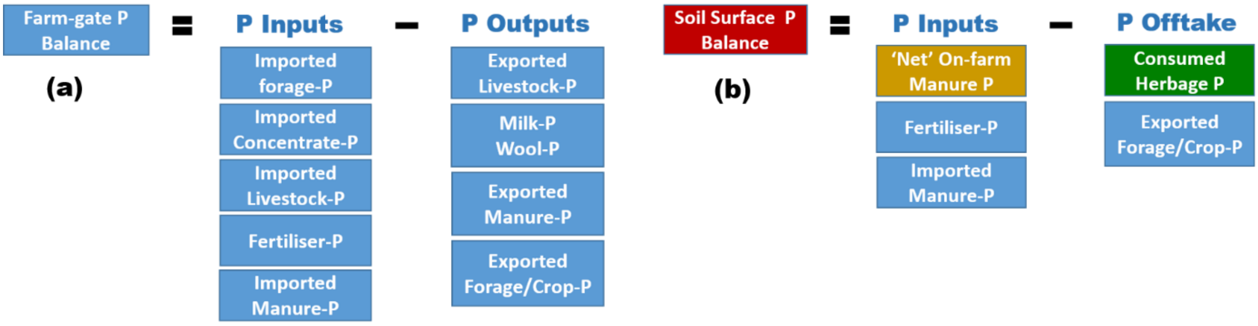


Figure 44(a) Farm-gate P balance calculation, and **(b)** Soil Surface P balance calculation

To obtain estimates for these two variables, a number of assumptions were made. Firstly, it was assumed that on average, over a complete year, the livestock P pool is in 'steady state', with P accumulation in imported livestock/live-weight gain balanced by P removal in exported livestock (*meat and culled animals*). Hence, with no net storage of P surpluses in livestock tissue, the soil is assumed to be the sole sink for P. If this is correct, the farm-gate P balance is mathematically identical to the Soil Surface (SS) P balance, and as shown in Figure 45a, may be calculated using the variables required to calculate the SS P balance. By rearranging the equation in Figure 45a, therefore, a value can be obtained for 'consumed herbage P', as shown in Figure 45b.

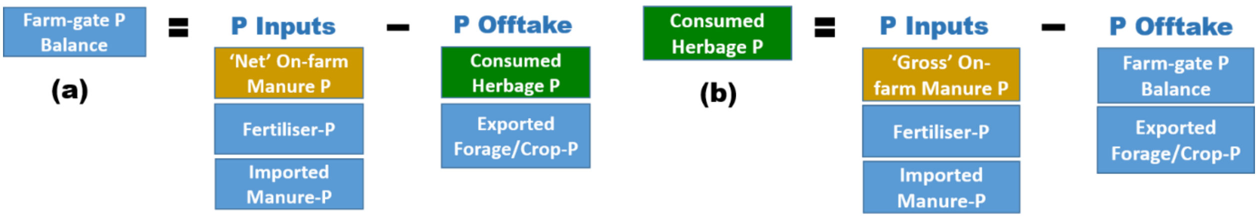


Figure 45(a) Alternative farm-gate P balance calculation, and **(b)** Consumed herbage P calculation

Other research (Gourley *et al*., 2011) has shown that losses of manure-nutrient from farms as a result of excretion on lanes and roadways during cattle movement, represent only a tiny percentage of total manure nutrients produced, and thus are considered negligible for present purposes. Equally, run-off losses of P from land, while ecologically significant, also represent a very small proportion of total P applied and are likewise considered negligible.

The only remaining variable to be quantified is the 'on-farm manure-P pool'. While we don't have actual measurements of manure-P production on farm, it is possible to estimate it using standard values for P excretion, as given in the NAP Guidelines. However, the results of a sister E&I project (*16/4/01 "Managing manure nutrients for sustainable grass-based dairy production"*) indicated that the concentration of P in dairy manure increases exponentially as farm-gate P balance increases, and so standard P excretion values could over- or under-estimate manure-P production. To avoid this scenario, predicted volumes of manure produced annually by different classes of ruminant livestock (*as given in the DEFRA Fertiliser Manual RB209 8th Edition*) were multiplied by the analysed concentrations of P in winter manure samples, to provide values for 'gross on-farm manure-P' (*on the assumption that manure P concentrations during winter are indicative of average manure P concentrations during the complete year*). The only exception was for sheep manure, since samples of manure/excreta were unavailable for analysis. In this case, NAP Guideline standards for P excretion had to be used instead. 'Net on farm manure-P' was then estimated by subtracting 'exported manure-P' from 'gross on-farm P'.

With estimates now available for 'consumed herbage-P' and 'net on-farm manure-P', and measurements of manure total P and C concentrations also available (*except for UB 1 where no slurry was available for analysis*) (Table 24), it was possible to drill down and determine which variable(s) had been responsible for the markedly different responses by soil Olsen-P to changes in farm-gate P surpluses on **Group 1** and **Group 2** farms (Figure 42). Intuitively, it seemed possible that the different relationships between soil Olsen-P and P surpluses in Figure 46a (*where farm-gate P balance has been replaced by SS P balance as the mathematically equivalent parameter*), had arisen because the effectiveness of manure P in increasing Olsen-P, was less for farms in **Group 2** than for those in **Group 1**.

Table 24. Farm-gate (FG) P balance, Effective SS P balance, various P pools, percentage of dietary P derived from forage or grass, and the C/P ratio of winter manures

ID	FG P Bal (kg/ha)	Eff SS P Bal (kg/ha)	Manure-P (kg/ha)	Feed P (kg P/ha)	Consumed Herbage-P (kg N/ha)	%Dietary P from grass	Manure C/P (g/kg DM)
UB 1	-0.4	-0.2	10.2	0.5	10.6	96	-
UB 31	5.4	5.7	12.6	2.0	12.3	86	63
UB 30	4.0	4.2	9.1	1.1	9.2	89	67
UB 14	8.1	8.4	15.7	4.3	13.0	75	32
UB 16	5.7	6.3	30.3	10.3	25.5	71	51
UB 27	6.1	7.3	62.3	19.2	56.8	75	42
MEAN	**4.8**	**5.3**	**23.4**	**6.2**	**18.5**	**82**	**52**
UB 11	7.7	0.0	38.6	20.4	30.9	60	48
UB 5	14.7	7.4	36.6	23.8	24.1	50	47
DM03	3.6	-2.1	28.1	18.4	23.5	56	62
DM02	10.4	3.8	33.2	18.5	25.9	58	48
DM05	7.9	1.2	33.4	23.2	23.1	50	54
DM08	10.3	2.4	39.9	26.5	31.4	54	49
DM12	7.9	2.6	26.9	19.5	19.8	50	56
MEAN	**105**	**2.7**	**33.8**	**21.5**	**23.8**	**54**	**52**

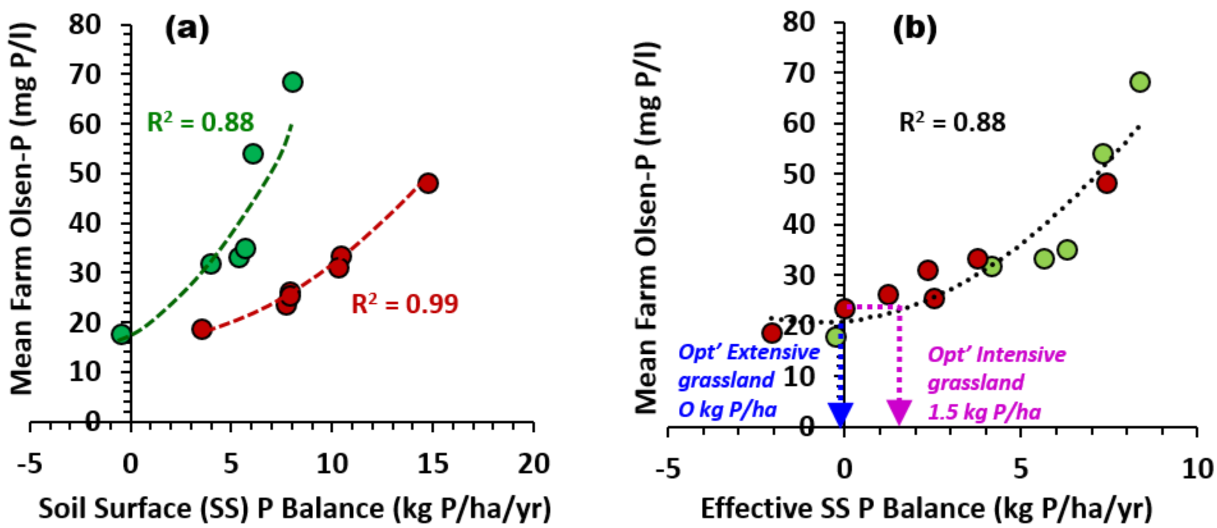


Figure 46 (a) mean farm Olsen-P versus SS P balance, and **(b)** mean farm Olsen-P versus 'effective' SS P balance – showing optimum values for 'effective' SS P balance for extensive grassland (target P index 2^-) and intensive grassland (target P index 2^+)

Results from a sister E&I project (*16/4/01*) indicated that manure-P from intensively managed dairy farms, on average is only 76% as effective as fertiliser P in raising soil Olsen-P. Significantly, therefore, it was found that by re-calculating SS P balances (*to give 'effective' SS P balances*) via iteratively applying separate (varying) coefficients to the 'net on-farm manure-P pools' in each farm Group, a best fit relationship was obtained when 'net on-farm manure-P' was multiplied by 1.02 on **Group 1** farms, and by 0.8 on **Group 2** farms. In other words, manure-P on **Group 2** dairy farms appeared to be only 80% as effective as fertiliser P (*in raising soil Olsen-P*), consistent with findings from E&I 16/4/01, whereas manure-P on **Group I** beef, B&S and dairy enterprises, appeared to be marginally more effective (102%) than fertiliser-P in raising soil Olsen-P.

The single curvilinear relationship in Figure 46b indicates that Olsen-P can be maintained within the target P Index 2^- range (*for extensively managed grassland*), and the target P Index 2^+ range (*for intensively managed grassland*), when effective SS P balance is optimised at 0 kg P/ha/year and 1.5 kg P/ha/year, respectively. As shown in Figure 47a, these effective SS P balance optima translate into farm-gate P balance optima of 0 kg P/ha/year, for **Group 1** farms with extensively managed grassland (**EG**), 1.5 kg P/ha/year, for **Group 1** farms with intensively managed grassland (**IG**), and 7 kg P/ha/year, for **Group 2** farms with intensively managed grassland (**IG**) – the same as obtained from Figure 42.

The question then is: why is manure-P on **Group 1** farms more effective in raising soil Olsen-P than that on **Group 2** farms? Research elsewhere has demonstrated that the effectiveness of manure-P in raising soil test P is negatively influenced by the manure C/P ratio and phytic acid content (Leytem *et al*., 2005; 2006). In the present study, however, the average manure C/P ratio proved to be identical (52) on both groups of farms (Table 24). As regards manure phytic acid content: while measurements of phytic acid were not made, it had previously been demonstrated by Jarett (2011) that forages contain less phytate (*phytic acid*) than concentrate feeds. As a working hypothesis, therefore, it was assumed that the greater the contribution of forage-P to ruminant dietary P intake, the lower the amount of phytic acid excretion in manure, and hence the greater the effectiveness of manure-P in raising soil Olsen-P (Leytem *et al*., 2006). Significantly, therefore, it was found that the proportion of dietary P coming from grass and forage was consistently greater on **Group 1** farms *(> 65%; mean 82%*) than on those in **Group 2** *(< 65%; mean 54%*), as shown in Table 24 and Figure 47b.

