An Roinn Talmhaíochta,
Bia agus Mara
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Evaluation of the Green Low-Carbon Agri-Environment Scheme (GLAS)

Evaluation of the impacts of the GLAS on Diffuse Agricultural Pollution

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Executive Summary

The Green Low Carbon Agri-Environment Scheme (GLAS) is a measure funded by the Rural Development Programme (RDP; 2014 to 2020). It promotes agricultural interventions which introduce or continue to apply agricultural production methods that aim to address the issues of climate change mitigation, water quality and the preservation of priority habitats and species. The objective of this study was to assess the effectiveness of GLAS as a contributory measure towards sustainable Irish agriculture and to fulfil, in part, Ireland's commitment towards the monitoring and evaluation requirements set out in the RDP. The study was focussed on the effect of GLAS on nutrient (nitrate and phosphorus) and sediment losses in runoff to rivers and lakes, and the emission of climate change gases (nitrous oxide and methane). This study has used computer models of pollutant emissions from agricultural land and the effect of changes in land management to forecast the potential long-term effect of GLAS management interventions.

The study covers the whole of Ireland, with results reported for each of the c. 3,200 Water Framework Directive (WFD) river waterbodies. Spatially distributed GLAS agreement data was provided for tranche 1 and tranche 2 (which closed for entry at the end of 2015), which included 280,000 different interventions (from a choice of over 20 actions in GLAS) on almost 40,000 different holdings. Tranche 3 of GLAS, which occurred after this project started, has increased the number of agreements by approximately one third.

The computer modelling approach applied used the PSYCHIC model (Davison et al., 2008) for phosphorus and sediment; N-CYCLE, NITCAT and MANNER models (Scholefield et al., 1991; Lord, 1992; Chambers et al., 1999) for nitrate and tier one and two IPCC methodology for methane and direct nitrous oxide gas emissions using country specific data on productivity and manure management (Duffy *et al*., 2016). These models were integrated in a single modelling framework that enabled sharing of assumptions regarding environmental pressures and farm management practices, and the models were adapted to share a common water balance and drainage pathway calculation. The model outputs were integrated with a common landscape connectivity and delivery model to calculate the proportion of surface runoff and entrained pollutants that were delivered to watercourses.

To facilitate the modelling, a number of key spatial environmental datasets were created as part of this project, including meteorological and soil datasets and also the land connectivity dataset, which was based upon the number and permeability of field boundaries.

The modelled annual average national diffuse agricultural pollutant losses, before the impacts of GLAS were accounted for, were 128 kT for nitrate, 2.4 kT for total phosphorus, 672 kT for sediment, 36 kT for nitrous oxide and 527 kT for methane.

This modelling approach provides an explicit and quantitative apportionment of baseline pollutant emissions, by source, land area, means of mobilisation and delivery pathway to waters. This apportionment shows that grassland is the major source of losses for all pollutants except methane, reflecting the fact that grassland is the dominant agricultural land use. For nitrate, the soil, fertiliser, manure and excreta sources are all relatively important (15% to 30% of the total). Soil and excreta are the most important sources for phosphorus emissions (c. 40% each). The majority of nitrous oxide emissions come from either fertiliser (30%) or excreta at grazing (50%). For methane, the majority of emissions are enteric (83%). The majority of nitrate is lost through leaching to groundwater (75%), with surface runoff relatively unimportant (5%). For phosphorus, the

contributions from surface runoff is greater (15%) but the main pathways are preferential flow (through drains) and direct excreta in water (i.e. livestock having access to water whilst grazing or traveling to or from the yard). Preferential flow is the dominant pathway for sediment transport (68%) with the remainder transport through surface runoff and no losses due to leaching. The importance of the preferential pathway for phosphorus and sediment means that emissions are concentrated in areas where field drains have been installed.

Verification of modelled predictions of nitrate and phosphorus loads was undertaken using observed loads for the period 2011-2013 for the 16 monitoring sites used for OSPAR reporting. Modelled loads did not include non-agricultural sectors or account for in-channel retention, but still achieved a good agreement with observed N loads (r^2 of 0.67). The agreement between modelled and observed phosphorus loads was lower (r^2 of 0.42) due to the greater variation in the contributions from non-agricultural sources and the potentially large impact of retention in the catchments with large lakes.

GLAS contains over 20 actions, some of which are mandatory depending upon the entry tier, the size of the farm or stocking density on the farm or if any of the priority environmental assets are applicable to the farm. Those actions that impacted on diffuse agricultural pollution were characterised as a percentage reduction in the pollutant emissions from one or more of the sources, land areas or delivery pathways. The magnitude of the reduction was determined either from computer models or a synthesis of published experiment data. The level of implementation of these actions was calculated by farm type and WFD waterbody using the holding level scheme agreement data for the *c*. 40,000 different holdings in tranches 1 and 2 of GLAS. Modifications to the impacts of the GLAS actions were made to reflect actual management changes rather than those implied by scheme descriptions, using the results of the GLAS attitudinal survey (Cao et al., 2018). Uptake of the actions ranged from Protection of Watercourses from Bovines (12% of grassland next to water); Catch Crops (10% of spring Cropping); Wild Bird Cover (7% of Spring Cropping) and Low Input Permanent Pasture (6% of grassland) to 2% or less for other actions (see table above). Given these low uptake rates, only a few GLAS actions are likely to be important for reducing pollutant losses at national scale, although all actions may be more important locally. A summary of the representation of GLAS actions is included as an Annex to this report.

The results showed that although approximately 32% of all agricultural land is managed by farms in GLAS, the percentage of the national pollutant load occurring from this land is only 25-28% for nitrate, phosphorus, nitrous oxide and methane. The values are lower than the proportion of land for these pollutants because dairy farms, which typically have the highest pollutant load intensity are less likely to be in GLAS.

Table ES-1: Overall uptake rates used in the modelling work, derived from data provided for tranches 1 and 2 of GLAS, expressed as a percentage of the applicable area in the whole of Ireland, calculated for those GLAS actions that were assumed to impact on sediment and nutrient losses.

Category	*Description*	*Applicability*	*Applicable Area ('000 ha)*	*Action Area[4] ('000 ha)*	*Uptake (%)*
Habitat Actions	*Arable Grass Margins (buffering)*	*Arable*	*370*	*3.1*	*0.8*
	Arable Grass Margins (land out of production)	*Arable*	*370*	*0.1*	*0.04*
	Environmental Management of Fallow Land	*Arable*	*370*	*1.3*	*0.4*
	Farmland Habitat (Natura)	*Arable*	*370*	*0.5*	*0.1*

Category	Description	Applicability	Applicable Area ('000 ha)	Action Area[4] ('000 ha)	Uptake (%)
	Farmland Habitat (Natura)	Grassland	3,764	89.6	2.4
	Low-input Permanent Pasture	Grassland	3,764	235.4	6.3
Pollutant Abatement Actions	Protection of Watercourses from Bovines[1]	Grassland next to water	2,009	249	12.4
	Riparian Margins	Grassland next to water	2,009	0.9	0.05
	Catch Crops	Spring Crops	200	19.1	9.5
	Minimum Tillage	Arable	370	6.3	1.7
	Low-emission slurry spreading[5]	-	-	-	-
Landscape Actions	Planting New Hedgerows - (Suspended under Tranche 2 & 3)	Arable	370	3.2	0.9
	Planting New Hedgerows - (Suspended under Tranche 2 & 3)	Grassland	3,764	55.8	1.5
Species Actions	Breeding Waders[2]	Grassland	3,764	1.0	0.03
	Chough[2]	Grassland	3,764	10.0	0.3
	Corncrake[2]	Grassland	3,764	0.0	0.003
	Curlew[2]	Grassland	3,764	2.0	0.1
	Geese and Swans[3]	Spring Crops	200	1.0	0.5
	Grey Partridge	Arable	370	0.4	0.1
	Grey Partridge	Grassland	3,764	0.8	0.02
	Hen Harrier[2]	Grassland	3,764	36.0	1.0
	Twite A[2]	Grassland	3,764	3.0	0.1
	Twite C	Grassland	3,764	0.0	0.0
	Wild Bird Cover	Spring Crops	200	13.7	6.8

1 Includes the fencing aspect of Riparian Management
2 Represented as LIPP in modelling
3 Represented as Catch Crop in modelling
4 Some actions measured in length (e.g. protection of watercourses from bovines) have been converted to area of fields impacted, using the LPIS field area where each action was located
5 No evidence for changes in nutrient use, so no impacts calculated. Would have impacts on ammonia losses, but these were outside the scope of the modelling project.

Nationally, the estimated reductions in long term annual average pollutant loads from farms in GLAS are 10% for phosphorus, 6% for nitrate, 5% for nitrous oxide, 3% for methane and 4% for sediment (see table below). When diluted by the unaffected loads from non-scheme farms, which occupy over two thirds of the agricultural area, the national impacts are 3% or less. There are significant spatial variations in the reductions achieved, with reductions in some catchments estimated to be over 15% even when the contributions from agricultural land not in GLAS are accounted for. For nitrate, phosphorus, nitrous oxide and methane, overall reductions are greatest in the central / western catchments which correspond to areas of high GLAS uptake. Sediment impacts are higher in eastern areas where there is more tillage land. There are significant differences by farm type, with larger (3-

15%) reductions on Specialist Dairy farms and small reductions (<1%) on Specialist Sheep farms. The major cause of these reductions is the Low Input Permanent Pasture action (and the comparable Natura Habitat and Farmland Bird actions), which have the highest uptake rate of all the actions and which were assumed to reduce fertiliser inputs and livestock (particularly on Specialist Dairy farms), thus controlling pollutant losses at source rather than trying to mitigate mobilisation or delivery. Wild bird cover and catch crops were effective actions for reducing sediment.

Table ES-1: Percentage reduction in annual average agricultural pollutant loads, due to GLAS actions. Results show reductions in the pollutant loads from just the farms in GLAS and from all farms in Ireland.

Farms	Nitrate	Phosphorus	Sediment	Nitrous Oxide	Methane
Only those in GLAS	5.7	9.6	3.7	4.7	3.0
All	1.5	2.7	1.2	1.3	0.7

Contents

1 Introduction

The Green Low Carbon Agri-Environment Scheme (GLAS) is a measure funded by the Rural Development Programme (RDP; 2014 to 2020). It promotes agricultural interventions which introduce or continue to apply agricultural production methods that aim to address the issues of climate change mitigation, water quality and the preservation of priority habitats and species. The objective of this study is to assess the effectiveness of GLAS as a contributory measure towards sustainable Irish agriculture and to fulfil, in part, Ireland's commitment towards the monitoring and evaluation requirements set out in the RDP. The focus of this study is the effect of GLAS on nutrient (nitrate and phosphorus) and sediment losses in runoff to rivers and lakes, and the emission of climate change gases (nitrous oxide and methane). Air quality (ammonia) and chemical (pesticides and herbicides) impacts are out of scope, although the methodology can be extended to include these and other impacts.

The Common Monitoring and Evaluation Framework (CMEF) of the RDP contains a set of Common Evaluation Questions, each of which is answered by the calculation of indicators relating to the base environment situation, and scheme inputs, outputs, results and impacts:

- Scheme input indicators concern the budget or resources allocated and are not of interest here, but can be determined from scheme records;
- Output indicators measure the activities directly realised through the agri-environment schemes. Reported indicators include the number of farm holdings and the physical area of land receiving support under the agri-environment schemes. They are the first step towards realising the objectives of the RDP;
- Result indicators measure the direct and immediate effects of the schemes, and include the land areas under *successful* land management that contribute to an improvement in environment quality; *and*
- Finally, impact indicators measure the improvement in environment quality that has demonstrably occurred.

Our approach in this work is to use computer models of pollutant emissions from agricultural land and the effect of changes in land management to forecast the potential long-term impact of GLAS management interventions. Specifically, we use computer models to quantify the proportion of the baseline total pollutant load that is managed by farms in scheme, that part which is potentially controllable by the selected management interventions, and the likely reduction in load on the assumption of best practice.

The computer modelling approach enables an explicit accounting of the spatial variation in agricultural intensity and soil / climate factors that control baseline emissions and the efficiency of the selected interventions. The computer modelling allows calculation of impact of the net contribution of individual and groups of interventions, allowing an assessment of their relative merits and the benefits of targeted uptake. The approach also allows a projection of the net impact in advance of the requirement for extensive (and expensive) river and groundwater monitoring to detect ecosystem response, and is not subject to the variability in weather that hinders change detection. The computer modelling framework, including a spatial database of land management and scheme interventions, also provides a single coherent source for calculation of simpler output and result indicators, including the area and nutrient inputs managed by farms in scheme.

Most importantly, the computer modelling approach presents an explicit and quantitative disaggregation of baseline pollutant emissions, by source, land area, means of mobilisation and delivery pathway to waters. This allows stakeholders transparent access to our assumptions regarding the relative importance of the sources and pathways affected by land management interventions, the contribution from the non-agricultural sectors, and hence the likely limits to the scheme effect and the anticipated effect size that environment monitoring schemes must be designed to detect.