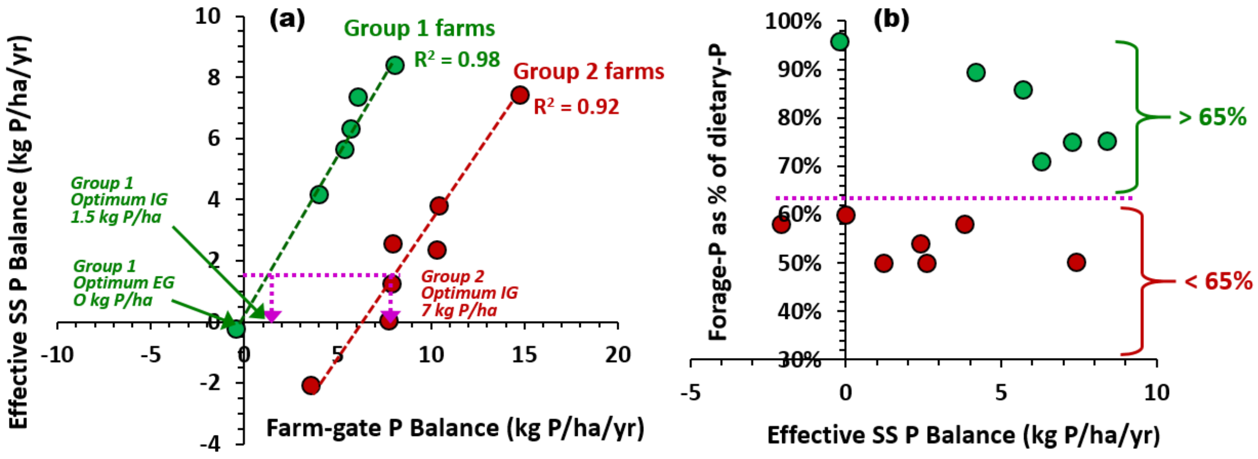


Figure 47 (a) effective SS P balance versus farm-gate P balance, showing optimum farm-gate P balances for Group 1 farms managed extensively or intensively, and Group 2 farms managed intensively, and **(b)** forage-P as a % of total dietary P versus effective SS P balance, showing the much greater proportion of dietary P from forage on Group 1 farms than on Group 2 farms

These results therefore suggest that the key factor governing the relationship between soil Olsen-P and farm-gate P balance, is the extent to which ruminant systems are reliant on either forage or concentrates for milk and meat production. Manure-P produced on intensive dairy farms, where milk production is driven to a large degree by concentrate inputs, appears to be less capable of influencing soil Olsen-P than manure-P produced on dairy, beef, and B&S enterprises where milk and meat production is primarily driven by grass and forage consumption.

From a DAERA Policy perspective, the results of this study appear to provide grounds for lowering the current farm-gate P balance limit on derogated farms to < 8 kg P/ha/year, and indeed extending it to all intensive dairy farms on which there is a high reliance on concentrate feed for milk production. There may also be grounds for applying a farm-gate P balance limit of < 5 kg P/ha/year to both intensive (*dairy*) and extensive (*dairy, beef, B&S and sheep*) livestock farms where there is a high reliance on grass and forage for meat and milk production. The latter limit, while appreciably greater than the optima of 1.5 kg P/ha/year and 0 kg P/ha/year, should nevertheless curtail excessive soil Olsen-P accumulation (> Index 2^+) on farms that are highly reliant on grass and forage for meat and milk production.

Before imposing such limitations, however, it is important to assess what changes would be required in P management on the different farm types in order to bring farm-gate P surpluses down below these limits. This assessment was done using available data for farms in UB and in the Dairyman Project. For beef and B&S enterprises, it proved simple: elimination of some or all of the fertiliser P inputs was sufficient to bring farm-gate P balances below 5 kg P/ha/year on all farms. For low and high intensity dairy enterprises where milk production is driven mainly by grass and forage, eliminating fertiliser P inputs likewise brought farm P balances down below 5 kg P/ha/year in the majority of cases. In contrast, for intensive dairy enterprises where milk production is largely driven by concentrate feed, elimination of fertiliser P alone failed to bring P balances down below 8 kg P/ha/year when they were greater than 12 kg P/ha/year to begin with; but reductions in the concentration of P in imported feeds along with P fertiliser elimination, did achieve this goal - without the need for any reductions in concentrate feed usage *per se*, and hence in farm profitability. However, while data were unavailable for farms with P surpluses > 15 kg P/ha/year, it is likely that reductions in imported feed usage may be required on such farms, if farm-gate P balances are to be brought down below 8 kg P/ha/year.

Conclusions

1. The present study has identified two types of ruminant farming systems requiring different farm-gate P balances to maintain average soil Olsen-P concentrations within the target Index 2^+ and 2^- ranges for intensively and extensively managed grassland, respectively. The two types are distinguished, not on the basis of enterprise type, i.e. dairy, beef, B&S, but rather on the extent to which production is driven by grass and forage as opposed to concentrate feed.

2. For intensively managed dairy farms, the optimum P farm-gate P surplus is higher when milk production is driven largely by concentrate feed (*7 kg P/ha/year to maintain P Index 2^+*), than it is when production is driven by grass and forage consumption (*1.5 kg P/ha/year to maintain P Index 2^-*). For extensively managed dairy, beef and B&S enterprises, where production is primarily driven by grass and forage consumption, a farm-gate P balance of 0 kg P/ha/year is sufficient to maintain soil at P Index 2^-.

3. There is evidence that the differences in P surplus optima reflect differences in manure-P effectiveness, which in turn may be linked to differences in the concentration of phytic acid in manure, since this compound can limit the effectiveness of manure-P in raising soil Olsen-P (*concentrate-fed animals potentially having more phytic acid in their manure than those fed primarily on grass and forage*). However, analysis of phytic acid in manure is required to validate this hypothesis – and this will be undertaken in the new E&I project: 18/4/03.

4. Subject to confirmatory evidence demonstrating a causal link between the proportion of forage/grass-derived P in ruminant diets and the phytic acid content of manure, there appears to be a case for lowering the current farm-gate P balance limit on derogated farms to < 8 kg P/ha/year, and indeed extending it to all intensive dairy farms on which there is a high reliance on concentrate feed for milk production. There also appears to be a case for applying a farm-gate P balance limit of < 5 kg P/ha/year to both intensive (*dairy*) and extensive (*dairy, beef, B&S and sheep*) livestock farms where there is a high reliance on grass and forage for meat and milk production. This limit, while greater than the optima of 0 kg P/ha

/year and 1.5 kg P/ha/year identified for these farming systems, should nevertheless curtail excessive soil Olsen-P accumulation (> Index 2^+) on farms where there is a high reliance on grass and forage for meat and milk production.

Applying the above farm-gate P surplus limits should be achievable on dairy, beef, and beef and sheep enterprises where production is primarily driven by grass and forage consumption, simply by eliminating some or all of the fertiliser P inputs. In contrast on intensive dairy enterprises, where milk production is largely driven by concentrate feed, elimination of fertiliser-P alone may not suffice, and reductions in the concentration of P in concentrate feeds may also be needed, and in cases where farm P surpluses are > 15 kg P/ha/year, reductions in total concentrate imports may also be required.

10.4 E&I Project 16/4/01 – Management of manure nutrients for sustainable grass-based dairy production in Northern Ireland (New Research)

WP2 - Assess the efficiency of screw-press mechanical separation for partitioning P (N, K and S) in dairy slurries (of differing initial DM contents) and anaerobic digestates, with or without MgCl2 pre-treatment to precipitate struvite (NH4MgPO4·6H2O).

It is currently estimated that almost 50% of farmed grassland soils in Northern Ireland (NI) are above Olsen soil phosphorous (P) Index 2+, the agronomic optimum for these soils, primarily in areas where intensive dairy production is practiced. There is a significant body of research demonstrating a strong relationship between soil P concentrations and P losses to water, with soils above Index 2 posing a significant risk to water quality. While farmers are currently prohibited from applying chemical P fertiliser to soils above Index 2 (except for 1st cut silage crops where a small amount of P is recommended on Olsen-P Index 3 soils), this restriction does not apply to organic manures. Consequently, depending on stocking rate (and hence manure application rates), it is possible that Olsen P concentrations in high P index soils (Index 3 and 4) may continue to increase regardless of restrictions on chemical P usage.

From an environmental perspective high P soils pose a significant threat to water quality. In a 5-year study to assess the relationship between soil P status and P losses in land drainage water and overland flow, Watson et al. (2007) concluded that despite Olsen-P ranging from 22 to 99mg P kg-1, it was difficult to identify a clear Olsen-P concentration or ‘break-point’ at which P losses increased. Nevertheless, the study demonstrated a substantial increase in P loss in both overland flow and sub-surface drains in response to P accumulation. These studies highlight the need to decrease the P concentration of high P soils to more sustainable levels, if the targets of the Water Framework Directive are to be achieved.

Previous research at AFBI Hillsborough (Frost and Gilkinson, 2007) and elsewhere (Hjorth et. al., 2010; Moller et al., 2000) has indicated that slurry separators, e.g. decanting centrifuge, screw press and brushed screen separators, can partition P into the separated solid fraction leaving a separated liquid fraction less enriched with this nutrient and more balanced as a fertiliser with higher ratios of nitrogen : phosphorous (N:P) and potassium :

phosphorous (K:P) (Frost and Gilkinson, 2007) and higher N efficiencies than the original slurry (Frost et al., 2014). Separating cattle slurry and applying the P-depleted high K:P ratio liquid fraction to P-enriched soils and the P-rich solid fraction to grassland of low P status or else to arable land (or exported), could be an effective strategy for reducing the manure P loading of P-enriched grassland without running down soil K status. This type of nutrient partitioning allows a more flexible approach to slurry management and minimises the chances of K become limiting to grass production as a consequence of efforts to minimise P build-up in soil (Bailey, 2015). To date, however, AFBI have not examined the use of a screw press to separate farm slurries of different dry matter (DM) contents or anaerobic digestates (Frost et al., 2014).

This aim of this work was to provide information on the costs, practicalities and efficiency of screw press separation when dairy slurries of different DM contents and anaerobic digestate are used as feedstock; information that is not currently available to the local farming industry. Detailed information on separator operating conditions and energy costs to achieve DM and nutrient (N, P, K) partitioning to the separated solid and liquid phases were investigated.

Technical Report Conclusion

This report provides information on the practicalities and efficiency of screw press separation along with energy costs, when dairy slurries of variable DM and plant nutrient content as well as anaerobic digestate are used as feedstock. As the influent feedstock DM decreased the DM and P separation efficiency increased whilst the K separation efficiency was similar for three out of the four feedstock analysed, and no clear trend existed in N separation efficiency versus feedstock type. The separation efficiencies for DM and P partitioning were reduced by using a smaller screen size, N remained similar and K increased in the separated solids. The separation efficiency results which we have reported correspond well with those found in published literature. Energy costs for separation were directly related to feedstock DM content. The results showed that it was possible to partition 14 – 29% of the total P from the influent feedstock to the separated solids fraction for export off-farm, suggesting that screw press separation could be used effectively to reduce the manure P loading of P-enriched grassland without running down soil K status. This type of nutrient partitioning allows a more flexible approach to slurry management.

Table 25: Energy requirements and cost estimates (per m3 of feedstock) to separate dairy slurries and anaerobic digestate with variable dry matter content during a two hour separation run (mean of three triplicate runs).

Feedstock	Flow rate (l/h)	Separator Energy (kWh)	Mixing Energy (kWh)	Total Energy (kWh)	Energy/m3 (kWh/m3)	Energy Cost* (£/m3)
Dairy slurry 1	8872	8.8	10.0	18.8	1.06	0.16
Dairy slurry 2	9358	8.6	10.0	18.6	0.99	0.15
Dairy slurry 3	11630	8.3	10.0	18.3	0.78	0.12
Digestate	11370	8.0	10.0	18.0	0.79	0.12

*Energy cost (£/m3) was estimated by applying an electricity price of £0.15/kWh.

WP3 - Assess the plant availability of P, K and S in liquid fractions of separated dairy slurries, anaerobic digestates and dairy slurries with a wide range of P concentrations.

Plant availability of P, K and S in the liquid fractions of separated dairy slurries is also being examined in a greenhouse pot experiment, to assess the availability of these manure nutrients for grass uptake and production. A pot experiment was established to evaluate the plant availabilities of P, K and S in slurry samples from representative benchmark dairy farms and in liquid fractions of screw-press separated slurries and digestates, relative to P, K and S supplied through inorganic fertiliser. The trial was conducted using a low P soil (Index 2- for P and Index 1 for K), and with perennial ryegrass as the indicator crop. Plants were established for 6 weeks and cut, and then liquid manures and inorganic fertilisers applied to the surface of pots for two subsequent harvests. Dry matter yields and nutrient offtakes were evaluated at each harvest and results for manure treatments compared with those for the fertiliser control (response curves) to assess relative availabilities of P, K and S in the different manures and manure liquid fractions and digestates.

The objective is to identify and verify potential measures that could be used on dairy farms of different intensity that would enable recycling of manures and liquid fractions to soils of even high P status to sustain productivity while at the same time reducing the risk of P run-off to water. Results will be available in the final year of the project.

10.5 E&I Project 16/4/02 – Quantification of phosphorus release from sediments in Lough Neagh and factors affecting the recovery of water quality

Key finding of the project

- Phosphorus released from Lough Neagh sediment will continue to delay significant improvements in water quality for approximately 40 years following the reduction of external nutrient loading.

Lough Neagh has a long history of nutrient enrichment originating from persistent inputs of nutrients, especially phosphorus and nitrogen, from catchment sources. These nutrients have resulted in algal blooms and an overall deterioration of the lake water quality. External loading has been extensively monitored and managed since the 1970s in an attempt to mitigate eutrophication and improve water quality.

Decades of excessive external loading has resulted in the accumulation of an extensive legacy P reservoir within the lake sediments, which is released as large pulses to the water column. This process of legacy P release, termed “internal loading”, undermines nutrient management mitigation measures in the catchment and is often responsible for impeding long-term water quality improvements following the reduction of external nutrient inputs.

It would not be feasible nor cost effective to implement within lake measures to control internal loading in a lake as large as Lough Neagh. It is therefore useful to estimate how long it will take the sediment P inventory to diminish via natural processes until it is no longer an important factor in the lake P budget. This project has quantified the sediment P reservoir and summarised the rate of ongoing within-lake processes critical to the removal of P from the lake system.

Multiple sediment cores were collected from the 10-14 m depositional zone that dominates Lough Neagh's lakebed. These cores were sectioned, radiometrically dated, and chemical extractions performed to both quantify the P inventory and characterise P forms within the depth of 0 – 45 cm. Concentration of total P within surface sediments (0 – 1 cm) was high with a mean of 2.87, range 3.48 – 1.94 mgP gDS^{-1}. This decreased with depth and time to a mean of 0.758, range 0.860 – 0.579 mgP gDS^{-1} by 45 cm. Chemical characterisation identified >60 % of surface sediment P to be labile, i.e. available for transformation and release, which on average dwindled to <35 % by 45 cm.

Mineralisation and release of P occurs only within the upper active layer (Z_{active}) sediment, meaning that the P bound within sediment beneath the active layer depth is generally unavailable. The depletion of the sediment P inventory was described using a condensed box model consisting of two "purging factors" removing labile P from the active layer (Z_{active}); the rate of mineralisation or diagenetic transformation of labile P (k_d, yr^{-1}), and the rate of sequestration of all P forms via burial in deeper sediment (ω cm yr^{-1}). The sum of these purging factors (λ, yr^{-1}) was used to predict the overall rate of P loss from the active layer. The model outputs are summarised in Table 26, the estimated timescale to recovery was taken as the time required to reduce the available labile P inventory by 75% (τ75).

Table 26: Summary of sediment model inputs for the five sediment cores including, burial velocity (ω), active layer depth (Z_{active}), diagenetic rate constant (k_d±SE), labile P loss rate constant, and final model outputs with response times for Lough Neagh sediment cores in years (τ_{75}).

Core ID	ω, cm yr^{-1}	Z_{active}, cm	kd, yr^{-1}	ω/Z_{active}, yr^{-1}	λ, yr^{-1}	τ_{75}, yr
LN11	0.229	8	0.0184 (0.00730)	0.0286	0.0470	30
LN15	0.519	29	0.0159 (0.00490)	0.0179	0.0338	41
LN17	0.304	21	0.0184 (0.00344)	0.0145	0.0329	42
LN18	0.377	23	0.0138 (0.00538)	0.0164	0.0302	46
LN19	0.145	10	0.0161 (0.00545)	0.0145	0.0306	45

The mean ± SE timescale of recovery from eutrophication derived from the application of the sediment model to five core samples was 41 ± 3 years. There was good agreement between the core samples, with a coefficient of variation (CV) of 15.6 %. Diagenetic transformation of particulate P to soluble P was an important pathway for the removal of labile P within upper "active layer" sediments. However, diagenesis was not responsible for the immobilisation of all labile P, as inferred by the persistence of potentially labile P in deeper sediments. The rate of labile P loss through burial accounted for 51.9 % ± 0.03 % (CV 12.5 %) of the total labile P losses from upper active sediment. This highlighted the capacity of Lough Neagh sediments to retain P within deep sediments and was a significant pathway for the long-term removal of P from the lake system.

The outcome of the sediment model suggests that internal loading will continue to delay significant observable improvements in water quality for approximately 40 years following the reduction of external nutrient loading. Therefore, it is important that lake managers take this lasting source of nutrients into account in order to set realistically achievable water quality targets over the medium and long term.

10.6 E&I Project 17/4/01 - Factors affecting the ecological recovery of Northern Ireland Freshwaters

Introduction

Agriculture is recognized as a major source of nutrient loading within freshwater systems contributing to the impairment of ecological quality across Europe (EEA, 2003) and within Northern Ireland (NI) (DAERA, 2018). Such impairments to system water and aquatic quality are addressed via targets for ecological quality within the Water Framework Directive (WFD: 2000/60/EC) and nutrient loading pertaining to Nitrogen (N) within the Nitrates Directive (ND) and associated regulations within Member States. Moreover, the WFD itself is a comprehensive piece of legislation which integrates many policies, including the ND, with the aim of establishing 'Good' status by 2027 based on both ecological and chemical status of surface waters. 'Good' chemical status requires specifically that nutrient concentrations do not exceed agreed target levels (see WFD UK TAG) so as to ensure the achievement of 'Good' ecological status for the biological quality elements (phytobenthos, fish, macroinvertebrates and macrophytes) as well as the functioning of the ecosystem more broadly. Specifically, Article 8 of the WFD sets the requirements for monitoring to obtain ''a coherent and comprehensive overview of the water status'' (EC, 2000). This includes the characterisation of the relative risk of diffuse pollution from agricultural areas and the effectiveness of measures implemented under action programmes. Therefore, implementation of the WFD and the ND rely on appropriate monitoring programmes, both chemical and ecological, that can determine the effectives of various measures to ensure that chemical status allows for the achievement of 'Good' ecological status. This project allows for the exploration of synergies within ecological monitoring under the WFD and chemical monitoring under NAP (WP 2,3,4,5), relating to N, to be explored with the diffuse risk being addressed specifically within WP 3 and the effectiveness of measures (e.g. EFS) within WP 5.

In NI, 68.7% of rivers currently fail to achieve at least “Good” Ecological Status with diffuse phosphorus inputs the key stressor from agriculture (Cave 2015, Cave & McKibbin, 2016). However it is important to note that while phosphorus can be the limiting nutrient in rivers and the prime cause of failure to meet “Good” ecological status, many systems are nitrogen, or both nitrogen and phosphorus, limited and this can vary through the year and from catchment to catchment (Köhler and Gelbrecht, 1998; Smith *et al*., 1999; Elser *et al*., 2007, Blaas and Kroeze, 2016). Therefore, addressing both N and P and how they vary spatially and temporally is important for the restoration of “Good” ecological status, and particularly under future forecasts of climate change where seasonal inputs may vary (Ockenden et al 2017). This project addresses how different stressors (N, P, sediment) may inhibit the recovery of NI Freshwaters to meet targets outlined within the WFD relating to the aquatic environment (WP 2, 3). It also supports on-going work within AFBI under the Nitrates Directive by providing ecological context to monitoring and mitigation measures. The importance of these linkages was emphasised in June 2020 by comments by Virginijus Sinkevičius, the European Commissioner for the Environment, who confirmed that the WFD will not be revised and that all countries should ensure that aquatic systems achieve ‘good ecological condition’ by 2027. Specifically, the European Commission position is that the coordination of policy with specific stressors to include agriculture together with related EU legislation which includes the Nitrates Directive, is paramount to achieving the ambition of the WFD.

Project Objectives

The overall objectives of the proposed research are to provide scientific evidence to:

1) Discriminate between current and legacy impacts of physical and chemical disturbance in constraining the recovery of macro-invertebrates communities to good ecological status across a gradient in land use intensity in Northern Ireland streams **(WP2)**
2) Determine the contribution of field (animal manure and soil P, rural septic tanks) bank and in-stream sediment sources to in-stream P and assess the impact of nutrient-sediment sources on phytobenthic composition and biomass **(WP3)**.
3) Identify the secondary pressures (natural environment, land use, in-stream stress, climate variables) impacting the achievement of good ecological status in river systems which have been subject to nutrient reduction initiatives **(WP4).**
4) Determine the indicative role of diatoms (biomass and composition) and macro-invertebrates (composition) to assess primary and secondary stress in the recovery of catchments to good ecological status across a gradient in agricultural disturbance **(WP4).**

In addition to these objectives, the project has sought an extension to extend the project and provide supporting ecological evidence in relation to the NAP/EFS scheme within **WP5**. This effectively widens the scope of the final work package which originally sought to provide recommendations on measures to improve ecological status of NI streams.

Results

Data processing and interpretation of all data is currently on-going. Therefore, here we present some of the key preliminary findings to date relating to the Upper Bann and Colebrooke catchments.

Historical data exists from previous macroinvertebrate field campaigns within the Upper Bann and Colebrooke (undertaken as part of DARD E&I Project 0803). The last of these surveys was conducted in 2009 and therefore, a repeat of a subset of sites of 12 sites within this historical dataset was undertaken to examine the recovery potential of macroinvertebrate communities and achievement of 'Good' ecological status. This was explored in the context of all nutrient (N, P: Figure 48) (dating from 1990-1999; 2009-2014 (DARD Project 0803); August 2016-present (DAERA E&I Project 16/4/03)). Figure 48 demonstrates how high concentrations of nutrients (SRP, TON) in the early 1990s for the Upper Bann catchment which demonstrated a decreasing trend through the data series. BMWP scores indicated un-impacted conditions at the beginning of the 1990's. This was followed by a marked decrease in BMWP scores to indicate moderate impacted, with an improvement to clean but slightly impacted in 1995 before a plateau with BWMP scores indicated moderate impacted conditions from 1995 to 2019. This apparent resistance within the community to changes in nutrient concentration, legalisation and management is currently being investigated further. However, we hypothesise that a key contributing factor may related to biological and hydro-morphological factors e.g. the ability of macroinvertebrates to recolonise and habitat degradation due to fine sediment.

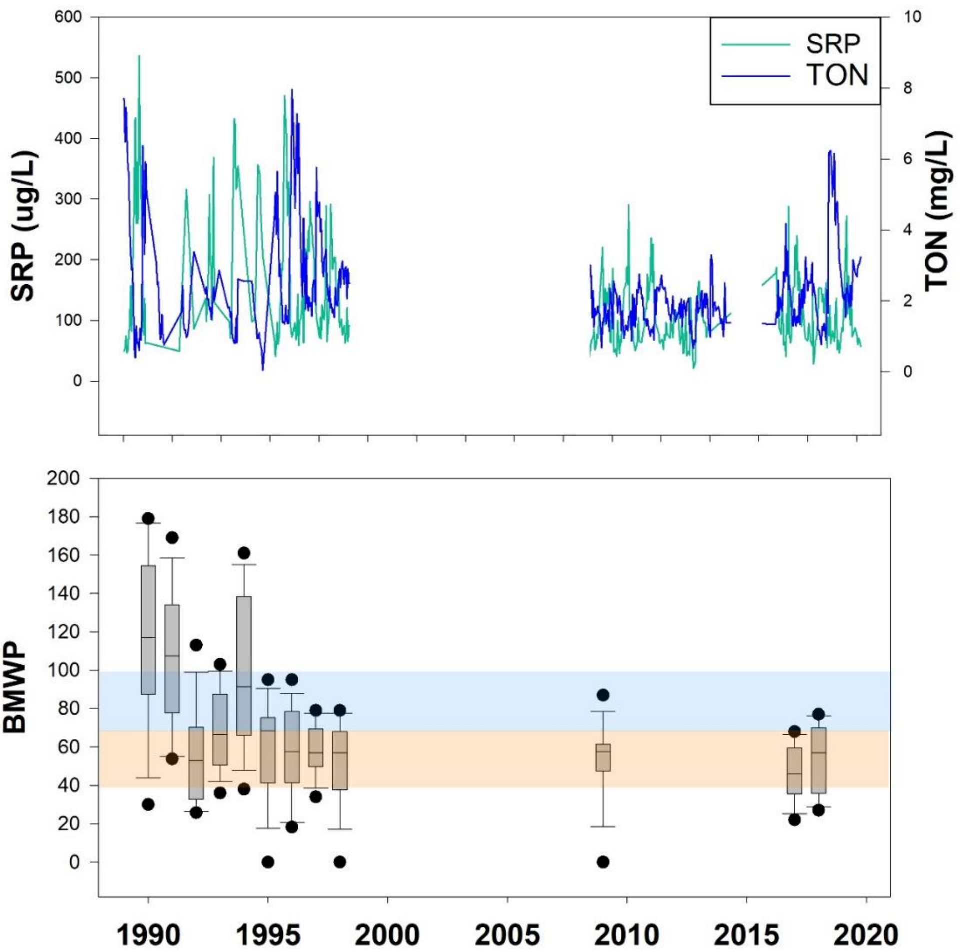


Figure 48: (a) Bi-monthly SRP and TON; (b) median BMWP scores (spring, summer, autumn) for the macroinvertebrate communities, for the Upper Bann Catchment 1990-2020.