We are specifically concerned with the following Common Evaluation Questions (CEQs) under the Rural Development Programme (2014-2020):

- FA-4B – To what extent have the RDP interventions supported the improvement of water management, including **fertiliser** and pesticide management?
- FA-4C – To what extent have RDP interventions supported the prevention of **soil erosion** and improvement of soil management?
- FA-5D – To what extent have the RDP interventions contributed to reducing **greenhouse gas** and ammonia emissions from agriculture?

1.1 Modelling Methodology

The methodology is based on Anthony *et al*., (2008; 2009) and Anthony *et al*., (2012) who developed and proved a generic methodology for calculating the effectiveness of mitigation methods for control of diffuse pollution. The methodology involves the derivation of a meta-model of export coefficients from the output of more detailed process based models applied to common descriptions of farm systems that are representative of typical practice.

The process models employed are:

- **PSYCHIC** – Phosphorus, Sediment and Water Balance (Davison *et al*., 2008)
- **NITCAT, NCYCLE** and **NEAPN** – Nitrate (Lord, 1992; Scholefield *et al*., 1991; Lord and Anthony, 2000)
- **IPCC Tier 1** and **2** – Nitrous Oxide and Methane (IPCC, 2006)

Each model has been previously used at catchment and national scale for policy support and has been adapted to share common farm management data inputs. See Section 3.3 for more detail. The models are spatially explicit, driven by data on local soil and climate conditions affecting the generation of runoff and drainage and the mobilisation of pollutants.

Each model has been adapted to share a common water balance and drainage pathway calculation based on the PSYCHIC model to ensure consistency of results. The models have also been integrated with a common landscape connectivity and delivery model (see Section 2.4) to calculate the proportion of surface runoff and entrained pollutants that are delivered to watercourses, to help represent the sensitivity of mitigation to the location of risk activities. This spatially explicit model takes account of field locations and boundary types.

Each model has also been adapted to output an explicit partitioning of total pollutant emissions for a common coordinate system of source types, source areas and delivery

pathways found on a farm. For example, incidental soluble phosphorus in surface run-off following spreading of dairy slurry to grassland on a dairy farm is explicitly recorded as:

- **Farm Type**: Dairy
- **Pollutant**: Phosphorus
- **Source**: Dairy (Animal)
- **Area**: Grass
- **Pathway**: Runoff
- **Type**: Slurry
- **Form**: Soluble
- **Timescale**: Incidental

Numerous combinations of coordinates are recognised (see Section 3.2) and form the basis for representing the effect of mitigation actions, which are defined as auditable on-farm activities to reduce pollution risk, such as change in the quantity or timing of fertiliser applied. The GLAS actions considered in this project were mapped to one or more mitigation actions (see Section 4).

Individual mitigation actions are characterised as a percentage **effectiveness** or reduction in the pollutant emission from one or more source coordinates. The effectiveness is determined either from computer models or a synthesis of published experiment data. A mitigation action is also characterised by a percentage **applicability**, which measures the proportion of the source area that the method is applicable to. For example, sowing of over-winter cover crops is only applicable to the arable land area where spring cereals will be sown. Measures of applicability were derived for each farm type for each WFD waterbody as required for each mitigation action.

The effect of any mitigation action at farm and catchment scale depends on the level of uptake or **implementation** and the magnitude of the pollutant emissions at the target coordinates relative to total emissions from the whole of a farm system. Implementation rates were derived from GLAS agreement data, with values derived by farm type for each WFD waterbody.

The individual process based models are first applied at farm scale for eight representative farm systems, using management information drawn from national government statistics (see Section 3.1). The descriptions of the farm systems include data on the timing and location of fertiliser inputs and livestock grazing, and explicitly account for gaseous nitrogen emissions in housing, storage and at spreading before calculation of leaching.

Simulations are carried out for every farm type for every soil series found in each WFD waterbody, using local climate information. Export coefficients are derived that express the modelled pollutant emissions as a linear function of the potential pollutants input to the farm system in the form of fertiliser and livestock excreta. In a deviation from typical export coefficient models, emissions are also expressed as a function of the land area where it is necessary to represent pollutant sources that are intrinsic or respond slowly to reducing

inputs, such as the nitrate emissions sourced from the background soil organic nitrogen supply rather than fertiliser applied. The export coefficients derived for soil series in each WFD waterbody are area-weighted to derive a single set of coefficients for each WFD water body and for each farm type. The coefficients are therefore spatially explicit, and sensitive to local environment conditions affecting pollution risk.

The export coefficients from all models are then combined to develop a single Framework Model of rules for calculating all pollutant emissions from farm inputs. At this stage, enhancements are introduced to represent the effect of localised soil management issues and additional point sources of pollution. By default, the source models assume that soils are not compacted or waterlogged. Using survey data on the extent of soil damage and more detailed computer modelling, modifiers are introduced to the rule base to increase water and gaseous emissions in affected catchment areas (see Section 3.4). Additional rules are also introduced to represent direct excretion by livestock into watercourses and runoff from farm tracks, based on livestock activity calendars and survey data on the number of unfenced fields and fords (see Section 3.1.4).

The Framework Model is used in combination with a spatially explicit database of crop areas, livestock numbers and inputs for each WFD waterbody to calculate pollutant emissions. This required the development of a map of agricultural census data, disaggregated by farm system type (See Section 2.5.1).

The development of the database of the targeting and effectiveness of mitigation actions, with which the impact of GLAS on pollutant losses were be derived, is described in Section 4, with a separate sub-section for each GLAS action that has a potential impact on the agricultural pollutants considered.

Section 5 describes the calculated baseline pollutant loads, with results shown nationally and by farm type and by WFD waterbody.

The calculated impacts of GLAS on these baseline pollutant loads are then shown in Section 6.

1.2 Geographical extent

The study covers the whole of Ireland, with results reported for each of the c. 3,200 WFD waterbodies (Figure 1-1). The reported pollutant losses are statistically disaggregated by representative farm type and land cover type (cultivated land, improved grassland and rough grazing). The WFD waterbodies range in size from just under 2 km^2 to 370 km^2, with an average area of *c*. 22 km^2.

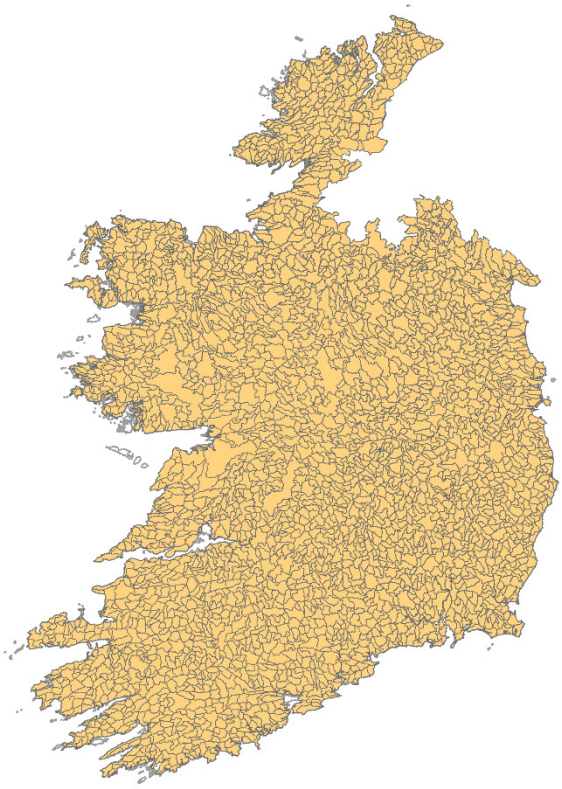


Figure 1-1 Catchments defined for Water Framework Directive reporting

2 Environment data

A number of key spatial environmental datasets have been created as part of this project in order to enable agricultural pollutant modelling. The creation of these layers are briefly outlined below.

2.1 Elevation and slope

30m resolution ASTER GDEM data (a product of NASA and METI) for Ireland were downloaded from the NOAA website (accessed 4 May 2016). The TIFF images were merged to create a raster dataset of elevation for the whole of Ireland. The Slope tool in ArcGIS was used to create a raster dataset of slope.

The average elevation and slope of each 1km^2 was calculated using Zonal Statistics in ArcGIS. To ensure that the average elevation and slope of coastal grid squares were calculated only for areas of land and did not include the sea, the grid squares were first clipped in ArcGIS to an outline of the Republic of Ireland downloaded from the Ordnance Survey Ireland website (Landmask, OSi National 250k Map of Ireland, accessed 6 April 2016), and zonal statistics were calculated for the areas remaining after clipping.

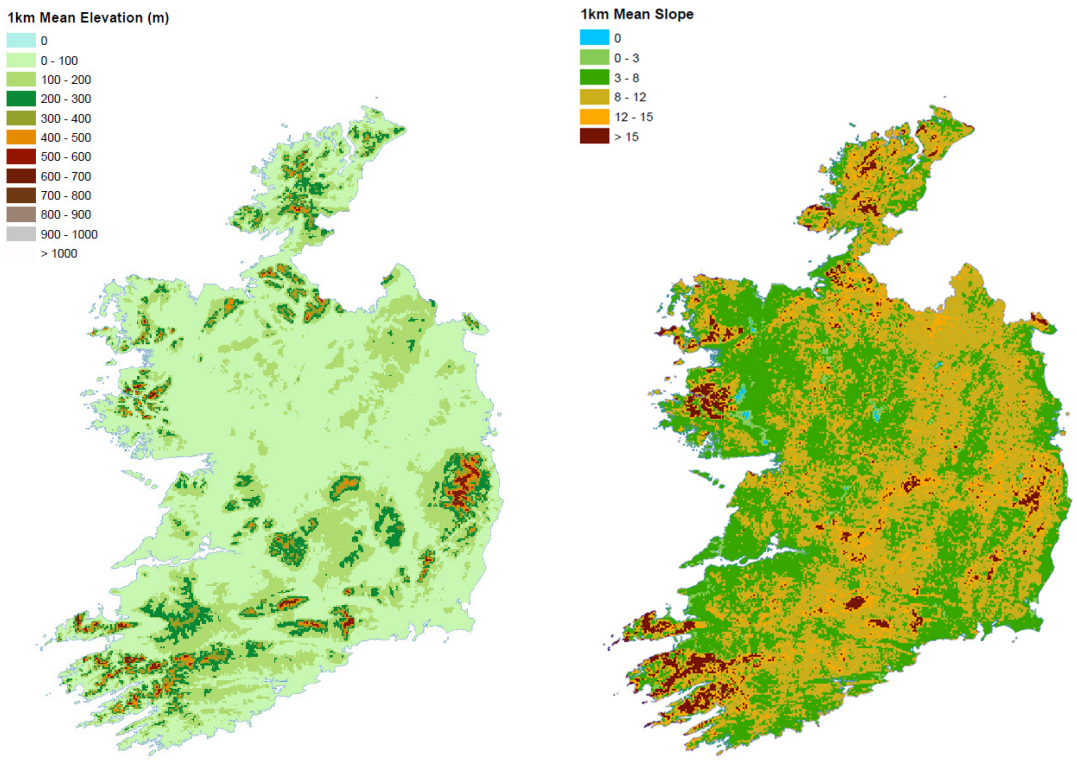


Figure 2-1 Mean elevation and slope calculated for each 1km^2 in Ireland.

2.2 Climate

2.2.1 Rainfall, sunshine and temperature

1x1km gridded datasets of average monthly rainfall, sunshine hours, and maximum and minimum temperatures for the period 1981-2010 were downloaded from Met Éireann (Walsh, 2012).

The reference points for each grid square were plotted in ArcGIS using Easting and Northing and transformed from Irish Grid coordinate system to IRENET 95 Irish Transverse Mercator. The Irish Transverse Mercator 6-digit Easting and Northing was rounded to the nearest 1000 and the centroid of each grid square found by adding 500 to the rounded Easting and Northing. Some grid squares (mostly coastal) were not covered by the 1x1km Met Éireann datasets, so the climate values for these squares were taken from the first nearest neighbour grid square. The mean distance to first nearest neighbour for these grid squares was 0.65km, and the maximum distance was 7.65km.