In addition to macroinvertebrates, assessment of phytobenthic biomass commenced in Spring 2018 and were assessed seasonally until Winter 2019 within historical monitoring sites across the Upper Bann and Colebrooke. Differences were observed in seasonal trends in concentration across catchment. Across both catchments lower chlorophyll-a concentrations were observed in winter. For the Upper Bann catchment, extremes of biomass concentration were confined between the summer and winter periods with average winter concentrations less 1.34 μg/cm^2. However for the Colebrooke catchment concentrations did not demonstrate significant differences across seasons: spring, summer and autumn (Figure 49). The average concentration for these months was 2.1 μg/cm^2. We hypothesise that these observed trends are due to system dynamics (hydrology), land-use stressors (sediment, nutrients) and grazing pressure (macroinvertebrates). We are currently exploring available environmental data and combining this with macroinvertebrate community data. This will then be used in a modelling exercise to partition out variances and determine drivers of observed patterns.

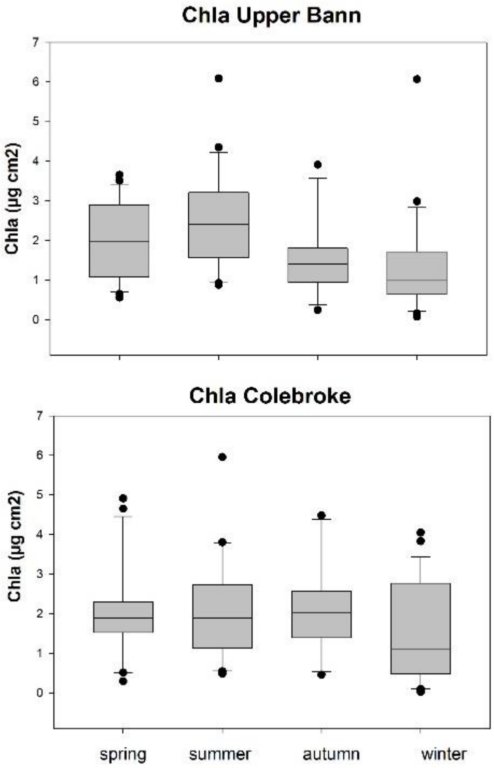


Figure 49: Seasonal chlorophyll-a concentrations across Upper Bann and Colebrooke

10.7 E&I Project 17/4/07- An evaluation of water quality monitoring options suitable for implementation in Northern Irish river catchments

This project evaluated the feasibility and requirement for investment in real-time, high resolution water quality monitoring infrastructure for surface water catchments in Northern Ireland (NI), the provision of which was a recommendation of the 2016 Sustainable Agricultural Land Management Strategy (SALMS) report (Expert Working Group on Sustainable Land Management, 2016).

A key objective of the project was to undertake an assessment of current exemplars of high-resolution water quality monitoring across northwest Europe, and beyond, with specific consideration of the technological and practical issues involved in establishing and maintaining such programmes in the long term. The project was broadened to include other examples of enhanced water quality monitoring. Emphasis was placed on identifying examples of monitoring approaches which demonstrated a positive influence on farmer behaviour toward water quality issues. This was because a key potential benefit identified in the SALMS report was the potential for high-resolution data to be used as a communication tool which could deliver positive behavioural change to improve the environment.

Recognising that amplified water quality (chemical) monitoring may yield agri-environmental insights beyond established low resolution monitoring in rivers, the objectives of this project, were to:

1. Investigate state-of-the-art exemplars of water quality high-resolution monitoring and consider all technological and practical issues for establishment.
2. Investigate this practice with reference to examples of farmer behavioural change.
3. Propose an ‘options matrix’ of water quality sampling practices to give optimal and enhanced solutions to specific water quality monitoring objectives.

A series of interviews and site visits were undertaken at 11 current monitoring programmes in Ireland, UK and northwest Europe, selecting programmes with comparable agricultural pressures to those in NI and with similarities in terms of landscape and climate (Table 27, Figure 50). Monitoring approaches applied in the programmes are diverse, ranging from enhanced grab sampling and laboratory analysis to sub-hourly sampling of multiple parameters and nutrients in autonomous high-specification, bank-side laboratories.

Table 27: Institutions visited during the project.

	Institution	**Location**
1	Norwegian Institute for Bioeconomy Research (NIBIO) JOVA Catchments, Norwegian Agricultural Monitoring Programme	Ås, Norway
2	Swedish University of Agricultural Sciences (SLU) National Agricultural Catchments Monitoring Programme	Uppsala, Sweden
3	IMBL (Bimmen-Lobith) International Commission for the Rhine	Germany/ Netherlands
4	Winterwijk Monitoring Catchment, Deltares	Utrecht, Netherlands
5	INRAE AgrHyS Environmental Research Observatory	Rennes, France
6	Rothamsted Research (RR) North Wyke Farm Platform	Devon, UK
7	Bristol University Avon Demonstration Test Catchment	Bristol, UK
8	Environment Agency Regulatory Monitoring, Support to DTC platforms	Reading, UK
9	Teagasc Agricultural Catchments Programme (ACP)	Corduff, Ireland
10	University of East Anglia Wensum Demonstration Test Catchment	Norfolk, UK
11	The Meettrailer, Rijkwaterstaat, Groot Ammers, Dordrecht	Dordrecht, NL

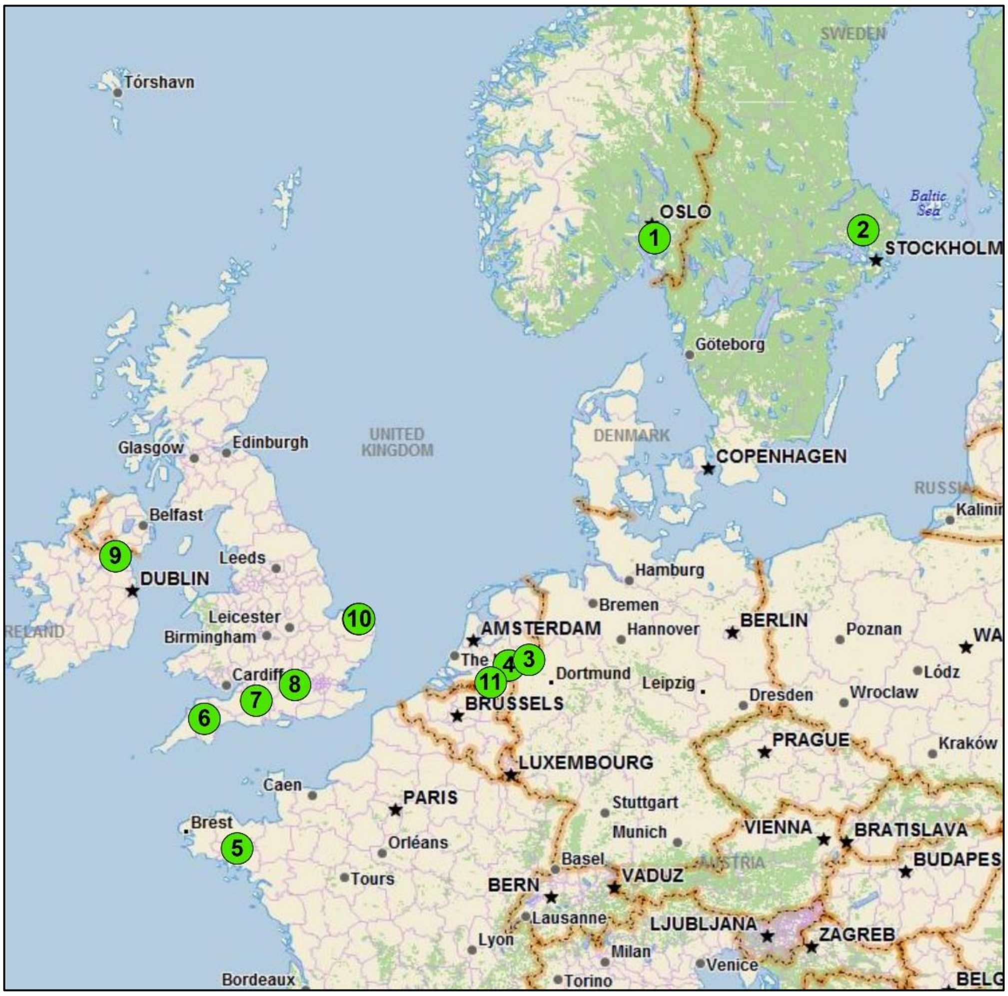


Figure 50: Sites visited in Ireland, England, Norway, Sweden, France, Germany and the Netherlands

Based on the evidence gathered, options were identified and evaluated for suitability to deliver optimal and enhanced solutions to specific water quality monitoring objectives in Northern Ireland. With the exception of the DEFRA DTC programmes in England and Wales, where a primary remit was to influence both policy and farm practice change, none of the programmes visited could readily identify influences that had caused behavioural change among farmers. This was principally because the other programmes were focussed on top-down policy change or surveillance rather than specifically focused on influencing farmer behaviour. Nevertheless, staff from the programmes were clear that the farmer engagement potential was very high and that the sites acted as important focus points for discussion on water quality issues for farmers, policy stakeholders and scientists alike, and so part of a suite of tools that might ultimately change behaviour.

With the objective to empower farmers with meaningful data and increase the potential for behavioural change, the recommendation in the SALMS report for investment in high-resolution nutrient monitoring stations at 60-80 locations is likely to be a high risk in terms of cost-benefit with estimated costs of £6-8 million for capital and annual running costs of up to £1 million. Objectives for enhanced water quality surveillance beyond statutory monitoring practices can be made effectively using, for example, the flow-weighted composite sampling approaches or 24/7 sampling approaches described in the project's final report. These methods provide meaningful data in terms of enhanced surveillance and can account for the hydrologically dynamic characteristics of catchments in NI. However, such set-ups have considerable lags between sample acquisition and analysis, and at lower frequencies are not likely to provide the instantaneous visual impact and source apportionment data that are meaningful to farmers and demonstrate the impact of their activities on water quality.

To act as focus points for pressures on water quality, and to be established as an objective to influence behavioural change from the outset, it is recommended that initial investments should consider:

- A technological trial and training facility that tests the suitability and reliability of a range of real-time analysers/sensors/probes, water delivery and data management systems for Northern Ireland rivers.

- Three mobile real-time water quality monitoring stations should be considered for short-term, roaming deployments across NI and used to demonstrate the contribution of different pressures to water quality. Sited close to existing hydrometric stations, these installations would act as focus hubs for farming groups throughout NI and for educational purposes at CAFRE sites and universities

- Following trials, a small number of fixed high-resolution monitoring stations in key locations within major NI catchments should be considered for enhanced surveillance and as catalysts for engagement and knowledge transfer. These would be located on larger catchments and cover a range of catchment typologies, source issues and pathways. A catchment outreach and educational programme should be developed around each site and include supplementary citizen science activities.

- Investment should be made in a project to quantify the behavioural change potential of both citizen science and mobile/fixed real-time water quality monitoring approaches. Most programmes visited for this report were designed to evaluate or survey policies already in place. An objective specifically aimed at bottom-up engagement and behavioural change would help to define the further utility of enhancing water quality monitoring options.

10.8 E&I Project 17/4/08 - Decision support systems to support the management of nutrient on Northern Irish farms

Summary of Findings

Report - The Closed Period for Slurry Spreading in Northern Ireland: Evidence Base

- Analysis carried out using a 9 year dataset from the AFBI Hillsborough farm (2009-2017), demonstrates significant variability in monthly soil moisture conditions throughout the year (see Figure 1 in report). For the months covered by the closed period, the average % number of days that the soil was above field capacity (i.e. dry enough for slurry application) were 48%, 18%, 3% & 3% for October, November, December and January respectively. However, the minimum-maximum range of values for these months ranged from 0-100% (Oct), 0-95% (Nov), 0-18% (Dec), 0-

20% (Jan). In February the average was 22% with a range of 0-53% of days when the soil was above field capacity. Except for January and December the number of days above field capacity can exceed 50% in any month of the year, and, in some years, even in the summer months, soil can be wet on a significant number of days as demonstrated Figure 1 in the report. This large variability in soil moisture conditions between months and years adds significant uncertainty into the decision making process around the timings of slurry application.

- The results from the SurPhos model simulation at the CENIT site demonstrate In comparison to the No Regulation (NR) scenario, the implementation of the NAP regulations (NAP scenario) results in an average reduction in Soluble Reactive P (SRP) export of 31% across the fully year, with reductions of 24% and 39% in the open and closed period respectively. It should be noted that even with the full implementation of the NAP regulations during the closed period SRP export is 52% higher than in the open period (See Table 1 & Figure 4 in the report)
- The Surphos model simulations show that on an annual basis there was a 46% difference in SRP export between the poorly drained and moderately drained plots, increasing to 87% difference between the poorly drained and well drained plots at the Moorpark Teagasc site (See Table 2 and Figure 5 in the report).
- While there is some variation in grass growth between years, during the period covered by the closed period, based on the NI wide Grass Check data grass growth is consistently low at around 2-6kg DM ha^{-1} day^{-1} with little uptake of nutrients expected.
- The Surphos model simulations show that limiting slurry application to periods when SMD will be above field capacity (≥0 mm) resulted in a 44% and 83% reduction in SRP loss during the open and closed periods respectively, in comparison to the NAP scenario (See Table 3 in the report)
- Using the SurPhos model, the estimated decline in P export in the open period for a 10% (P10 scenario) and 30% (P30 scenario) reduction in the P content of the slurry are 10.7 & 29.2% respectively and 10% and 29.4% respectively during the open and closed periods.
- The estimates from the SurPhos model simulations indicate that a reduction from 50 (A50 scenario) to 30 m^3 ha^{-1} (A30 scenario) results in a 36% reduction in P export during this month (See Table 3 in the report). If a similar restriction was applied during the closed period, there would be approximately a 47% reduction in P export and there would be an 86% reduction in SRP loss if application rates are reduced from 50 m^3 ha^{-1} (A50 scenario) to 10 m^3 ha^{-1} (A10 scenario) during the closed period. (See Table 3 in the report)
- If the closed period was extended to include the entirety of the shoulder period, there would be an average reduction in SRP export of 14% during the resulting shortened open period (1st March-30th Sept). However, it should be noted that this scenario does not take into account the increased volume of slurry that would need to be applied during the shortened open period (See Table 3 in the report).
- The current evidence base supports the closed period as the most effective strategy for minimising the risks associated with slurry spreading from 15th October to the 31st January. Removal of the closed period and its replacement with the current NAP

regulations, (i.e. restrict applications based on soil moisture and rainfall conditions), will not sufficiently mitigate the risk associated with slurry applications during this period.

Report - The Application of the SWAT Model in the Upper Bann Catchment in Northern Ireland

- This study used the R^2 and Nash-Sutcliffe efficiency (NSE) to evaluate the model performance in predicting discharge. The closer NSE is to 1, the more accurate the model is. The accepted model should have a NSE of over 0.5 in predicting discharge.
- The final calibrated modelled NSE was 0.76 and 0.70 at BN (mid-catchment) and RM (headwater) respectively during the calibration period (daily). During the validation period the NSE metric achieved 0.81 at BN and 0.76 (RM). Comparing the observation and simulation means, SWAT underestimated the discharge at RM (Table 5 in the report). The results did improve further down the catchment at BN compared to RM where the discharge was slightly over-predicted (bias -6.8%). The results during the validation period were encouraging as they were as least as good as the calibration period results in most cases. The results indicate the importance of using accurate rainfall observations to drive hydrological models and also that model results tend to improve with increasing catchment size, especially when running on a daily timestep.
- In terms of loads the model predicted 1.92 kg P /ha/year (total P) from the outlet, of which 0.75 kg p/ha/year was in the form of soluble P during the calibration period (2004-2013). In comparison, the observed load at the outlet was 1.10 kg P/ha/year TP and 0.56 kg P/ha/year SRP, which is less than the model predicted (bias = 74% TP and 34% SRP).
- The model performed satisfactorily in predicting discharge but not in terms of predicting TP concentrations once the improved rainfall data set was used. SRP concentrations were also predicted reasonably well by SWAT in this catchment.

Report - The Riparian Ecosystem Management Model (REMM) to Evaluate the Environmental Farm Scheme Buffer Strip Designs

- This study used the Riparian Ecosystem Management Model to evaluate the effectives of the 4 buffer strips designs offered to farmers in the EFS scheme. The study was carried out using the Upper Bann catchment.
- The REMM model evaluated the impact of soil type, slope and buffer strip design of sediment and phosphorus loss at field scale in the Upper Bann catchment.
- Across all of the soil groups the 10-metre grass buffer performed the best in relation to phosphorus reduction. This was followed by the performance of the 10-metre tree buffer. On soil typology 1 and 4 (SWG1NT and SWG1ST), there was a difference of 5% between the grass and tree vegetation at a 10-metre width. This was larger for SBPGN and HRS, with a difference of 14% and 17% between 10-metre grass and 10-metre tree. Table 2 in the report presents the phosphorus reduction total percentages for each of the buffer designs and soil types, on a medium slope (6%). Reduction in TP loss varies significant between soil types, from as low as 4% to a maximum of 40%. (See table 2 in the report)

- For sediment reduction, across all the soil typologies, the buffer design which resulted in the greatest decrease was the 10-metre tree buffer, with an average of 49%. The design which resulted in the lowest decrease in sediment was the 2-metre grass buffer (average 41% across soil types).
- REMM was also used to evaluate the impact of different slopes on sediment and nutrient loss from different buffer strip designs and soil type. For soil typology 1, the lowest slope (1%) resulted in the greatest decrease in phosphorus in three of the four buffer designs (2-metre grass, 10-metres grass and 10-metre tree). The 2-metre tree buffer saw the greatest reduction on a medium slope (6%). This result is unique to soil typology 1, as the other soil groups show the highest phosphorus reduction to occur on the medium and high slopes (6% and 11%).(See table 3 in the report).

10.9 EU Exceptional Adjustment Aid (EAA) Soil Sampling and Analysis Scheme 2017-2018

The Exceptional Adjustment Aid (EAA) Soil Sampling and Analysis Scheme, funded by the EU, was delivered by AFBI to farm businesses across Northern Ireland. This was one of the largest soil testing schemes ever carried out within the British Isles and provided soil test data for some 20,000 fields on more than 1000 farms in Northern Ireland. Notably, ~100,000 fields, representing >20% of the total stock of intensively managed grassland fields in Northern Ireland, were actually offered by farmers for sampling in the scheme – i.e. 5 times the number that actually could be funded. This indicates the potential of such schemes to quickly shift attitudes towards soil testing by grassland farmers and compares with the prior situation in which less than 2% of grassland fields in Northern Ireland were being sampled regularly. Furthermore, 70% of the farmers who registered for the province-wide component of the scheme, subsequently attended a training course on nutrient management planning, again illustrating the potential of such schemes to markedly and rapidly shift attitudes towards nutrient management planning. The ambition is that the need for soil testing to underpin sustainable farm nutrient management, which is implicit in Northern Ireland's NAP, becomes fully accepted and embedded within the grass-based livestock industry.

The scheme had two components; an "**Open Scheme**", to which all livestock farmers in Northern Ireland were eligible to apply, and a "**Catchment Scheme**", targeted at farmers within specific geographical areas of the Upper Bann river catchment.

Thanks to the Open Scheme, which covered 12,218 fields on 522 farms across all counties in Northern Ireland, individual breakdowns of the distributions of soil P, K, pH and lime requirement statuses of grassland are now available for representative samples of farms in each of the three major ruminant livestock sectors – dairy, beef and sheep.

Results from the Open Scheme, are assumed to be representative of all farmed grassland, and suggest that 43% of grassland (excluding rough grazing) across Northern Ireland is under-limed with a total lime requirement of 1.2 million tonnes, requiring an expenditure of £30 million. Correcting this soil acidity problem could potentially increase grass DM production in Northern Ireland by some 1.73 million tonnes over the next 5 years, with a feeding value worth up to £216m *(£125/t DM*), and thus representing an almost 7 fold return on the lime investment. As expected, the dairy sector was shown to have the most acute P problem, with 50% of grassland fields over-supplied with P (Index ≥ 3). But beef and sheep sectors also have significant P problems, with 40% of fields oversupplied in both Lowland and Disadvantaged areas (DA), and 30% over-supplied in Severely Disadvantaged areas (SDA). From anecdotal discussions with farmers at nutrient management training sessions, it seems that long-held (*but now out-dated*) views by beef and sheep farmers, concerning the continued need for chemical P fertilisers on grassland, may be responsible for sizeable areas of grassland remaining P enriched despite adequate manure-P resources being present on most farms to meet crop P requirements.

(a) Colebrooke and Strule Soil Testing and Training Initiative (2018-2019)

The Colebrooke and Strule Soil Testing and Training Initiative (CSSTTI) was designed to increase the use and awareness of soil sampling as an essential farm nutrient planning tool and to build on the results of the EAA Soil Sampling and Analysis Scheme completed in 2018. The Initiative, funded by DAERA, was delivered by AFBI and targeted sub-catchments of the Colebrooke and Strule river systems – the geographic location of the catchment schemes is illustrated in Figure 51 (a). These sub-catchments were chosen to further analyse the relationships found in the Upper Bann sub-catchments in regions of differing geology, topography and farming activity. Integration of a training component on farm nutrient planning and interpretation of soil analytical results was again a key element of this initiative, with CAFRE providing free courses for all participants.

The Initiative proved very successful with 578 farmers participating (289 in both the Colebrooke and Strule regions) representing 70% and 60% of the agriculturally declared land in the Colebrooke and Strule respectively. The results from the sub-catchments indicate that over supply of plant available soil P is an issue in both areas, with 33% of total tested fields (equating to 41% of the land area) being above Olsen P Index 2 in the Colebrooke and 50% (equating to 49% of the land area tested) above Index 2 in the Strule. Figure 51 (b) illustrates the distribution of Olsen P results in terms of the recorded farm enterprise sectors. It will be observed that the Dairy sector in both catchments has a significant issue with 60% of tested fields being above Olsen P Index 2. In general, beef enterprises appear to be less intensive in the Colebrooke in terms of available soil P levels, with only 23% of the tested fields being above Index 2 compared to 42% in the Strule, with a similar trend being observed in mixed livestock enterprises, 38% in the Colebrooke and 55% in the Strule. However, soil P oversupply is still a significant issue in the non-dairy livestock sector, where these enterprises account for 58% of the land area which tested above Index 2 within the 2 sub-catchment areas.