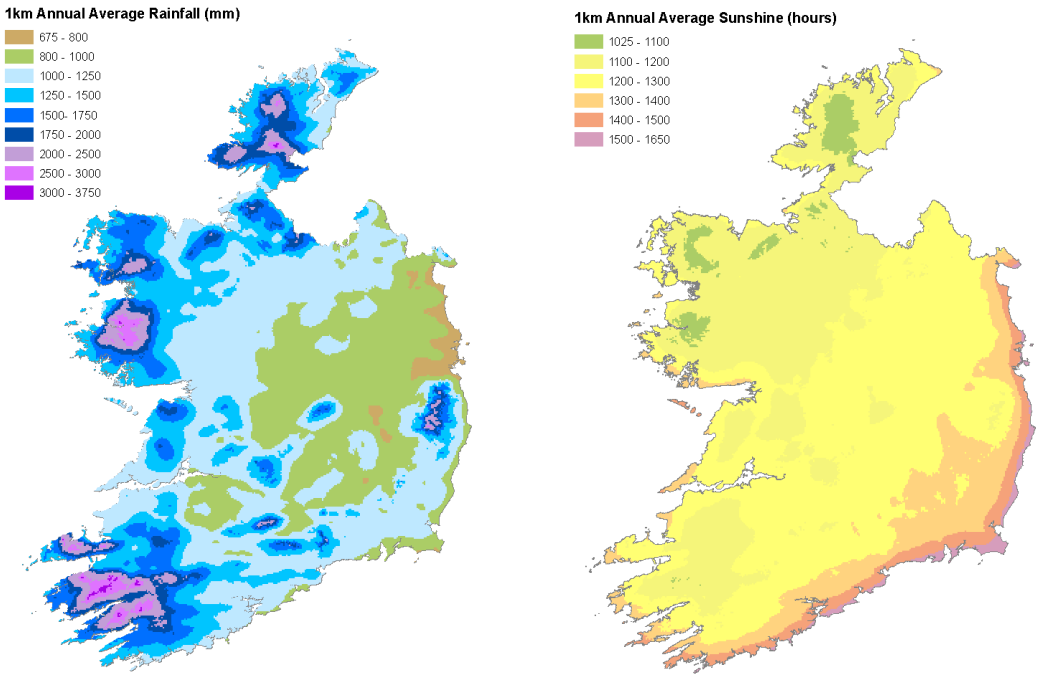


Figure 2-2 Annual average rainfall and sunshine hours for the period 1981-2010

2.2.2 Raindays

Data on daily rainfall at 25 weather stations across the Republic of Ireland were downloaded from Met Éireann. Some stations had data covering the period January 1986 – March 2016, whilst the shortest period of data ran from 2011 – March 2016. There was no rainfall data available for some days. The average proportion of days with rain in each calendar month was calculated by dividing a count of the total number of days in the month with rainfall > 0.1mm by a count of the total number of days in each calendar month with rainfall data (this could include data showing that no rainfall fell on a day). This proportion was then multiplied by 30 days to give a standard number of days for each month.

A regression model was fitted between the standardised monthly number of days with rain and average monthly rainfall using the function 'lm' in R from the 'stats' package (R Core Team, 2016). Out of a number of models compared, a quadratic model was found to give the highest adjusted R-squared and was used to calculate standardised monthly number of days of rain for each 1km2. Decimal number of days were used to give an average monthly number of days with rain. The fitted model was of the form:

$$R_d = a + bR_M + cR_M{}^2$$

where R_d is rain days and R_M is average number of average monthly rainfall.

Table 2-1 Summary of the fitted model used for calculating rain days from average monthly rainfall.

		Std. error	
a	7.68	0.76	
b	0.156	0.015	
c	-0.00035	0.00007	
Adjusted R-squared			0.73

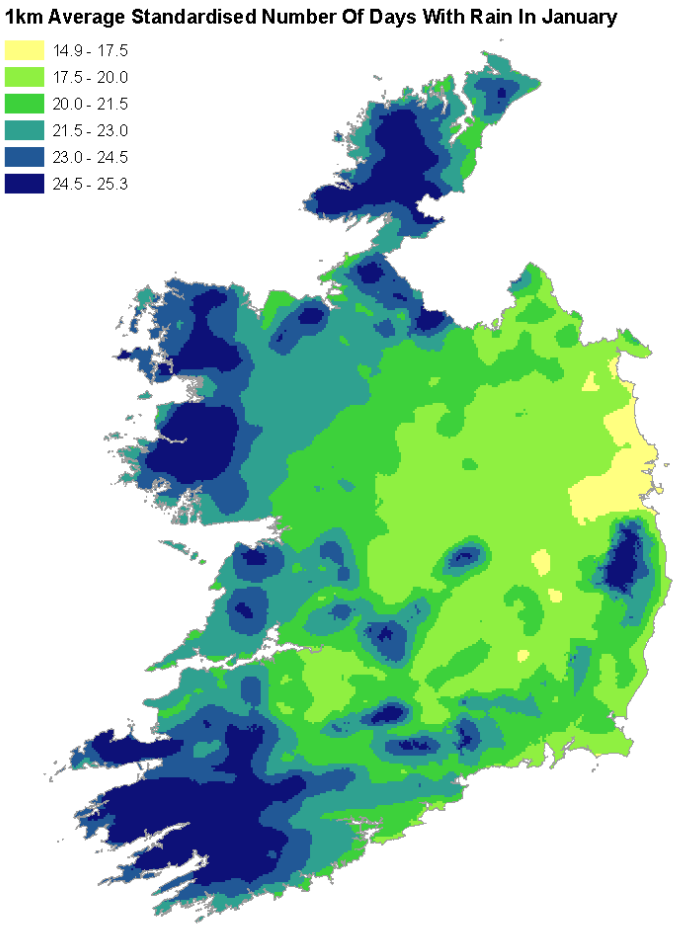


Figure 2-3 Rain days in January, dervied from data for 1986 – 2016.

2.2.3 Wind speed

Data on monthly average wind speeds at 23 weather stations across the Republic of Ireland were downloaded from Met Éireann. Some stations had wind speed data covering the period January 1986 - March 2016, whilst the shortest period of data ran from 2011 – March 2016. An overall average monthly wind speed in metres per second was calculated for each station using all available monthly data.

The distance of each weather station from the sea in km was calculated in ArcGIS using a polyline outline of Ireland.

Regression models between the log of monthly wind speed and distance from the sea were fitted using the function 'lm' in R from the 'stats' package. The fitted models were of the form:

$$log(W_b) = a + bD_s$$

Where W_b is a base average monthly wind speed (in metres per second) and D_s is distance from sea (in kilometres). Monthly values for the coefficients *a* and *b* are shown in Table 2-2. Base monthly average wind speeds were generated for each 1km^2. These were then adjusted for elevation using the altitude factor in the Draft Irish National Annex to Eurocode 1 (2009) to give the final predicted average monthly windspeed (W_M) for each 1km^2:

$$W_M = W_b(1 + 0.001E)$$

where *E* is elevation in metres.

Table 2-2 Summary of the fitted model for deriving wind speed.

Month	a	Std. error (a)	b	Std. error (b)	Adjusted R-squared
January	1.873	0.059	-0.00703	0.00173	0.44
February	1.806	0.058	-0.00668	0.00171	0.39
March	1.757	0.055	-0.00574	0.00162	0.34
April	1.682	0.044	-0.00581	0.00129	0.47
May	1.706	0.042	-0.00563	0.00124	0.47
June	1.573	0.044	-0.00618	0.00129	0.50
July	1.557	0.045	-0.00605	0.00133	0.47
August	1.590	0.047	-0.00604	0.00137	0.45
September	1.625	0.053	-0.00699	0.00157	0.46
October	1.713	0.060	-0.00713	0.00175	0.41
November	1.792	0.057	-0.00735	0.00169	0.45
December	1.882	0.057	-0.00779	0.00168	0.48

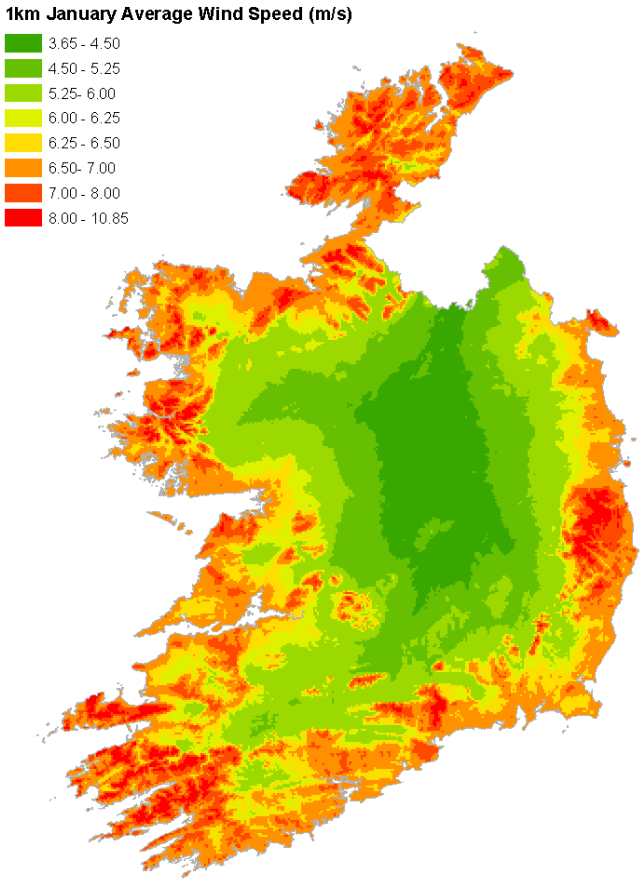


Figure 2-4 Wind speed in January, dervied from data for 1986 – 2016.

2.3 Soils data

A map of representative soil series was produced, based on matches between the Irish Soil Information System association map (2014) and the Teagasc-EPA Soils and Subsoils map of drainage categories (2009), such that a series mapped to a particular location was treated as representative of the drainage and texture properties of the soil in that location.

The Irish Soil Information System association map, at a 1:250,000 scale, was the most recent available work in soil mapping of Ireland, including data on soil texture and other properties for most soil series. The map shows the location of associations, each of which contains several series typically found in association with one another. These series may have different Great groups, texture and drainage properties. The Teagasc-EPA Soils, Subsoils and Wet/Dry maps, with a working scale of 1:50,000 – 1:150,000, were used as a source of further detail on the likely location of series within the association, based on their drainage properties.

Each series in the Irish Soil Information System was assigned a 'best match' to the IFS code in the Teagasc-EPA Wet/Dry map, based on Great group, acidity/basicity and soil depth. The soil was treated as shallow if the Irish Soil Information System description used the word 'over' (if over bedrock, gravels etc. but not over till), otherwise it was matched as a deep soil.

The Irish Soil Information System and Teagasc-EPA Wet/Dry map were intersected in ArcGIS, so that each area was defined by an association code and an IFS code. The series within an association are ranked according to how frequently they occur in the association. Each area was assigned the highest ranked (i.e. most frequently occurring) series within the association that had a 'best match' to the IFS code for that area. Where there was no 'best match' series for a particular association-IFS combination, the series from within the association that best matched the IFS code was selected based on great group, subgroup (indicating wetness and whether soil was humic/histic or not), soil depth and acidity/basicity.

Areas with association code 1xx were assigned to 'Peat' regardless of the IFS code for those areas, and divided into three types of peat (Raised Bog, Atlantic Blanket Bog, Montane Blanket Bog) using the Irish Peat Map (DIPM2) (Connolly and Holden, 2009). The Irish Peat Map is a raster grid at a coarser scale than the association map and did not cover all the areas with association code 1xx. Where this was the case, remaining areas with association code 1xx were filled with nearest neighbour properties (restricted to selecting Raised Bog or Atlantic Blanket Bog only).

Association codes not matching any series were assigned as follows:

Association code	**Series**
Water body	Water
Urban	Urban
0xx	Sands
Tidal marsh	Tidal Marsh
Salt marsh	Salt Marsh

Next any remaining unassigned areas of the following IFS codes (i.e. not matching one of the associations above) were assigned as follows:

IFS code	**Series**
Water	Water
Made	Urban
AeoUND	Sands
MarSands	Sands
MarSed	Tidal Marsh

The remaining unassigned areas with the association code 'Rock' or the IFS code 'Scree' were classified calcareous and non-calcareous rock categories based on their parent material (from the Teagasc-EPA Wet/Dry map).

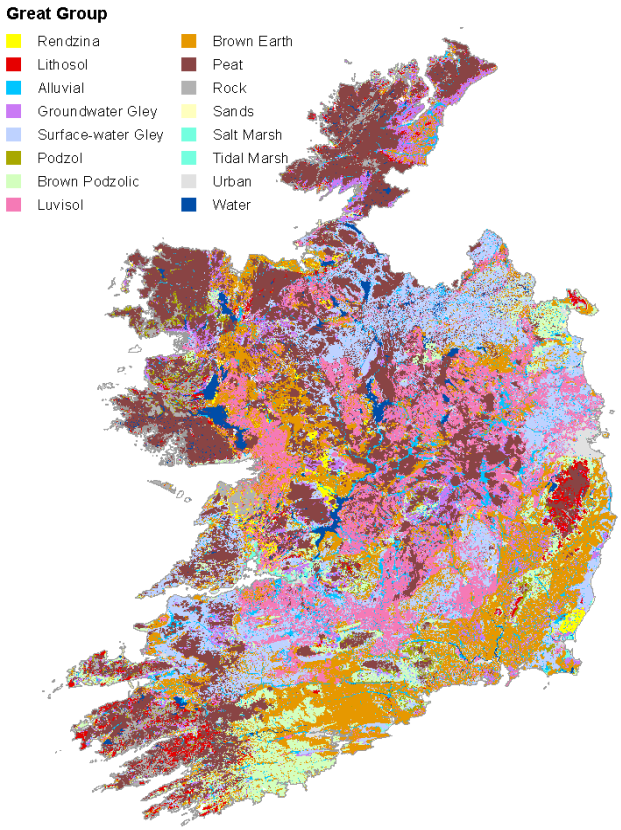


Figure 2-5 Great group assigned to each area of land

Some series in the Irish Soil Information System did not have associated texture (sand, silt, clay, organic carbon percentage content) information. Where there was another series with the same best-match IFS code in an association, the next highest-ranked series with the same best-match IFS code that did have texture information was used for that association.

If there was not another series with the same best-match IFS code in the association, the series with missing texture data was kept, and its texture properties were drawn from another series in the same Great Group with a similar texture description. To achieve this match, the following rules were used in order of priority:

- Same Great Group
- Same texture (e.g. loamy, fine loamy, silty)
- Same depth (shallow/deep i.e. whether description includes the word 'over')
- Same subgroup
- Acidic/basic properties match
- Underlying material (e.g. siliceous stones) matched as closely as possible.

The representative soil series and their associated properties were used to develop additional soil datasets. HOST class (Boorman *et al.*, 1995) was derived using the methodology of Schneider (2007), which was developed for use with the European Soils Database. Bulk density for each horizon in each soil series was calculated using the pedo-transfer functions for Ireland in Reidy *et al.*, (2016). Water capacities were derived using the pedo-transfer functions in DEFRA (2008), which extended the SEISMIC soils data to the whole of the UK. Maps of soil texture and HOST class are shown in Figure 2-6 and Figure 2-7.

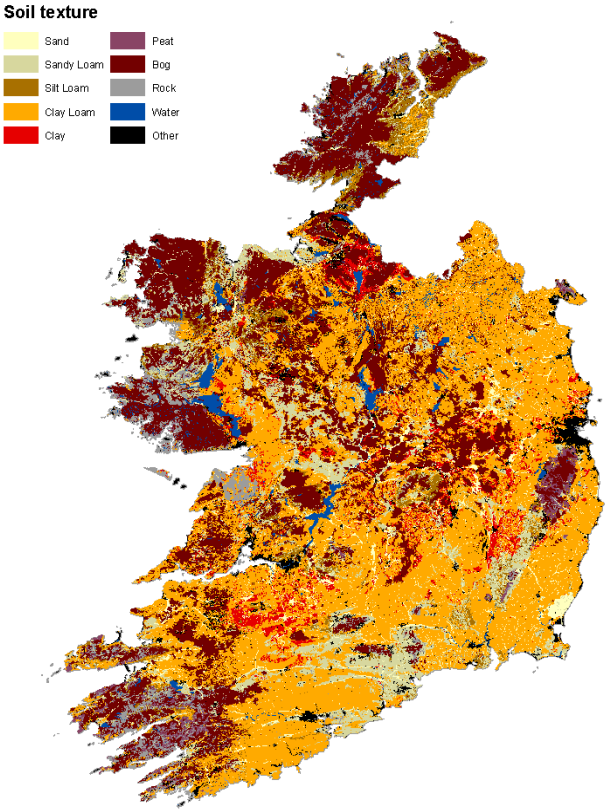


Figure 2-6 Top soil texture for each area of land

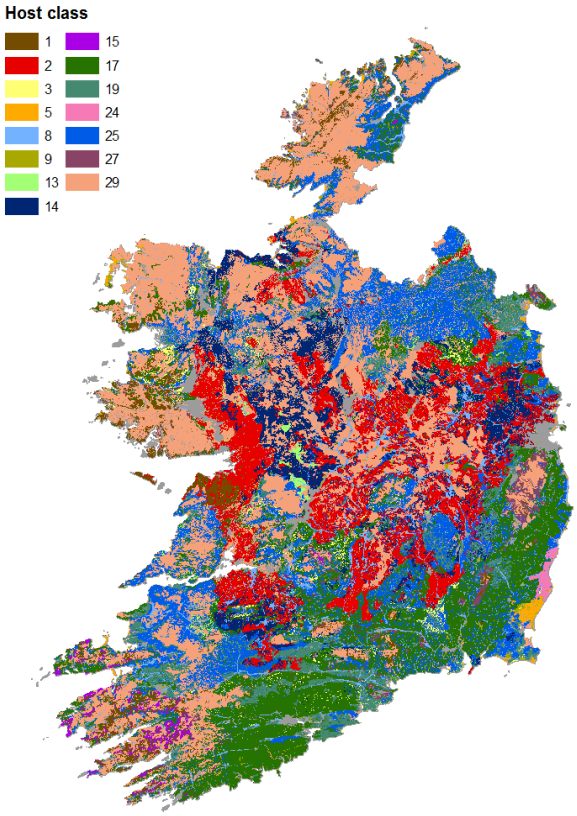


Figure 2-7 HOST category for each area of land

2.4 Landscape connectivity

The ADAS Framework Model, based on an integration of the PSYCHIC and NEAP-N models, calculates the volume of surface runoff and mobilised pollutant load for each field, taking account of local crop cover, soil erodibility and the soil moisture deficit at the time of rainfall (see Section 3.3). Only a proportion of the mobilised load will be delivered to the edge of the field where it might enter a natural watercourse or drainage ditch.

Transmission losses along the runoff pathway, caused by depression storage and infiltration of surface runoff or by the deposition of the suspended pollutant load at a break of slope, can be considerable. Walling and Zhang (2011), for example, measured gross and net soil erosion for 248 fields across England and Wales using the Caesium-137 radiometric technique to map the redistribution of mobilised soil. The within field transmission losses varied from less than 20% to more than 80% of mobilised material. At the edge of the field, surface runoff and the suspended pollutant load may also be halted or filtered by boundary features such as walls, hedge lines and grass strips. Runoff from a field that is located several field lengths away from a watercourse will have to break through several such boundaries to successfully deliver the pollutant load (assuming there are no field gates or tracks aiding delivery). Again, slope to channel transmission losses can be significant. Owens *et al*., (1997), for example, used the Caesium-137 technique to construct a sediment budget for the catchment of the river Start in Devon, and reported that 26% of the soil eroded from slopes was stored at intermediate locations, generally upslope of hedge boundaries.

The net effect of within field and slope to channel transmission losses is termed a connectivity ratio (Walling and Zhang, 2004). For national scale modelling, the connectivity ratio has been estimated to lie in the range 0.20 to 0.70, based on the knowledge of sediment budgets for small agricultural catchments in England and Wales. National maps of the ratio have been created by integrating spatial data sets of, for example, distance to river, slope shape and gradient, vegetation roughness and runoff to create a compound spatial index that is scaled to this range (McHugh *et al.,* 2002). This index was previously applied in the development of the diffuse pollution Screening Tool for Scotland and Northern Ireland (Anthony *et al*., 2006).

For this study, the opportunity was taken to create an integrated Framework Model for all pollutants to develop an enhanced connectivity index that more explicitly represents the within field and slope to channel transmission losses. The aim of this was to better represent the effect of mitigation actions that moved potential pollutant sources away from fields immediately adjacent to watercourses, and away from high-risk areas of steep slopes.

Within Field Transmission

The within field transmission is based on the concept of sediment transport capacity of surface runoff, which is influenced by slope gradient, and by runoff frequency and depth. Providing that the soil detachment capacity of rainfall and runoff is high, the transport capacity is the limiting control on export from a field and is represented in most soil erosion models. A large number of empirical transport capacity equations have been developed, largely based on measurements at plot scale, which are generally of the form:

$$f(q,s) \propto k \cdot q^{b} \cdot s^{c}$$

where *q* is the depth of runoff, *s* is the sine of the slope gradient, and *k* is a constant affected by particle size and density. The range of the coefficient b is 1.0 to 2.4 and of the coefficient *c* is 0.9 to 1.9 (Julien and Simons, 1985; Prosser and Rustomji, 2000). A median value of 1.4 is recommended for both coefficients (Prosser and Rustomji, 2000). Transport capacity increases with depth of runoff as the drag and lift forces for particle detachment and saltation increase, whilst steep slopes reduce depression storage, increasing the frequency of runoff, and are associated with more rapid flows and concentrated flow in rills or gullies that are more likely to reach the edge of a field and cross any physical barriers at the field edge.

Based on the transport capacity equation, a within field transmission index has been developed based on average field slope and soil class from the Hydrology of Soil Types (Table 2-3). Field slope is self-explanatory, and the transmission index should be calculated separately for each part of a field within the given slope ranges to develop an average index for a field. The Hydrology of Soil Types (Boorman *et al*., 1995) is a classification of soils based on their hydrological properties, including dominant flow paths. Each HOST class has been assigned a Standard Percentage Runoff (SPR) factor that is proportional to the magnitude of rapid response flow that occurs during rainfall events. The HOST classes have been ranked according to the SPR index. The ranking has been modified where a high SPR factor was due to subsurface flow via drains and macropores rather than as surface runoff. The effect is that the transmission index varies little with slope for free draining soils and varies by a factor of two for slowly permeable soils.

The within field transmission index was constrained to lie in the range 30 to 90%, based on the literature and experience from previous modelling studies. Zhang *et al.,* (2011), for example, measured gross and net soil erosion for 248 fields across England and Wales using the Caesium-137 radiometric technique to map the redistribution of mobilised soil. The within field transmission losses varied from less than 20% to more than 80% of mobilised material (Figure 2-8). The average value was in the range 40 to 60%, with higher values for arable land indicating that delivery of mobilised material was more efficient than on grassland.

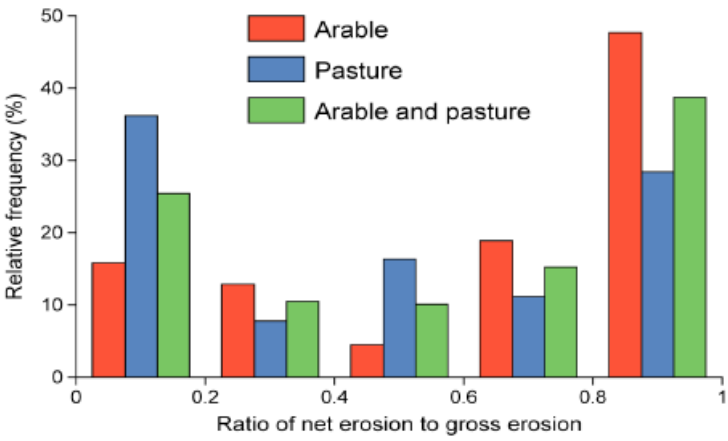


Figure 2-8 Frequency distribution for the ratio of net erosion to gross erosion for individual fields monitored using the Caesium-137 radiometric technique as part of Defra project SP0413 (Zhang *et al*., 2011).

Table 2-3 Within field transmission index (%) as a function of field slope and soil class from the Hydrology of Soil Types.

	Slope (Degrees)			
HOST Classes	<3	3 to 7	7 to 12	>12
1 2 4 13	30	30	30	35
3 5 6 14 16 17	30	30	35	40
7 8 9 10 11 19 22	35	35	45	55
12 15 18 20 21 23 24 25 26 27 28 29	40	45	65	90

Slope to Channel Transmission

Beyond the field edge, the probability of surface runoff reaching a watercourse is dependent on distance and the number and type of barriers encountered. To simplify matters, the conceptual model assumes that field boundaries are either 'taps' (natural watercourses or drainage ditches) or 'barriers' which can be either permeable (such as fences and grass strips) or impermeable (such as walls and hedge lines with banks), and ignores the potential for tracks and gateways to act as conduits for pollution. The probability of surface runoff reaching a tap, from where it enters the main river system, is dependent on the number and type of field boundaries encountered along the surface flow path.

Using the LPIS digital field boundary data and the EPA 1:50,000 river network, the number of fields that runoff would have to cross before reaching a natural watercourse was calculated using a Geographic Information System. Where a field is located immediately adjacent to a watercourse, the probability of the surface runoff entering the water depends only on the permeability of the field boundary. Where a field is located several fields distant from a watercourse, the probability of delivery depends on the permeability of all the intervening field boundaries. It is calculated as the product of the permeability coefficients for each boundary encountered.

The Land Use/Cover Area frame Survey (LUCAS) is a harmonised in situ land cover and land use data collection exercise that extends over the whole of the EU's territory (Toth *et al.*, 2013). As part of this survey, land use was recorded along 3,484 transects of varying length in Ireland for 2012, from which the type of field boundaries could be derived. For this study, it was assumed that fences, banks and grass strips, and remnant hedges are permeable, with a permeability factor in the range 80 to 100%. Walls, trees and shrubs, and hedges are impermeable, with a permeability factor into the range 0 to 20%. A separate permeability factor was calculated for field boundaries next to water and for boundaries between fields.