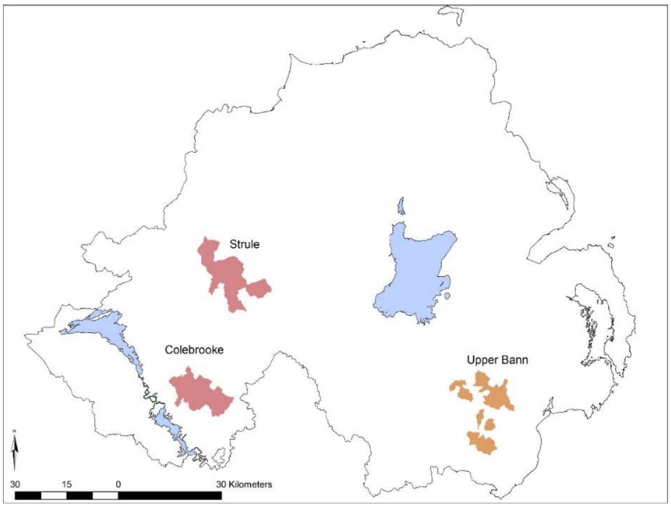
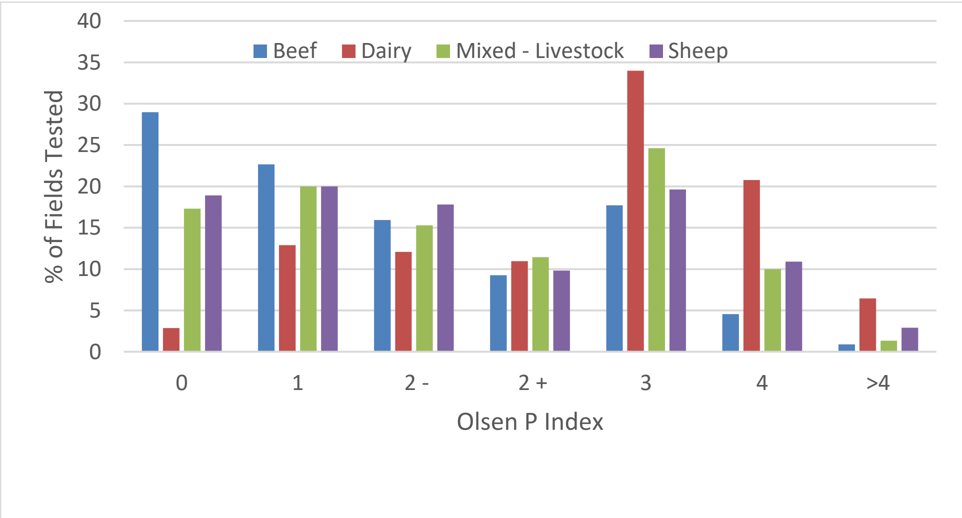
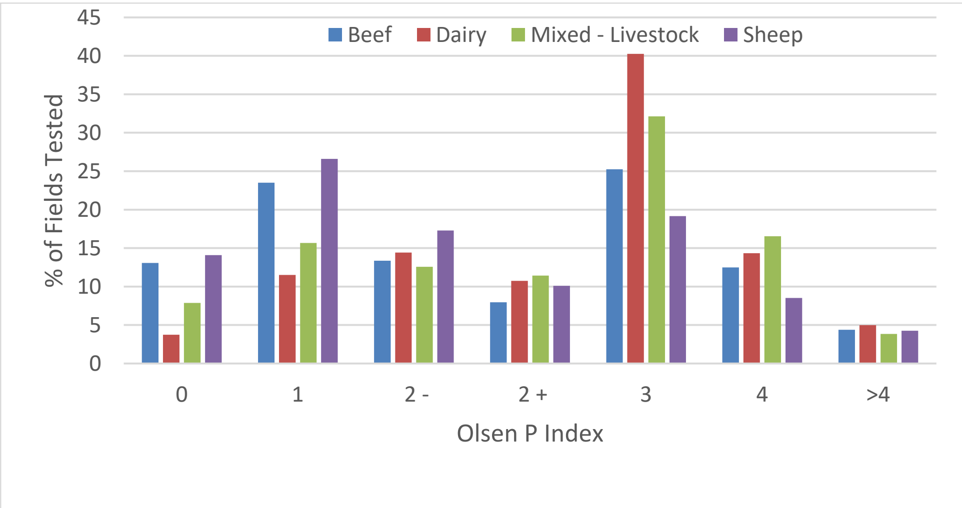


Figure 51 (a): Location of the Strule, Colebrooke and Upper Bann sub-catchments.



Colebrooke sub-catchments



Strule sub-catchments

Figure 51 (b): Distribution of Olsen P soil test results with reference to the recorded farm enterprise type in the Colebrooke sub-catchments and the Strule sub-catchments.

10.10 – E&I project 18/4/01 - Mechanical processing of farm slurries and anaerobic digestate to facilitate nutrient redistribution through export of separated solids

The objectives for this project are being addressed in four WPs as follows:

- **WP1**: A desktop study and report on the economics and logistics of mechanical separation of slurry and digestate.
- **WP2**: Examination of separated solids treatments for export off farm.
- **WP3**: Evaluation of the effect of slurry application rate and type on agronomic performance and environmental risk to water and air quality.
- **WP4**: Final report and two peer reviewed publications.

Results to date;

WP1: A wide-ranging literature review on solid-liquid separation of slurries and digestates was completed and reported in the 2018 Derogation Report.

WP2: Fuel performance. Screw press separated slurry and digestate solids were both pelletised and also processed via agglomeration (Fig. 51) to produce fuel feedstock. Calorific values, combustion performance and emissions testing were carried out in a multi-fuel biomass boiler to determine how separated solids feedstock compared to commercial wood chip and wood pellet fuels. Table 28 contains laboratory Proximate and Ultimate analyses of the biofuels and suggests that the separated solids fuel had a lower calorific value and higher ash content than the woody biofuels. The much higher ash contents in the biosolids fuel (22 wt%) was related to high levels of Na, K, Ca, Al, Fe, Cu and P. High ash content containing alkali metals in the separated solids can affect many aspects of the combustion process by impeding air-flows, reducing combustion temperatures and causing fouling, corrosion and breakdowns in boilers and this requires further investigation. Ideally, the nitrogen content (N wt%) of fuel should be no higher than the threshold value of 0.3 wt%. In general, it is widely observed that bio-waste has higher nitrogen content between 0.2 - 6.1 wt%. The nitrogen content of the separated biosolids fuel was 1.69 wt% and as such could lead to an elevated concentration of flue gas NOx during combustion compared to the wood fuels which would be undesirable. The separated solids had a lower calorific value (GCV) of 16.6 MJ/kg compared to standard wood fuels however the energy content was still 86% that of woodchip.

For the combustion experiments, the separated biosolids fuel achieved a high combustion temperature of 421°C (similar to both the wood chip and wood pellets) however combustion completed more rapidly than with the wood fuels. This would be expected given the lower energy density due to the higher ash content. Overall combustion efficiency was similar for all fuels tested. For separated biosolids combustion, the NOx, sum of both nitrogen monoxide (NO) and nitrogen dioxide (NO2), showed very similar concentration to the NO, indicating that NO was the main component in the NOx emissions for all the tested fuels. This would have been directly related to the raw fuel N content and affected by the combustion temperature. The NO concentration was significantly higher for the separated biosolids (202 ppm) than wood chip (124 ppm) and wood pellets (70 ppm). The flue gas particulate matter (PM) gravimetric emission results did not show any significant differences between any of the PM classes. This indicated that the boiler flue gas particulate reduction system, a series of 10 turbulators, worked at a high level of efficiency and was the primary effector of regularity of these emissions.

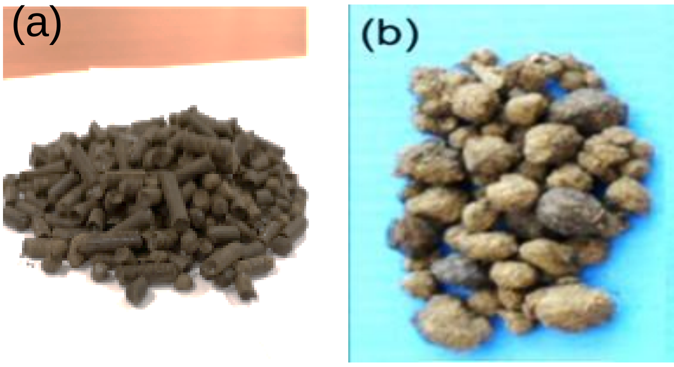


Figure 52: Pelletised (a) and agglomerated (b) digestate solids.

The fact that fuel produced from separated biosolids had similar PM emission levels to woody fuels is encouraging. Though actual grate ash was not measured, visual inspections of the ash chamber during and after combustion, showed much higher volumes for bio-waste fuels than those for the wood fuels. Unlike the ash collected from the wood fuels, the ash collected from the bio-waste fuels was slagged and very hard to break, which could be due to the use of binders as well as the ash constituents. However, the ash-related problems might be improved by co-firing with wood fuels in the further applications.

Table 28: Proximate and Ultimate (mg/kg) analyses of separated solids and woody fuels.

Parameter	Separated Solids	Wood Chip	Wood Pellets	$\bar{x}$	*sd*
DM (wt%)	85.4	86.4	91.9	86.5	2.8
GCV (MJ/kg)	16.6	19.4	19.9	18	1.4
Ash (wt%)	22.2	1.9	0.6	12.8	8.7
N (wt%)	1.69	0.4	0.1	2.3	1.6
C (g/kg)	352.4	445.3	451.3	395.2	39.8
Aluminium	50462.2	18.4	47.5	230.6	142.1
Calcium	17353.3	5091.9	791.4	13659.7	8162.8
Copper	227.5	3.5	1.4	81.5	76.6
Iron	1384.7	52.6	82.4	595	447.1
Potassium	10583.3	2363	520.1	17588.5	13508.7
Sodium	6578.2	294.8	134.2	3198.7	2288.3
Phosphorus	7875.9	848.1	103.6	7230.6	5466.1

(DM = dry matter; GVC = gross calorific value; Ash = ash content; N = nitrogen; C = carbon; $\bar{x}$ = mean; *sd* = standard deviation)

Based on the current value for biomass energy in the heat market, calculations (Table 29) can be made based on the inherent gross energy embedded within the bio-waste fuels. The poorer combustion properties of the biosolids (with reference to the higher ash and emissions) will result in restrictions on what technologies this fuel can be combusted in, which will have a negative effect on the attractiveness of the fuel and hence its financial value. Therefore the values in Table 29 need to be considered in the context of a joined up supply and utilisation chain and contract, potentially resulting in a significantly lower price due to reduced use competition.

Table 29: Potential economic value based on energy content.

Fuel	Source	Heat Energy kWh/kg	GCV MJ/kg	MC %	Value / tonne
Wood Pellets	Commercial	4.80	17.28	7.25	£185
Digestate Agglomerates	AFBI	4.61	16.596	0%	£163
Digestate Pellets	AFBI	4.84	17.424	0%	£171

WP3: Agronomic performance of separated liquid fractions. A rainfall simulation trial was established in order to assess the agronomic performance and environmental impact of using mechanically separated slurries and digestate relative to nutrients supplied via inorganic fertiliser at two different agronomic rates equivalent to $33m^3$/ha and $50m^3$/ha. The rainfall simulation trial run-off boxes were used to assess both (i) nutrient uptake and herbage response to application of separated liquids and (ii) potential impact on water quality through simulated application of separated liquids to grassland. The rainfall simulation trial used soil of Index P 2^+ and K 2^-, with a tetraploid variety of perennial ryegrass seeded as the yield indicator crop.

Sixty four boxes (Figure 53) were each packed with soil, and tamped down to achieve a soil bulk density of 0.9 g/cm^3 per box. Grass seed was sown in July 2019, along with a basal dressing of inorganic N, P and K as per RB209 recommendations for grass establishment. Following a 6 week grass establishment period and harvest, representative samples of cattle slurry (dairy and beef), pig slurry, anaerobic digestate and the liquid fractions of separated slurry and digestate were collected from AFBI Hillsborough and treatments were broadcast at different rates, $33m^3$/ha, $50m^3$/ha, representing typical and maximum slurry application rates. Additionally an N, P, K fertiliser control was applied, at rates equivalent to average N and P loading supplied by the slurry applications, yielding 8 treatments in total (control, inorganic fertiliser, dairy slurry, beef slurry, anaerobic digestate, separated anaerobic digestate, separated dairy slurry, pig slurry). Run-off boxes were maintained at field capacity via regular watering and weeds were removed periodically from boxes by hand. Grass was harvested at the end of the experiment and nutrient off-take and dry matter yield calculated. Herbage response was compared to the fertiliser control to assess relative availability of N, P and K in slurry and processed slurry treatments.

The effect of fertiliser treatment on dry matter (DM) yield and nitrogen (N) off-take was examined. Initial herbage data analysis showed that grass DM yield and N off-take varied between treatments, with the highest overall yield from chemical fertiliser (2473 kg DM ha^{-1}) at the average agronomic rate (33m^3/ha equivalent). The low yield was a reflection of a newly established sward, autumn sowing/third cut, with a similar yield to other published data. Manures, both processed and unprocessed, produced significantly lower DM yields and N off-takes than the equivalent rates of chemical fertiliser, with dry matter (DM) yields from boxes treated with mechanically separated digestate peaking at just 866 kg ha^{-1}. However, the initial results also indicated that separated anaerobic digestate and dairy slurry, produced higher yields of DM than un-separated AD or dairy slurry, as is evident from Figure 54a – albeit a full statistical analysis has yet to be conducted.