A connectivity index was calculated for each LPIS field. For use in the modelling framework, and an area-weighted value derived for all fields on the same soil series within a WFD waterbody. Figure 2-9 shows an area-weighted connectivity index for each WFD waterbody.

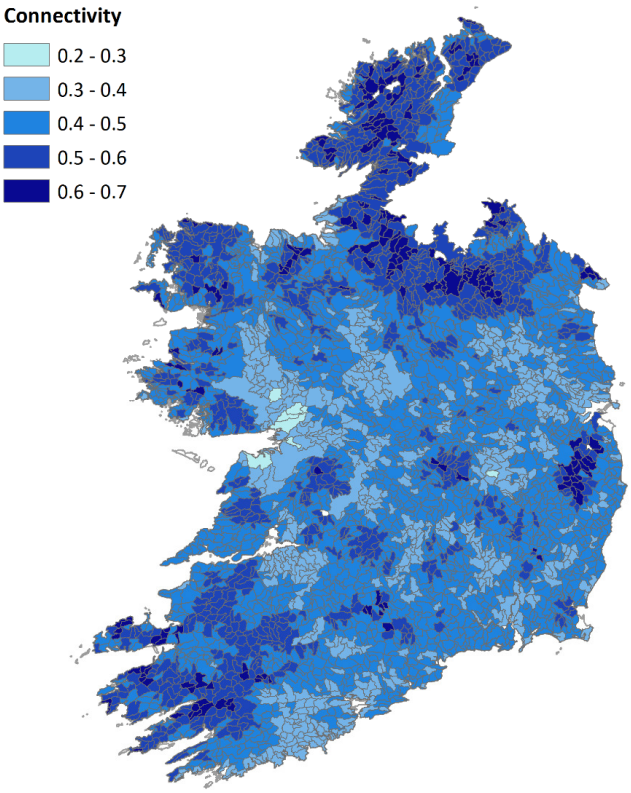


Figure 2-9 Average calculated connectivity for each WFD waterbody.

When calculating the probability of delivery to a tap, fields that were artificially drained were also assumed to be adjacent to a watercourse, even if not found to be so from the river line data. This is because fields with these systems would also have drainage ditches that would act as artificial watercourses and provide a rapid pathway to the main river channel. If a field is tile drained, then surface runoff is not required to cross intervening fields to reach a watercourse. The probability of a field being tile drained is based on a grouping of the HOST class of the soil as used in the PSYCHIC model (Davison *et al.,* 2008).

Table 2-4 HOST classes which are assumed to be artificially drained (Davison *et al*., 2008)

HOST Classes	Drained if used for arable crops	Drained if used for grassland
1-8, 11-13, 15-17, 26-29	No	No
9, 10, 14, 18- 22	Yes	No
23-25	Yes	Yes

2.5 Land Cover and Land Use

A land cover database for Ireland was produced using a combination of Land Parcel Information System (LPIS) parcel data, and other land cover data layers. The database gives hectares of arable, grass, rough grazing, woodland, peatland bog, other vegetated areas,

urban, rail and road network, other non-vegetated areas, freshwater, and saltwater land cover in each Water Framework Directive waterbody.

The LPIS parcel shapefile covers much of the agricultural land in Ireland, but inclusion of other data layers was required in order to map non-agricultural land, and land in agricultural use that is not included in the LPIS dataset. Note that the non-agricultural land was summarised at WFD waterbody scale (the final scale of the modelling) and not mapped in detail. A number of additional land cover data layers were acquired to provide information on land cover in areas not covered by LPIS parcels:

- Ordnance Survey Ireland National 250k Map of Ireland
    - Built Up Areas
    - Airfields
    - Lakes and Reservoirs
    - Rail Network
    - Roads
- EPA 1:50,000 river network map
- Derived Irish Peat Map Version 2
- CORINE Land cover 2012 – National

The EPA 1:50,000 river network map was used to estimate the area covered by rivers in each WFD waterbody. A number of known errors were identified, and they were corrected through manual inspection with reference to the 1km mean slope database. The database was simplified to only include river waterbodies – for example, where river waterbodies were connected through a lake waterbody, the river waterbody rather than the lake waterbody was recorded as the downstream waterbody. The main river reach width (m) in each catchment was estimated using a predictive model (McGinnity *et al.*, 2012) based on upstream catchment area at the WFD waterbody outlet (km^2) and Shreve index (Shreve, 1974). Railway lines were buffered by 5m, giving a railway width of 10m for calculation of the area occupied by railways. There were many areas where peat polygons in the Irish Peat Map that overlapped with LPIS parcels (many of which had the land cover description 'Bog', others with land cover descriptions that indicate that transitional land cover classes were present on peat soil areas, as described in Connolly and Holden, 2009). To avoid double-counting these areas, the areas of the peat polygons that intersected LPIS parcels were removed.

Due to varying scales and accuracy, there were some discrepancies between maps, with different maps recording that the same area of land had different types of land cover. If the areas of all land cover types had been summed, in many cases the total would be an area greater than the total area of the catchment. Therefore a hierarchy was used to determine the order in which the land cover areas described by each map were added to the catchment total.

After each step, the total remaining area in each WFD waterbody without an assigned land cover was calculated, and this was used as a maximum for additions from the next land cover data layer until the total area of the catchment was reached.

Table 2-5 Percentage of Ireland occupied by different land covers

Land Cover	Percentage
Arable	5.8
Grass	64.0
Rough	6.4
Wood	4.1
PeatBog	10.5
Other (non-vegetated)	1.2
Other (vegetated)	3.4
Urban	2.0
Rail and road	0.3
Freshwater	2.5

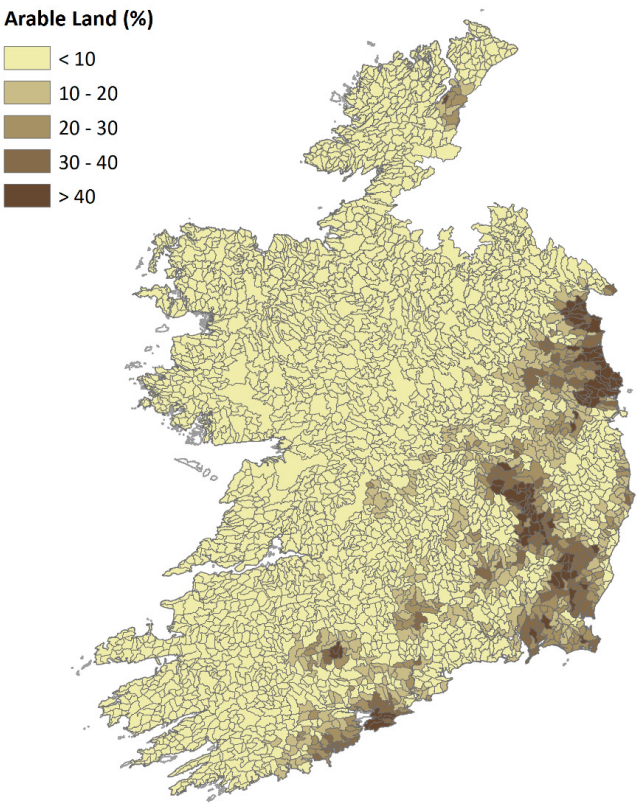


Figure 2-10 Percentage of each WFD waterbody catchment occupied by arable land.

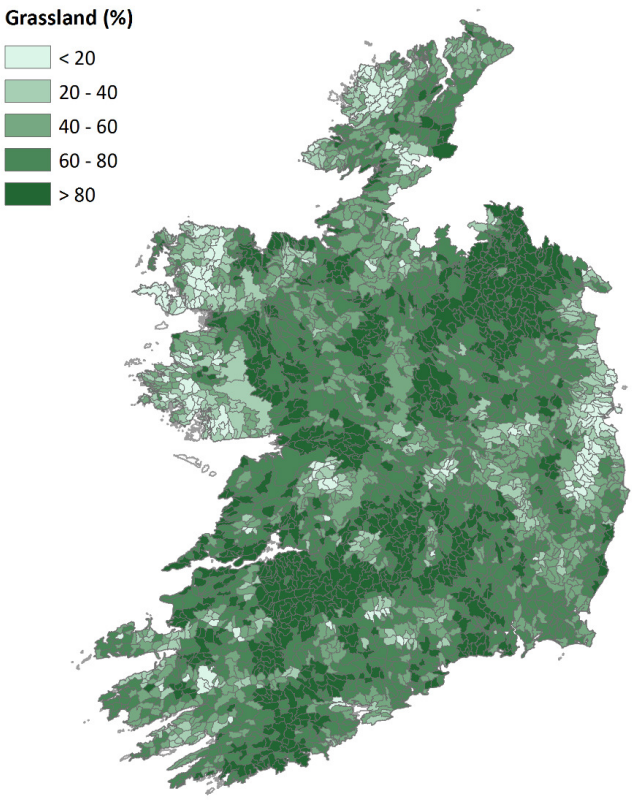


Figure 2-11 Percentage of each WFD waterbody catchment occupied by improved grassland.

2.5.1 Creation of farm type dataset

Using holding level agricultural census data provided by DAFM for 2015, the farm type for each holding was derived using the methodology described in Appendix 2 of the 2010 Irish Census of Agriculture. In this approach, standard output (SO) coefficients, estimated regionally per hectare of crop or per animal, were applied to the individual holding's crop and livestock activities. Farm type was then defined depending upon the dominant source(s) of output for a holding. Note that pig and poultry information was not used in this calculation – the pig data provided could not be fully mapped to the rest of the holding level data and so was therefore kept separate, whilst no poultry information was provided. These omissions are likely to be of minor importance given the relative insignificance of these industries in Irish agriculture.

Holdings were allocated to one of the following farm types, with national crop and livestock numbers by farm type shown in Table 2-6. Due to the small number of 'Other' farm types and limited area of land occupied by them, this farm type was not used in the modelling work (see subsequent section) and the cropping and livestock were allocated to alternative farms (in order to preserve the overall totals).

- Mixed Crops
- Mixed Crops & Livestock
- Mixed Grazing Livestock
- Specialist Beef
- Specialist Dairy

- Specialist Sheep
- Specialist Tillage
- Other

The land within each holdings was allocated to the different WFD waterbodies based upon the fields belonging to that holding as identified from LPIS dataset. Grazing livestock were assumed to be evenly spread across the land belonging to a farm, and livestock were distributed between waterbodies based upon the proportion of each holding within them.

The pig numbers were only available spatially at county scale. They were disaggregated to WFD waterbody scale by assuming the pigs in a county were evenly spread across all managed agricultural land.

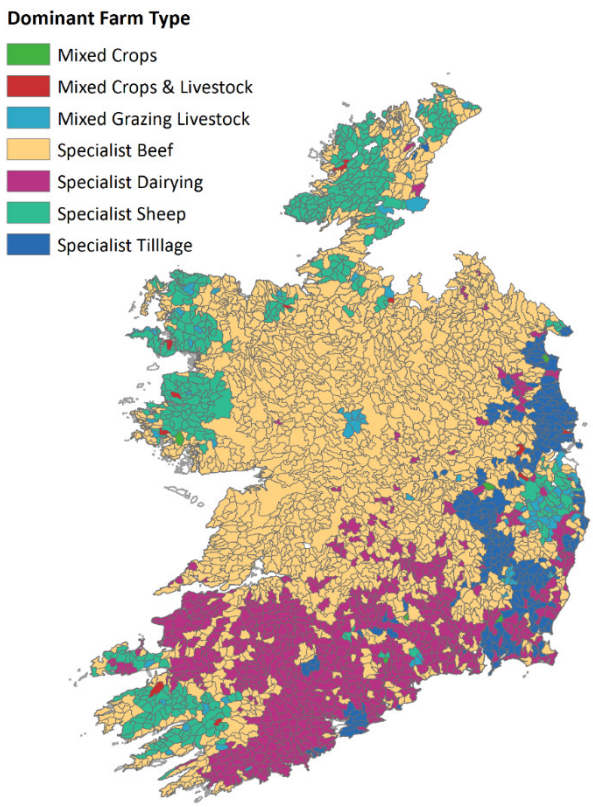


Figure 2-12 Dominant farm types, by farm area, in the WFD waterbodies

Table 2-6 Summary of national crop areas and animal numbers

	Mixed Crops	Mixed Crops & Livestock	Mixed Grazing Livestock	Specialist Beef	Specialist Dairy	Specialist Sheep	Specialist Tillage	Pig	Other	Total
Count of Farms	**11,420**	**2,123**	**11,494**	**69,540**	**16,044**	**13,803**	**4,822**		**694**	**129,940**
Improved Grass	167,298	64,026	407,444	1,824,102	877,553	285,757	50,462		5,225	3,681,866
Rough Grazing	11,947	1,007	26,035	90,938	16,554	60,733	1,458		123	208,794
Arable Land	17,202	51,267	8,158	25,097	26,424	1,629	213,986		6,006	349,769
Dairy cows & heifers	955	9,978	113,082	52,242	1,852,776	243	1,170		931	2,031,377
Other cattle	18,795	126,012	593,430	3,122,537	665,913	30,561	34,923		3,321	4,595,492
Sheep	8,428	58,397	623,802	362,985	36,404	1,272,056	15,127		886	2,378,085
Lambs	17,954	40,236	259,210	245,735	20,909	457,153	20,992		695	1,062,884
Pigs								1,563,039		1,563,039

3 Agricultural Pollutant Modelling

This chapter provides an overview of the evidence and assumptions made in developing the representative farm systems for Ireland (Section 3.1), an introduction to the source apportionment system (Section 3.2) and process based models used to calculate pollutant emissions from each coordinate (Section 3.3), and the linked models used to estimate the impact of water logged and compacted soils (Section 3.4). The chapter concludes with the method for calculating the effects of mitigation methods, and the net effect of multiple mitigation methods targeting the same pollutant source coordinates (Section 3.5).

A summary of all the data sources referenced in the following sections is provided in the table below.

Table 3-1 Main data sources used in the calculation of baseline pollutant losses.

Dataset / Parameter	Source
Agri-environmental Data	
Soil series distribution	Irish Soil Information System association map
	Teagasc-EPA soils and subsoils map of drainage categories
	Irish Peat Map (Connolly and Holden, 2009)
Soil series properties (incl sand, silt, clay)	Irish Soil Information System
Soil series properties (bulk density)	Irish pedo-transfer function (Reidy *et al.,* 2016)
Soil series properties (water capacity)	UK pedo-transfer functions (Defra, 2008)
Soil series properties (HOST class)	European methodology (Schneider, 2007)
Surface runoff connectivity	Irish LPIS field parcels
	European LUCAS data (Toth *et al.,* 2013)
Climate data	Met Éireann (Walsh, 2012)
Slope	NOAA website
Land cover	Irish LPIS field parcels
	Ordnance Survey Ireland National 250k Map
	EPA 1:50,000 river network map
	CORINE Land cover 2012
Livestock numbers & crop areas	DAFM
Farm type	Methodology in Irish Census of Agriculture
Extent of machinery compaction	Welsh data (Anthony *et al*., 2012)
	English and Scottish Data (AIC agronomist survey)
Extent of livestock compaction and poaching	Expert judgement
Extent of seasonal waterlogging	English data (Anthony *et al.,* 2012)
	English and Welsh data (Forbes *et al*., 1980)

Dataset / Parameter	Source
Farm Management Data	
Livestock excreta quantities / properties	Irish data (Brogan *et al*., 2001; Duffy *et al*., 2016; Government of Ireland, 2014)
Duration of livestock housing	Irish GHG Inventory (Duffy *et al*., 2016)
Animal waste management systems	Irish GHG Inventory (Duffy *et al*., 2016)
Livestock excretion in watercourses	Range of international data sources (summarised in Anthony and Morrow, 2011)
Manure storage	NARSES model (Webb and Misselbrook, 2004)
Manure application timing	English and Welsh data from Manures-GIS ADAS, 2008) constrained by Irish Closed Periods and Irish survey data (Hennessy *et al.,* 2011)
Manure application location	Expert judgement informed by Irish survey data (Hennessy *et al.*, 2011)
Hard standing areas	English and Welsh data (Webb *et al.,* 2001)
Management of dirty water	Welsh data (Anthony *et al*., 2012)
Crop yields and residue contents	Irish GHG Inventory (Duffy *et al.,* 2016)
Fertiliser application rates	Teagasc Fertiliser Survey (Dillon *et al.*, 2018)
Fertiliser application timing	British Survey of Fertiliser Practice (2008 – 2010) constrained by Irish Closed Periods. Validated against Teagasc Fertiliser Advice.
Pollutant Loss Calculations	
Drainage, phosphorus and sediment losses from fields • Monthly crop parameters • Monthly tramlined area • Equation parameters	PSYCHIC model (Davison *et al*., 2008)
Nitrate losses from fields • Export coefficients per unit fertiliser, manure, excreta and area • Denitrification modifiers for grassland	NEAP-N model (Lord and Anthony, 1996) NITCAT model (Lord, 1991) MANNER model (Chambers *et al*., 1999) NCycle model (Scholefield *et al*., 1991)
Non-field losses of nitrate and phosphorus • Losses from manure heaps • Losses from tracks and steadings	English data (Nicholson *et al*., 2011) FIO-Farm model (Anthony and Morrow, 2011)
Methane and nitrous oxide losses	IPCC 2006 methodology with Irish specific data on productivity & manure management (Duffy *et al*., 2016). Indirect nitrous oxide losses calculated from nitrate leaching losses.
Modifiers to field losses due to compaction, poaching and waterlogging	Range of international data (described in Section 3.4)

3.1 Representative farm types

3.1.1 Purpose of the representative farm types

Farm system types were defined as it is believed that baseline pollutant emissions and the potential for mitigation varies with system and location. The purpose of the farm type definitions is to provide all the management data required as input to the process based pollutant models, and the data required to weight the results of the pollutant modelling. The use of farm types allows management practices (e.g. manure management of adult beef cattle) to vary from one farm type to another. The modelling framework determines the relative proportions of animal and crop categories within a catchment that are managed according to the different farm types.

The assumed practices on each farm type were documented in farm management workbooks. The data in these workbooks were based upon recent survey data, prioritising data from Ireland where available, with data for the UK used if required. A summary of the key data sources for these farm management workbooks and the assumptions regarding farm management are given below. The most important survey data are those on the cropping areas and livestock numbers, fertiliser rates, fertiliser and manure timing, duration of livestock grazing and whether livestock are housed on slurry or FYM. The majority of the datasets used are based on national stratified surveys and/or are already used in national policy work in Ireland, and so are considered to be the best and most appropriate data available.

3.1.2 Farm management workbooks

Farm management workbooks were created in Microsoft Excel for each farm type, detailing all the activities and practices required to enable the use of the pollutant models. The workbooks contain the following worksheets:

- Livestock
- Manure stores and hard standings
- Cropping
- Fertiliser calendar
- Manure calendar

3.1.3 Crop areas and livestock numbers

This project defined eight farm types (see Section 2.5.1). The statistical average crop areas and livestock numbers for these farms derived from national data were modified using expert judgement to make them more representative of a typical working farm, and so these adjusted farm types are referred to as 'Representative Farm Types' (RFTs).

As an example of the expert modification, 4% of specialist beef holdings in Ireland have dairy cows, but this amounts to less than one cow per farm averaged across all specialist beef holdings and so dairy cows were thus removed from the specialist beef farm. As a guideline for these modifications, livestock and cropping were considered important for any particular

farm type if more than 10% of holdings of that farm type had that activity, and the number of livestock or area of cropping on that farm type accounted for more than 10% of the national total. Note that despite these modifications to the representative farms, the total number of livestock and area of crops was preserved at WFD waterbody scale.

Because the pig information provided could not be linked to the rest of the holding level census information, a separate pig farm type was created. This farm is assumed to have no land, and all manure generated is exported off-farm and spread on neighbouring farms - although for the purposes of reporting pollutant losses in later sections, the losses from the storage and spreading of pig manure are still allocated to the pig farm. The size of the pig farm was based on the average herd size, and so is smaller than a specialist pig farm as it will reflect the smaller herds found alongside other livestock on other farm types.

Table 3-2 and Table 3-3 give the results of these livestock number and crop area adjustments for each representative farm type.

Table 3-2 Crop areas (ha) on the representative farm types.

	Mixed Crops	Mixed Crops & Livestock	Mixed Grazing Livestock	Specialist Beef	Specialist Dairy	Specialist Sheep	Specialist Tillage	Pig
Permanent Pasture	14.0	26.0	34.5	26.0	53.0	20.5	8.5	-
Rotational Grass	-	4.0	-	-	1.5	2.0	-	-
Rough grazing	1.0	0.5	2.5	1.5	1.0	4.5	-	-
Winter Wheat	-	2.5	-	-	-	-	11.0	-
Winter Barley	-	4.0	-	-	-	-	10.0	-
Spring Barley	0.5	12.0	0.5	-	-	-	16.0	-
OSR	-	-	-	-	-	-	2.0	-
Maize	-	-	-	-	0.5	-	1.0	-
Potatoes	-	0.5	-	-	-	-	1.5	-
Beans	-	0.5	-	-	-	-	1.5	-
Fodder Crops	-	0.5	-	-	-	-	1.0	-
Vegetables	-	1.5	-	-	-	-	-	-
Oats	-	1.5	-	-	-	-	3.0	-
Fallow / Set Aside	-	-	-	-	-	-	0.5	-
Total	16	54	16	28	56	27	56	-

Table 3-3 Livestock numbers on the representative farm types.

	Mixed Crops	Mixed Crops & Livestock	Mixed Grazing Livestock	Specialist Beef	Specialist Dairy	Specialist Sheep	Specialist Tillage	Pig
Dairy Cows and Heifers	0	0	5	0	70	0	0	0
Dairy Heifers in Calf (>= 2 Years)	0	0	1	0	40	0	0	0
Dairy Heifers in Calf (< 2 Years)	0	0	3	0	10	0	0	0
Bulls	0	0	0	0	0	0	0	0
Beef Cows and Heifers	0	10	12	11	3	0	0	0
Beef Heifers in Calf (>= 2 Years)	0	4	4	3	1	0	0	0
Beef Heifers in Calf (< 2 Years)	0	14	14	12	12	0	0	0
Other Cattle (>= 2 Years)	0	7	4	5	2	0	0	0
Other Cattle (< 2 Years)	0	14	8	7	10	0	0	0
Other Cattle (< 1 Year incl Calves)	0	11	11	8	16	0	0	0
Sheep	0	27	54	6	0	93	0	0
Lambs (< 1 Year)	0	19	23	4	0	35	0	0
Breeding Pigs	0	0	0	0	0	0	0	132
Fatteners (> 20 kg)	0	0	0	0	0	0	0	795
Fatteners (< 20 kg)	0	0	0	0	0	0	0	325

3.1.4 Livestock management

The quantity of excreta produced by livestock was taken from COGAP rules (Government of Ireland, 2014), the phosphorous content from Brogan *et al.,* (2001) and the nitrate content from the National GHG Inventory (Duffy *et al.,* 2016). Annual values by livestock type are reported in Table 3-4.

Table 3-4 Quantity and properties of livestock excreta (Brogan *et al*., 2001; Duffy *et al*., 2016; Government of Ireland, 2014)

Livestock	Daily Undiluted Excreta (L)	Annual N Excretion (kg)	Annual P Excretion (kg)
Dairy Cows and Heifers	45	100.6	13
Dairy Heifers in Calf (>= 2 Years)	36	63.4	10
Dairy Heifers in Calf (< 2 Years)	36	63.4	8
Bulls	40	73.8	10
Beef Cows and Heifers	40	73.8	10
Beef Heifers in Calf (>= 2 Years)	36	74.4	10
Beef Heifers in Calf (< 2 Years)	36	74.4	8
Other Cattle (>= 2 Years)	18	37.2†	5
Other Cattle (< 2 Years)	36	63.4	8
Other Cattle (< 1 Year incl Calves)	20	27.6	3
Sheep	4	6.5	2
Lambs (< 1 Year)	0.7	0.6	0.2‡
Sows in Pig and Other Sows	10.9	20.0	8
Gilts in Pig and Barren Sows	10.9	20.0	8‡
Gilts Not Yet in Pig	5.6	9.2	3.7‡
Boars	7.8	16.0	6.4‡
Other Pigs (> 20kg)	5.1	9.2	1.7
Other Pigs (< 20kg)	1.3	3.0	10.6‡

† Accounts for proportion of the year these livestock are on farm
‡ derived from other stock categories based upon excreta volume

The location of cattle and sheep throughout the year determines the amount of excreta deposited in fields and the amount and type of manure to be handled on the steading. Sheep are assumed to be grazed for the whole year, except for a 6-week period during winter. All sheep were assumed to spend the winter period on land attached to the farm, and not to be wintered on another farm. For cattle, the dates of animals being turned out and brought in from grazing were taken the National GHG Inventory (Duffy *et al.,* 2016) (Table 3-5), with most adult animals spending seven to eight months out grazing. O'Mara

(2006) shows that the length of the grazing period for cattle is typically a few weeks less than this in the North and a few weeks longer in the East.

Table 3-5 Number of days per year that beef and dairy cattle are at grazing or housed (Duffy *et al*., 2016).