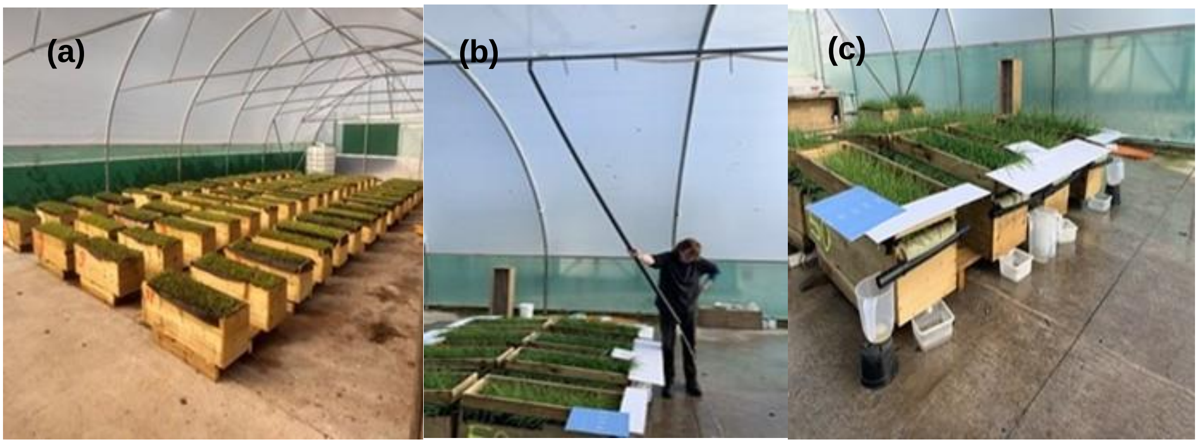


Figure 53: Established rainfall simulation trial, AFBI Loughgall, August 2019 showing 64 boxes (a), staff calibrating the rainfall rig flow rate (b) and rainfall simulation with runoff and leachate collection (c).

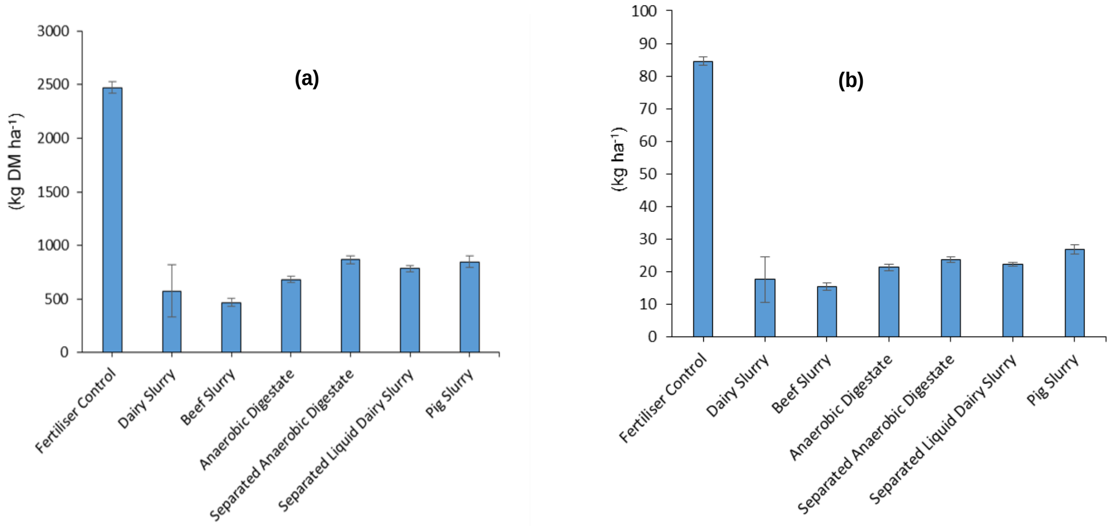


Figure 54: (a) Average of DM yield (kg DM ha^{-1}) and (b) average of N off-take (kg ha^{-1}) on a DM basis respectively, from one seasonal application of chemical fertiliser (control), anaerobic digestate, whole and separated slurries.

Furthermore, this effect of separation is also evident in the N off-take data, with greater N offtake on plots treated with separated anaerobic digestate or dairy slurry treatments relative to their unseparated counterparts (Figure 54b). These increases in DM yield and N-offtake were due to the N in separated treatments being more readily available to plants than in the unseparated treatments. Further investigation of P and K offtakes across different rates will be carried out, along with an assessment of nutrient availability and agronomic performance relative to both the chemical fertiliser control and between straight and processed (separated) organic fertilisers.

Full experimental results will be delivered in the 2021 Derogation Report, including the results of rainfall simulation, effects on water quality (surface run-off and leachate) and herbage (N, P, K offtake and yield) data.

WP4: To be completed by 2021.

10.11 – E&I Project 18/4/02 - Assessing biosecurity implications of the treatment and export of slurry and manure and identification of best practices

The slurry processing methods studied in this desk-based project fall into three main groups; anaerobic digestion (AD), storage methods (including aeration and additives) and separation. Four pathogens were considered; *Salmonella* spp., *Clostridium botulinum*, *Escherichia coli*, and *Mycobacterium bovis*. In addition, the presence, persistence and mobility of antibiotic resistant determinants was examined. The methods and pathogens considered were chosen, in consultation with experts, to have maximum relevance to the Northern Ireland agri-food sector.

Methods:

<u>**WP1** a systematic literature review on the biosecurity implications of slurry processing methods</u>

Literature searches were conducted for each combination of slurry separation method and pathogen using the Web of Science platform including all available databases. Literature from sources that were not peer-reviewed (e.g. conference abstracts or grey literature) were excluded. Article abstracts were screened to reject irrelevant studies, with remaining articles examined in full to extract relevant information.

<u>**WP2** a Rapid Risk Assessment approach applied to biosecurity risks in the processing and transport of slurries, building on information synthesised in WP1</u>

For the purposes of this assessment a similar framework to Rapid Risk Assessments, as laid out by the European Centre of Disease Control (ECDC) (Leitmeyer *et al.* 2011), was applied. This approach was deemed the most appropriate as the purpose of this methodology is to use readily available information to describe risk whilst also recording levels of uncertainty. The RRA was carried out in 5 stages:

1. Defining Risk Questions
2. Collecting and validating event/situational information
3. Systematically collecting information and extracting relevant evidence
4. Appraising evidence
5. Estimating risk

WP3 Costs of solids treatment via lime stabilisation

This work package gives a summary of the costs, mechanisms, logistics and practicalities of solids treatment via lime stabilisation. This was assessed in line with fixed and mobile plant apparatus, including kit, mobilisation, power, licencing and regulations.

Results:

WP1 Of the three processing methods reviewed, separation was found to have the least associated literature. In addition, there were no papers assessing the impact of slurry separation on *Clostridium botulinum.* The evidence for the effects of separation on *Salmonella* and *E. coli* and antibiotic resistant genes (ARGs) was mixed, with some evidence showing that survival through the process is possible.

There were more publications found associated with slurry storage methods than for separation. However, many of these studies were carried out in the Americas, and consequently, most focussed on processes not common in NI, such as lagoon storage, stacking and composting. Again, there were no papers assessing the impact of slurry storage on *Clostridium botulinum*. Generally, papers relating to slurry storage and both *Salmonella* and *E. coli* cite temperature as an important factor in bacterial kill. However, it should be noted that other, less understood, processing parameters also have an influence here, though the mechanisms of interaction with bacteria remain unclear. Literature relating to slurry storage methods and AMR (antimicrobial resistance) is complicated and varied. Generally speaking, more evidence is presented that swine slurries may be a greater risk to the maintenance of antimicrobial resistance within the farm than cattle slurries, though available research is limited. Additionally, from this literature, composting appears to have the greatest potential for the reduction of ARGs (antibiotic-resistant genes) and AMR bacteria. However, variation in the methods used between these studies means little replication has been carried out in observing specific phenomena, therefore the drawing of robust conclusions here is not possible.

Of the three slurry processing methods addressed in this review, anaerobic digestion (AD) had the most associated literature. Again, temperature was observed to be the important factor in achieving reduction or removal of *Salmonella* and *E. coli*. Where mesophilic AD was observed to reduce or eliminate detectable *Salmonella* and *E. coli*, thermophilic methods were found to be more effective, removing detectable pathogens in less time when compared to mesophilic processes. A limited number of papers describe the survival of *Clostridium botulinum* bacteria, spores or neurotoxins. Two studies report thermophilic AD has the ability to reduce *Clostridium botulinum* bacteria to below detectable levels, whereas the evidence for mesophilic methods is mixed with some potential shown for the survival of this bacteria. Only one study addresses the survival of neurotoxins and spores through AD, identifying potential for both of these factors to remain viable through the AD process. The evidence available for the effect of AD on AMR bacteria and associated elements is varied. Results from these studies show some AMR elements to increase, others to decrease and some to exhibit no changes. It is therefore hard to draw any clear conclusion.

It should be stressed that given the paucity of literature available for many of the processes considered, caution should be taken in drawing any strong conclusions from this body of evidence. This work could serve to highlight areas where further R&D is required to provide more practical evidence for the safe processing of slurries and address gaps in information for policy development.

WP2. Where information was lacking from this systematic review (WP1), reference was made to other studies and disciplines (predominantly food sciences) with the aim of gaining a better understanding of the dynamics of target microorganisms through their observation in a different context. Taking into account this additional data source, a Rapid Risk Assessment was carried out to produce the following risk estimation matrix:

Method	Evidence supports pathogen kill	Evidence confirms/suggests potential for pathogen survival	Residual risks	Options for mitigation
Separation	None	*Salmonella*, *E. coli*, *Clostridium botulinum* spores	There is no evidence available for the response of *Clostridium botulinum* bacteria & toxins, MAP or *Mycobacterium bovis*, however, it would be expected that risks associated with these hazards remain. Cannot ensure kill of *C. botulinum* spores[1]	Heat, or UV or ultrasonic treatment could be applied prior to separation Liming of separated solids will effect a pathogen kill but will also cause increased ammonia emissions All of the above will be additional costs to the farmer/processor
Storage	Some evidence of decline of inactivation of *Salmonella*, *E. coli*, MAP, *M. bovis*[2]	*Clostridium botulinum* spores	Pathogen kill can only be ensured by exposure to heat for sustained periods of time.	Validation of process conditions for pathogen inactivation. Monitoring of temperatures throughout the

			Decline in pathogens variable depending on the storage conditions There is no evidence available for the response of *Clostridium botulinum* bacteria & toxins Cannot ensure kill of *C. botulinum* spores[1]	process to ensure they remain within critical parameters to provide greater assurances of biosecurity.
Mesophilic AD	None	*Salmonella*, *E. coli, C. botulinum* bacteria and MAP have been shown to survive *Clostridium botulinum* spores	There is no evidence available for the response of *C. botulinum* toxins, or *M. bovis*, therefore cannot be estimated Cannot ensure kill of *C. botulinum* spores[1]	Heat treatment prior to digestion, or of the digestate following AD process
Thermophilic AD	*Salmonella*, *E. coli, C. botulinum* bacteria, MAP, *M. bovis*	*Clostridium botulinum* spores	There is no evidence available for the response of *C. botulinum* toxins, therefore the risk cannot be estimated Cannot ensure kill of *C. botulinum* spores[1]	Validation of process conditions for pathogen inactivation. Additional disease surveillance in animals from which feed stocks are derived.

[1]It is important to note that none of the above methods are likely to reach temperatures that ensure the inactivation of C. botulinum spores (e.g. 121°C for an appropriate period of time). We therefore suggest appropriate surveillance for this pathogen, and biosecure disposal of slurries from infected or high risk herds should be enacted.

[2]Evidence suggests inactivation of these pathogens, but this can only be assured where high temperatures are sustained for an appropriate amount of time.

WP3 Liming of slurry and digestate solids would appear to be a well tried and tested method for stabilising the organic material and rendering it safe in terms of pathogen spread. This methodology has been used successful in the waste water sludge sector enabling significant quantities of sewage sludge to be treated to compliance under the Safe Sludge Matrix and ultimately returned to agricultural land in accordance with the “Sludge (use in agriculture) Regulations 1989”. From an initial assessment of potential methodologies, it would seem that the most satisfactory and regulatory compliant method by which to lime treat solids would be by sub-contracting to specific waste management companies proficient in these activities; thus deploying lime stabilisation equipment at a number of farms. The farm feed mixer system is also tried and tested and could be applicable for a number of these larger farmers also however the environmental regulations and requirements within the "Site Specific Working Plan" will prove a challenge to businesses which have never considered lime stabilisation before. A further consideration will be emissions to air of this process. Increasing the temperature and pH of these waste materials will lead to the volatilisation and loss of potentially significant quantities of ammonia.

Further discussion:

Transport

The risks associated with the transport of slurries is very much context dependent. Factors such as the disease status of the animals producing the manure, the slurry processing method applied (if any) and the susceptibility and biosecurity standards on the recipient land will heavily influence the degree of risk. Appropriate awareness of factors such as these is necessary for mitigating potential for the spread of disease and AMR bacteria. It is also important to consider the transport of slurry processing machinery. This is particularly relevant in slurry separation, where it is increasingly common practice to move rented machinery between farms. Evidence shows the potential for slurry separation to concentrate AMR elements, potentially introducing these to novel environments.