	Days Housed	Days Grazing
Dairy Cows	117	248
Suckler Cows	141	224
Dairy Heifer	128	237
Other Heifer	139	226
Under1yr	223	142
Oneto2yrs	157	208
Over2yrs	20	345
Bulls	157	208

Table 3-6 Housing dates for different cattle types, by calving date and by region[1] (O'Mara, 2006).

			Spring-calving cows			Autumn-calving cows
			First third of cows to calve	Second third of cows to calve	Last third of cows to calve	
Dairy	Region 1	Turnout date by day + night	8 March	16 March	9 April	8 March
		Housing date	29 Nov	29 Nov	29 Nov	29 Nov
		Days	**266**	**258**	**234**	**266**
	Region 2	Turnout date by day + night	15 March	21 March	14 April	15 March
		Housing date	22 Nov	22 Nov	22 Nov	22 Nov
		Days	**252**	**246**	**222**	**252**
	Region 3	Turnout date by day + night	29 March	29 March	18 April	29 March
		Housing date	8 Nov	8 Nov	8 Nov	8 Nov
		Days	**224**	**224**	**204**	**224**
Suckler	Region 1	Turnout date by day + night	1 April	1 April	1 April	1 April
		Housing date	15 Nov	15 Nov	15 Nov	15 Nov
		Days	**228**	**228**	**228**	**228**
	Region 2	Turnout date by day + night	5 April	5 April	5 April	5 April
		Housing date	6 Nov	6 Nov	6 Nov	6 Nov
		Days	**215**	**215**	**215**	**215**
	Region 3	Turnout date by day + night	13 April	13 April	13 April	13 April
		Housing date	31 Oct	31 Oct	31 Oct	31 Oct
		Days	**201**	**201**	**201**	**201**
			Suckler	**Dairy cross**		
	Beef	Turnout date by day + night	15 April	15 May		
		Housing date	12 Nov	12 Nov		
		Days	**211**	**181**		

The proportions of excreta from grazing livestock deposited on yards or in housing that were then managed as slurry or as farmyard manure (FYM) were taken from the National GHG Inventory (Duffy *et al.,* 2016), with the vast majority of livestock on pit systems and thus producing slurry. All pigs were assumed to be on slatted floors, thus producing slurry, based on the assumption in Ireland's National Inventory Report 2016 (Duffy *et al.,* 2016). The

[1] Regions are: 1 South and East; 2 West and Midlands, 3 North West.

majority of slurry produced was assumed to be stored in tanks rather than lagoons, based upon data from Hennessy *et al.,* (2011) (Table 3-8).

Table 3-7 Allocation of animal wastes to animal waste management systems (Duffy *et al.*, 2016).

	Pit	Bedding	Pasture
Dairy Cows	29	2	69
Suckler Cows	27	10	63
Dairy Heifer	35	0	65
Other Heifer	38	0	62
Under1yr	41	20	39
Oneto2yrs	34	9	57
Over2yrs	3	2	94
Bulls	38	13	49

Table 3-8 Percentage of farms with soiled water storage by facilities (Hennessy *et al.*, 2011).

	Dairy	Cattle	Sheep	Tillage	All
Soiled Water Tank	56	20	27	55	33
Slurry Tank	35	68	67	33	57
Silage Effluent Tank	4	8	5	12	6
Lined Lagoon	3	2	0	0	2
Unlined Lagoon	0	1	1	0	1
Reedbed	1	0	0	0	0
Other	1	1	0	0	1

In general, observations of grazing cattle have reported that the quantity of excreta deposited is in direct proportion to the amount of time spent in the riparian area or watercourse (see, for example, Bagshaw, 2002). On this basis, the quantity of excreta deposited directly in a watercourse during livestock movement to and from the milking parlour or between pastures, is expected to be small. Experiments on three breeds of cattle in France testing the effect of distance and walking speed on milk yields, reported that walking speeds (inclusive of halts and solicitations) ranged from 3 to 6 km hr^{-1} over distances of 3.2 to 5.6 km (d'Hour *et al.,* 1994). Allowing for a general slowing down and halts for drinking, the time taken to cross a 5 to 20 m watercourse would be less than 10 minutes. Therefore, it would be expected that a maximum of 1% of the daily excreta output would be directly deposited in a watercourse crossed twice daily by dairy cattle (20 minutes out of 1,440 minutes in a day). However, there has been some suggestion that the frequency of defecation or urination is greater when cattle enter a watercourse. Davies-Colley *et al.,* (2004) monitored the impact of a herd of 246 dairy cows crossing a stream ford in New Zealand. A total of 25 defecation events were recorded when 170 cows were videoed

crossing the 17 m ford, and 11 events following the passing of all 246 dairy cows along the 200 m approach. Defecation counts on the raceway and in the ford indicated that the cows defecated 50 times more per unit length of their path through the stream than elsewhere on the raceway, but they were also travelling 10 times slower, indicating a 5 fold increase in the defecation rate. Overall, the statistics indicated that 10% of cows defecated when crossing the ford once. In contrast, (Demal, 1982) reported on the monitoring of livestock activities at or near a stream at five cattle access sites on the river Avon, Ontario in 1982. The sites were monitored for two dry-weather days during the period from July to September when cattle were in pasture. A total of 10 access events were monitored, lasting from 1 to 45 minutes. Measurements taken during the events included the number of cattle crossing the river channel (less than 25 m wide), the number of cattle watering at the channel edge, and the number of in-stream defecations and urination. On average during each channel access event, 76% of the animals present entered the stream, and of those 12% urinated and 18% defecated. The average duration of each access event was 14 minutes. There was no evidence of an increased rate of defecation during crossing, as the measured rate was less than the expected daily average rate. Based on these data, it was simply assumed that an average of 10% of cattle crossing a watercourse defecate, regardless of the crossing time. If each animal defecates an average of 12 times a day (North Wyke Research, 1999), then 2% of the daily excreta produced during the grazing season would be direct to the stream on a farm where stream crossing was necessary. Beef cattle are assumed to be moved between fields every few days, which may also require them to ford a stream – although this would only be a small fraction (< 1%), the consequences of this for pollution could still be significant due to the direct deposition of the excreta. Sheep do not like to spend time in water, so the effects of them crossing streams between fields would be minimal.

The periods of time spent by grazing animals on farm tracks when moving between fields were based on the survey data collated by Anthony and Morrow (2011). Dairy cattle were estimated to spend between 20 and 80 minutes per day on the farm tracks when they were required to travel from the milking parlour to the grazing area, equating to around 2% of the whole year.

Anthony and Morrow (2011) also summarised the available data on the time cattle spend in streams and on channel banks for drinking, shade and access to palatable vegetation. Based on a number of reports, they assumed that grazing cattle will spend between 1 and 5% of the grazing day in a watercourse, providing that it is not fenced off. In this project it was assumed that cattle spent 30 minutes in or directly adjacent to the water per day they were grazing (2% of the time grazing), which equates to roughly 1% of the year.

Dairy cattle were assumed to spend 3 hours per day on feeding, loafing and collecting yards when waiting to be milked.

3.1.5 Manure management

The proportions of managed manure that were stored, rather than spread immediately, were taken from the NARSES modelling system (Webb and Misselbrook, 2004). For cattle manure, NARSES assumes 69% of FYM is stored, whilst the figure is 70% for pig FYM. Field heaps were assumed to be uncovered and on a permeable base. Solid manure stored on the steading was assumed to be on an impermeable base, and stored in such a way that runoff, or the impacts of runoff, were negligible. The increase in solid manure volume due to straw bedding was calculated to be 30% for dairy animals on a solid manure system and 15% for other cattle and sheep (Defra, 2006). Storage of farmyard manure was also assumed to

result in a reduction of manure volume of 30% due to the effects of decomposition (MAFF project WA0519, 1997). Ammonia emissions and nitrogen mineralisation during storage of slurry and farmyard manure are calculated according the NARSES model (Webb and Misselbrook, 2004) to estimate the total quantity and plant available nitrogen on spreading to land.

Manure applications to land are prohibited during 'closed periods' (Table 3-9; Government of Ireland, 2014), which extend from October/November until January. In a survey of farm management, Hennessy *et al.,* (2011) found the proportions of manure applied during different windows outside of these closed periods (Table 3-10). These two datasets were used to constrain and redistribute existing monthly distributions of manure application timings available for England and Wales, taken from the Manures-GIS modelling system (Defra Project WQ0103), which specifies timings for different manure types to grassland, winter sown arable crops and spring sown arable crops. This system is an integration of Defra funded surveys of manure management, by farm type, across England and Wales (see, for example, Smith *et al*., 2000; 2001a, 2001b). Figure 3-1 shows the manure timings produced for slurry and FYM applied to grassland and arable land.

The choice of fields receiving the manures was partly based on expert opinion, considering crop rotations and the need for clean forage production, and data available from Hennessy *et al.,* (2011). Given the dominance of grassland over arable agriculture, the vast majority of manure is applied to grassland. Manure was typically spread at a rate of 30 t ha^{-1}, but always respecting a field nitrogen load limit of 250 kg N ha^{-1}.

The manure management on the representative farm types was designed to represent a baseline for typical agricultural practice, which could be improved upon by the implementation of mitigation actions. As such, all manure was applied using a broadcast spreader. On grassland, manure applications were not incorporated, and on arable land, they were incorporated within 5 days. Default manufactured fertiliser rates were not adjusted to take account of any nutrients in livestock manures available on the representative farm types.

Table 3-9 Periods when application of fertilisers to land is prohibited by region[2] (Government of Ireland, 2014).

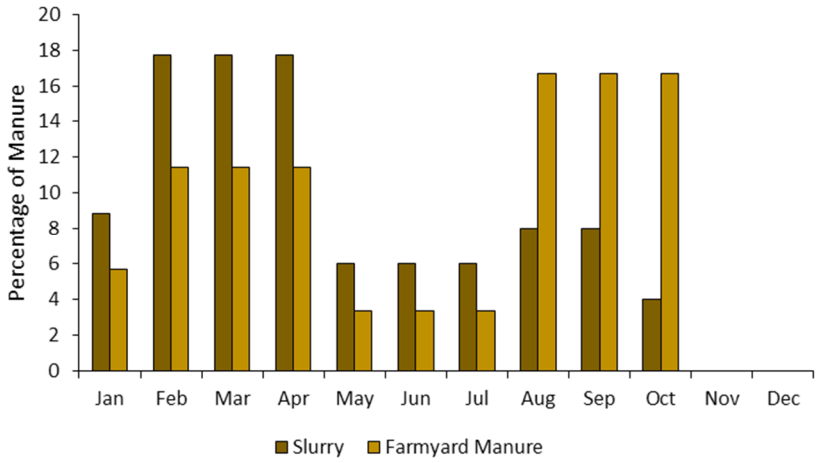
		Chemical fertiliser	Organic fertiliser (other than farmyard manure)	Farmyard manure
Region 1	Start	15 September	15 October	1 November
	Finish	12 January	12 January	12 January
Region 2	Start	15 September	15 October	1 November
	Finish	15 January	15 January	15 January
Region 3	Start	15 September	15 October	1 November
	Finish	31 January	31 January	31 January

[2] Regions are: 1 South and East; 2 West and Midlands, 3 North West.

Table 3-10 Estimated percentage of total slurry and farmyard manure applied in various periods by farm type (Hennessy *et al.*, 2011)

		Close to April 30th	May 1st to July 31st	August 1st to Close
Slurry	Dairy	52	35	13
	Cattle	52	39	9
	Sheep	48	36	16
	Tillage	62	18	20
	All Farms	52	36	12
Farmyard manure	Dairy	27	14	59
	Cattle	42	15	43
	Sheep	30	23	47
	Tillage	40	10	50
	All Farms	35	15	50

Arable



Grass

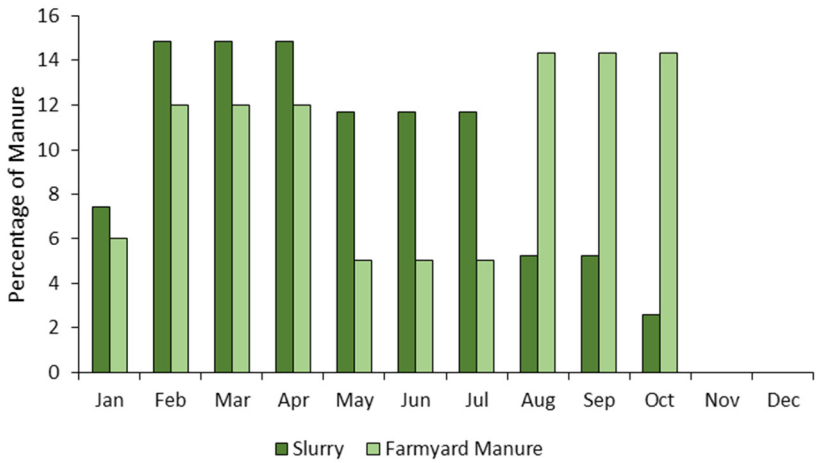


Figure 3-1 Manure application timings to different crop types, based on data from Manures-GIS (ADAS, 2008) constrained by closed periods for manure application in Ireland and surveyed data available in Hennessy *et al*., (2011).