Considering additional technologies

Considerations could be given to the adoption of biosecurity technologies used in the water industry for the control of microorganisms. For example, membranes can be used to offer microbial protection. Another consideration could be the use of thermochemical treatments, such as combustion, gasification or pyrolysis. These technologies would not only have the potential to provide biosecurity assurances, but could also provide low carbon and renewable energy, as well as resources for further circular bioeconomy.

Future avenues of research

This work has shown a stark lack of empirical evidence outlining the biosecurity risks in slurry processing, and subsequent transport. There is little information on the parameters controlling the survival and growth of key livestock pathogens within slurry, it is therefore difficult to give practical guidance as to safe processing limits. Further studies providing primary data on the inactivation of pathogens in slurries are clearly necessary. This body of evidence will likely require both laboratory experiments and on farm studies. Additionally, surveillance of the efficacy of processes such as mesophilic and thermophilic AD plants would be required on an individual plant basis. The testing of digestate for the presence of viable cells of pathogens or the presence of *Clostridium botulinum* spores or toxins is considered essential.

11. CONCLUSION

11.1 Derogation

In 2019, 441 farm businesses out of approximately 24, 594 direct aid claimants (i.e. 1.7%) operated under an approved derogation in Northern Ireland, compared to 471 (i.e. 1.9%) in 2018.

11.2 Water Quality

Nitrate concentrations in Northern Ireland surface freshwaters remain relatively low, with the average nitrate concentration for 99.8 % monitoring stations below 25 mg NO_3/l in 2019. Surface freshwater nitrate concentration trends indicated a decrease or stability at 58.1 % of sites across Northern Ireland between 2012-2015 and 2019. This is a reduction from 100 % in the 2016 report, 87% of sites in the 2017 report and 69% in the 2018 report.

The fact that nitrate levels have shown an increase in three consecutive years adds to the weight of evidence of deterioration in water quality, already indicated by increasing levels of phosphorus. In the Article 10 report for 2016, less than 2% of sites showed an increase in nitrate between the reporting periods (2008-2011 to 2012-2015). The recent increases at 41.8% of river sites adds weight to the cause for concern.

Groundwater nitrate concentrations across Northern Ireland are generally low. In the period 2012 to 2015, NIEA monitored nitrate concentrations at 56 groundwater monitoring sites across Northern Ireland in which the average nitrate concentration was 6.26 mg NO_3/l. In 2019, nitrate concentrations were monitored at 54 groundwater sites across Northern Ireland giving an average concentration of 6.58mg NO_3/l. In 2019, 52 out of 54 stations across Northern Ireland had nitrate concentrations in the 0-24.99 NO_3/l bracket. Therefore 2 of the 54 stations across Northern Ireland had nitrate concentrations over 25 mg/l NO_3/l. The two stations in the higher brackets (i.e. between 25->50mg/l NO_3/l and +50mg) are located in the high derogation catchment of Strangford. In the Strangford catchment 6 stations were used to calculate an average concentration for 2019 of 19.94 mg/l NO_3/l. The average concentration for the Strangford catchment 2012-2015 was 21.3 mg/l NO_3/l. From the stations monitored 1 station in the Belfast East groundwater body has had a nitrate concentration greater than 50 mg/l NO_3/l in 2012-2015, 2018 and 2019. This station is located in a former nitrate vulnerable zone before Northern Ireland was designated total territory. The station was purposely installed in that location to monitor groundwater quality in this area of arable farming.

Nitrate concentration trends in groundwater across Northern Ireland indicate a decrease or stabilisation in Upper Bann, Lagan, Blackwater, Ballinderry and Lower Bann derogation catchments in 2019 compared to 2012-2015. The Strangford catchment shows a slight decrease in average nitrates concentration in 2019 when compared to 2012 to 2015 for the Belfast East groundwater body and a larger decrease in average nitrates concentration in

2019 when compared to 2012 to 2015 for the Ards Peninsula groundwater body. Cumulatively these results from the two groundwater bodies give a modest overall decrease in nitrate for the Strangford catchment since the 2012-2015 period.

Phosphorus concentrations were assessed using current WFD standards for rivers and lakes. For SRP in rivers, 56.1% of sites were classed as either High or Good status. This has fallen since the baseline in 2012-2015, when 66.3% of sites were High or Good. Overall, 28.5% of sites deteriorated in WFD SRP class in 2019. In the middle and eastern parts of Northern Ireland the majority of catchments were classed as Moderate or Poor status.

Lakes have also shown deterioration based on TP concentrations with 17 of the 21 classed as Moderate or worse status in 2019, compared to 12 in the previous reporting period (2012-2014). The overall trend between 2012-2014 and 2019 in lakes was a deterioration in WFD TP class at 10 sites (almost 50%).

In this report, both the assessments using Nitrates Directive and WFD criteria show deterioration as indicated by the percentage of river and lake sites that are exhibiting increasing nitrate and SRP levels. This is a cause for concern, and adds to the evidence of the sustained upward trend exhibited in the SRP Water Quality indicator in the draft Programme for Government (PfG). This suggests that agricultural activities remain a significant and increasing pressure on water quality.

As previously highlighted, the 2019 results should be treated with a degree of caution as natural variation in nutrient concentration is expected year to year due to seasonal and climatic changes.

11.3 Advisory Support

As in previous years, DAERA delivered a number of training and advisory events for farmers across Northern Ireland and provided information and guidance to farm businesses using a wide range of media, including one to one advice for derogated farms, where requested. A Summary NAP 2019-2022 Regulations booklet was published and distributed to all farm businesses. Work commenced to provide updated guidance documents on the NAP 2019-2022 and derogation workbooks.

CAFRE has lead responsibility for the development and maintenance of a suite of five online calculators designed to help farmers to manage their farms to comply with various aspects of the NAP Regulations. These calculators were updated in line the NAP 2019-2022.

There are 24,500 farm businesses in NI of which 20% use online calculators. The total number of users increased by a further 8% in 2019/20. The calculators are available on the DAERA web-site at: www.daera-ni.gov.uk.

11.4 Compliance

Compliance observed during on farm inspections of selected derogated farms in 2019 showed 3 cases of non-compliance from 31 farms inspected. Two had water pollution incidents: one from their on-farm storage facilities, and one had insufficient storage; and the other farm had undertaken closed season spreading. Administrative checks on the 441 fertilisation accounts submitted online for the calendar year 2019 also indicated an increase in numbers of non-compliance compared to 2018. Non-compliances being attributable to having partial or zero records, P balance and Nitrogen loading exceedances. DAERA continues to review training delivery and provide information for farmers to help address these non-compliances.

11.5 Research and Monitoring

To underpin the implementation of the Nutrients Action Programme and Derogation for Northern Ireland (NI), the Agri-Food and Biosciences Institute (AFBI) has been carrying out a broad range of research studies aimed at understanding the sources, transportation and resulting impacts on aquatic ecosystems of farm nutrients. The research spans a continuum of temporal and spatial scales from short-term lab experiments to long term catchment monitoring programs, and includes the monitoring work within the Colebrooke and Upper Bann catchments which has been implemented specifically to meet the terms of the Derogation, and is reported in Sections 4, 5 and 7 of this report. The continued and expanding investment in monitoring and modelling studies is helping to improve our understanding of nutrient cycling/transport at soil, field, farm, landscape and catchment scales. Research is also focussing on water ecology, on the use and efficacy of Decision Support tools and associated models, high resolution water monitoring, and LiDAR-based run-off risk maps to facilitate reductions in nutrient entry to water bodies, and on technologies designed to reduce the potential for manure to pollute the environment or spread disease.

The fortnightly monitoring for SRP and TON in derogated and non-derogated sub-catchments in the Upper Bann catchments, over each hydrological year (from 1st October), show a number of differences. For SRP, concentrations were higher in the derogated catchment, with median concentrations of 114 ug/L in 2016/17, 98 ug/L in 2017/18 and 111 ug/L in 2018/19 compared to 99 ug/L, 89 ug/L and 86 ug/L, respectively, in the non-derogated catchment. These differences were, however, not significant. For TON, median concentrations in the derogated catchment were 1.61 mg/L in 2016/17, 2.04 mg/L in 2017/18 and 2.52 mg/L in 2018/19 compared to 1.29 mg/L, 1.32 mg/L and 2.08 mg/L, respectively, in the non-derogated catchment. In all years the derogated catchment had higher recorded concentrations but these were only significantly different in 2017-18. Further comparison is limited by the resolution of fortnightly data, which tends to miss short duration storm events and is biased toward low flows. These low flow conditions and concentrations, however, are those that in-stream organisms are exposed to for most of the time so their impact on ecological state may be greater than short duration, high concentrations storm events.

Time series from the higher resolution monitoring infrastructure provides greater insights into nutrient losses from the catchments and the ability to accurately estimate loads, with Figure 23 showing the patterns of loss in 7-hourly data for Total P (TP) and Total Oxidised Nitrogen (TON) from October 2018 to February 2020. Base flow concentrations of TP in the Derogated catchment are consistently higher than in the non-derogated catchment, although differences are less clear during storm event pulses. For TON the difference in concentrations between catchment is more pronounced and were particularly notable during the autumn/winter 2018/19 nitrogen peaks associated with the summer drought. Although concentrations of TP are higher in UB03 (Figure 23), flows are lower and so the load for the derogated catchment (UB03) is less, at 0.86 kg/ha, than in the non-derogated catchment (UB15a) which lost 1.31kg/ha over the October 2018-19 period. From October 2019 to the end of the first half of the hydrological year (1^{st} April 2020) the loads lost were broadly similar (1.03 kg/ha (UB3) and 1.10 kg/ha (UB15a), compared to the same period in the preceding year (01/10/2018-01/04/2019: 0.53 kg/ha (UB3) and 0.76 kg/ha (UB15a)). TON losses from the derogated catchment (UB3) were higher (23.42 kg/ha) compared to the non-derogated catchment (17.4 kg/ha) over the October 2018-19 period, with greatest differences in loading occurring during the autumn/winter period. The difference may however have been exacerbated by drier conditions in the derogated catchment during this period, increasing mineralisation of N and reducing denitrification.

Research on ecological recovery in these catchments found that the un-impacted conditions (as indicated by high BMWP scores) at the beginning of the 1990's have shown a marked decrease to BMWP scores indicative of moderately impacted system between 1995 to 2019. This is despite an improvement in nutrient concentration recorded in the streams. This apparent resistance within the community to changes in nutrient concentration, legalisation and management is currently being investigated further.

The long term monitoring programme of the Lough Neagh Rivers and lake has found that despite the lack of an overall trend in P catchment inputs Lough Neagh water concentrations remain high, likely driven by sediment P. River catchment inputs of N to Lough Neagh have decreased since the mid 1990's and were closely correlated with lake N concentrations. Dissolved organic N concentration (DIN) remains relatively low in Lough Neagh compared to values in the mid-1990s. The major drivers of primary production have changed across the time series. DIN has become more important in relation to phytoplankton biomass in recent years in Lough Neagh. The mass of P released from Lough Neagh sediment has increased since the mid-1990s. A study on the P in the Lough Neagh sediment found P released will continue to delay significant improvements in water quality for approximately 40 years following the reduction of external nutrient loading.

While long term monitoring on the Lough Neagh system, Colebrooke and Upper Bann catchment have provided invaluable insight in nutrient dynamic and impact in freshwater systems, work is on-going on the future of monitoring in these systems. To this end a number of key steps have been identified including the establishment of three mobile real-time water quality monitoring stations should be considered for short-term, roaming deployments across NI and used to demonstrate the contribution of different pressures to water quality.

In addition a number of fixed high-resolution monitoring stations should be established in key locations within major NI catchments should be considered for enhanced surveillance monitoring and as catalysts for engagement and knowledge transfer. These monitoring stations should be linked with bottom-up engagement and behavioural change aimed at reducing the impact of land use practices on aquatic ecosystem.

Work continued on identifying and mitigating the sources of impact on aquatic ecosystems, with an evaluation of the current evidence base highlighting that it is currently the most effective strategy for minimising the risks associated with slurry spreading from 15th October to the 31st January. Removal of the closed period and its replacement with the current NAP regulations, (i.e. restrict applications based on soil moisture and rainfall conditions), will not sufficiently mitigate the risk associated with slurry applications during this period. The EAA and CSSTTI nutrient management scheme have highlighted the extent of the challenges with over 38% of soil above the agronomic optimum level of soil P required to support grass production. Continued application of slurry to these soil, poses significant long term risk for water quality. To address this there is a need to export slurry off farms where necessary, with work on-going to identify the best technology and assess the biosecurity implications of the treatment and export of slurry and manure