Table 3-11 Estimated percentage share of total quantities of slurry applied to various crops by farm type (Hennessy *et al*., 2011)

	Grazing	Hay/Silage	Maize	Tillage
Dairy	37	60	2	1
Cattle	36	63	0	1
Sheep	34	66	0	0
Tillage	42	33	6	19
All Farms	37	60	2	1

3.1.6 Hard standing management

Estimates were made of the volume of dirty water generated on the hard-standings, based on the yard areas and volumes of rainfall and wash water. Hard standing areas for cattle and sheep were taken from Webb *et al*., (2001), where they are expressed on a per head basis by livestock type. On farms with housed livestock, the loading area for removal of finishing pigs was based on a required area of 1 m^2 per pig greater than 80 kg. It was assumed that 60% of hard standings were covered (Defra, 2006), with any rainfall falling on the uncovered yards collected and sent to the slurry store or a dirty water tank or potentially draining to a watercourse. The milking parlour was assumed to be washed out every day, requiring management of an additional 25 litres of water per cow milked (Laws and Chadwick, 2005). The destinations of the rainwater and parlour washings were taken from a survey for Wales (Anthony *et al*., 2012), which was stratified by farm type (Table 3-12). All water falling on covered yards or buildings was assumed to be clean and go straight to drains as buildings were adequately guttered – although this is an unrealistic assumption, the additional slurry / dirty water volume that would be generated from imperfect guttering would cause little change to the nutrient quantity in the slurry / dirty water. The quantity of excreta deposited on the yards was calculated from a calendar of livestock activity, based on the survey data synthesised by Anthony and Morrow (2011), and the proportion lost in yard runoff was inversely related to the frequency of yard cleaning.

Table 3-12 Modelled proportions (%) of dirty water from farm hard standings sent to different destinations, by farm type

Destination	Dairy	Other
Dirty Water Store	35.0	20.0
Slurry Store	55.0	20.0
Other*	10.0	60.0

*Including discharge to fields, ditches or watercourses.

3.1.7 Crop management

Aside from fertiliser and manure applications to crops, which are dealt with the in the surrounding sections, the only other aspects of crop management that are required for the pollutant modelling are estimates of yields and residue nitrogen contents, which were taken from Ireland's National Inventory Report 2016 (Duffy *et al.,* 2016).

3.1.8 Fertiliser management

Nitrogen and phosphorus fertiliser rates for major arable crops and grassland were provided by Teagasc, based upon a draft version of a forthcoming publication on fertiliser use in Ireland between 2005 and 2015 (Dillon *et al.*, 2018). Where possible, rates were provided by farm type (Table 3-13).

Table 3-13 Average fertiliser application rates in Ireland between 2005 and 2015 for different crop types, by farm type (Dillon *et al.*, 2018). A dash marks where data was unavailable for a specific farm type, and the overall average rate was used instead.

	N					P_2O_5				
	Cattle	Dairy	Sheep	Tillage	All	Cattle	Dairy	Sheep	Tillage	All
Grass	56	153	41	68	83	16	25	16	18	18
Winter wheat	-	-	-	197	199	-	-	-	55	57
Spring wheat	-	-	-	-	114	-	-	-	-	44
Winter barley	-	-	-	176	18-	-	-	-	60	60
Spring barley	130	117	-	136	134	69	50	80	55	57
Winter oats	-	-	-	146	146	-	-	-	48	48
All cereal crops	133	121	124	-	158	69	50	80	-	57
Fodder crops	-	158	-	138	139	-	135	-	121	121
Root crops	111	116	-	115	108	76	80	-	101	89
Maize	-	127	-	-	120	-	96	-	-	85

Timing of fertiliser applications was based on an analysis of data from the British Survey of Fertiliser Practice from 2008 to 2010. This timing data was constrained by the closed periods for fertiliser applications in Ireland (Table 3-9). For grassland, it was possible to compare this timing distribution for nitrogen applications with fertiliser advice provided by Teagasc for livestock farms, which shows a good agreement between the data sources (Figure 3-2). The nitrogen and phosphorus timing distributions used for grassland and a selection of arable crops are shown in Figure 3-3.

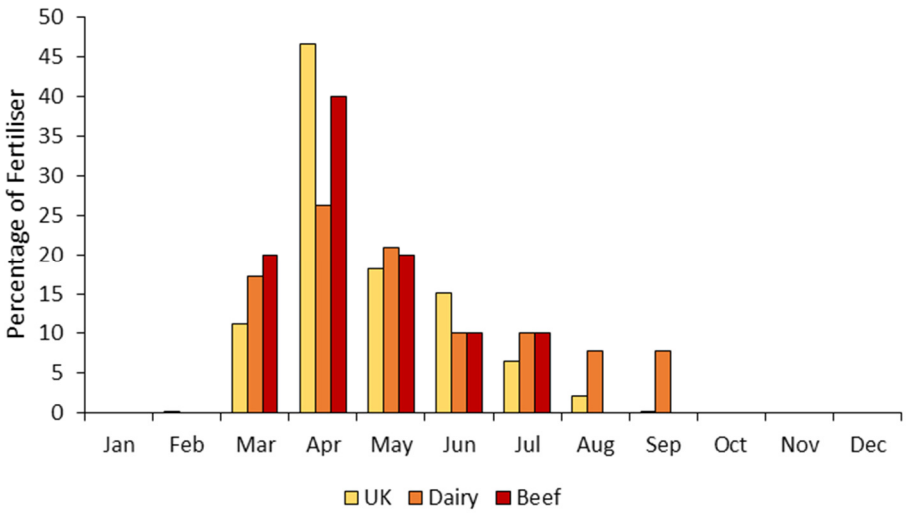


Figure 3-2 Nitrogen fertiliser timing data for grassland based on Teagasc advice for Dairy and Beef farms stocked at the national average stocking rate, and for the UK from the British Survey of Fertiliser Practice.

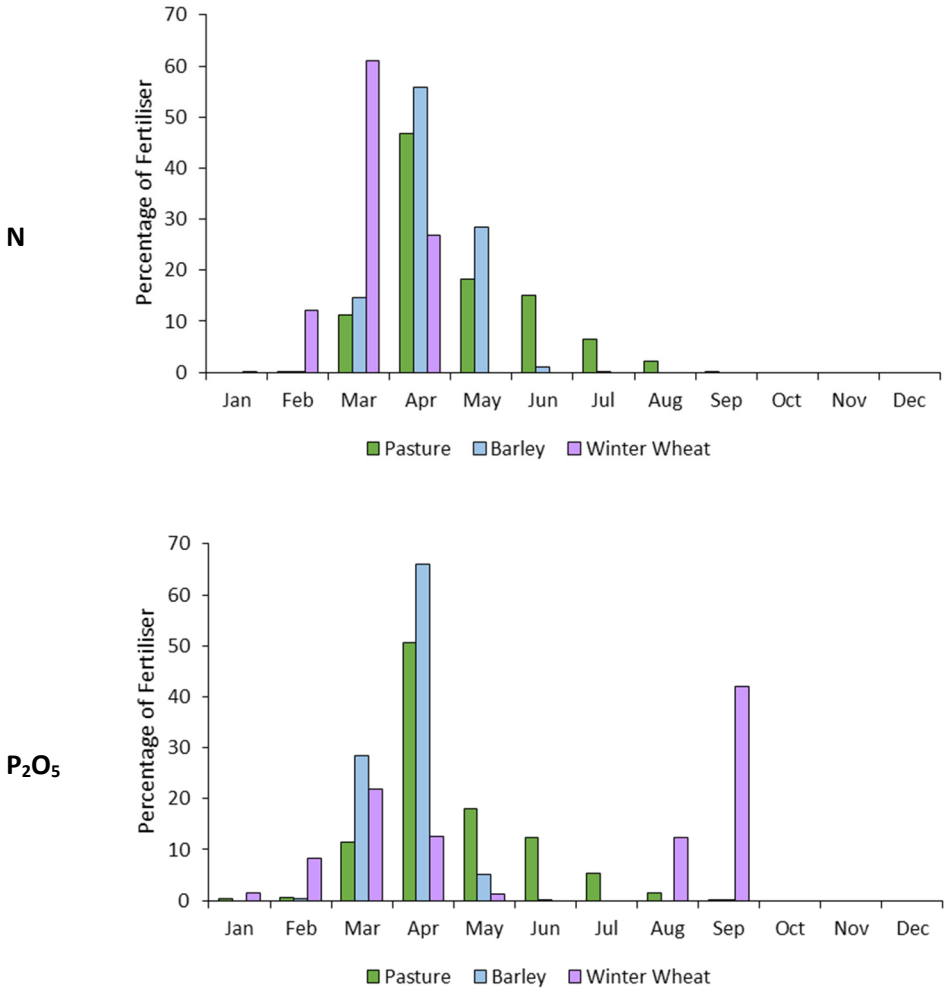


Figure 3-3 Fertiliser timings for common crop types based on British Survey of Fertiliser Practice data for 2008 to 2010, constrained by fertiliser closed periods in Ireland.

3.2 Source apportionment system

The process based diffuse pollution models selected for this project were each capable of disaggregating total emissions into losses from specific source types (such as cattle manure and fertilisers), source areas (such as arable and grassland) and delivery pathways (such as surface runoff and drain flow) on the representative farm types. All emissions could therefore be explicitly referenced by a source apportionment coordinate system (Table 3-14). This was done to aid querying and explanation of model results, and to ensure that the effects of mitigation actions were applied only to the pollutant source that they affected.

Table 3-14 Components of the source apportionment system used by the modelling framework in this study.

Farm	Pollutant	Source	Area	Pathway	Type	Form	Timescale	Condition
Cereal	Sediment	Dairy	Arable	Runoff	Soil and crop	Particulate	Short	Default
General	Nitrate	Beef	Grass	Preferential	Fertiliser	Dissolved	Medium	Machinery Compaction
Horticultural	Phosphorus	Sheep	Rough	Leaching	Farm yard manure	Gas	Long	Livestock Trampling
Pig	Nitrous oxide	Pig	Yards	Air	Slurry	Indirect Gas		Seasonal Poaching
Poultry	Methane	Poultry	Housing	Direct	Litter			Feeder Poaching
Dairy	FIOs	Chemical	Tracks		Voided			Trough Poaching
CS-LFA		Soil	Fords		Enteric			Waterlogged
CS-Low			Field storage		Dirty water			
Mixed			Steading storage					

3.3 Calculation of pollutant emissions

Baseline pollutant losses from the representative model farm types were calculated using a range of computer models used in policy support at farm and national scale. The only requirement was that the losses could be explicitly disaggregated between source types (fertiliser, excreta, soil and manure), source areas (arable, grassland, rough grazing and steading), and delivery pathways (surface runoff, leaching and preferential flow) according to the source apportionment coordinate system (see Section 3.2). Where necessary, modifications were made to the models to represent the effects of soil compaction and poaching, based on observed levels of soil damage (see Section 3.4). The models were applied to the detailed field and farm scale descriptions of activities for each representative farm type, at all locations across Ireland. The emissions were then re-expressed as a proportion of the farm scale potential pollutant inputs, such as the nutrient load in fertiliser and excreta, to derive an export coefficient emissions model specific to each WFD waterbody that was sensitive to the local soil and climate conditions.

3.3.1 Methane and nitrous oxide

The primary sources of nitrous oxide emissions are the combined nitrification and denitrification of inorganic soil nitrogen (influenced by applications of mineral fertiliser and organic manure, and excreta deposited in the field), and of nitrate leached from agricultural land. Nitrification is the aerobic microbial oxidation of ammonium to nitrate, and denitrification is the anaerobic microbial reduction of nitrate towards nitrogen gas. Nitrous oxide (one of a range of oxides of nitrogen) is an intermediate product in the denitrification process and a by-product of nitrification. The primary agricultural source of methane is as a by-product of enteric fermentation, by both ruminant and non-ruminant animals. Methane is also produced by the anaerobic decomposition of animal manures, and organic soils can be a source for methane due to anaerobic fermentation under saturated conditions. The rate of methane produced from enteric fermentation is dependent on the level of feed intake, quantity of energy consumed and feed composition and the emissions from animal manures are dependent upon temperature and the biodegradability of the manure (Monteny *et al.,* 2006). Emissions from manures are greatest when associated with slurry storage and least when excreta is deposited directly to pasture. Hence, the system of manure management affects emissions.

Methane and nitrous oxide emissions were calculated according to the methodology of the Intergovernmental Panel on Climate Change (IPCC, 2006) wherein data on livestock numbers, crop areas, and the nitrogen contents of fertiliser and manure are multiplied by agreed emission factors, using country specific data on productivity and manure management (Duffy *et al*., 2016). Where appropriate, some of the country specific data were replaced by the management data for the representative farm types. Several changes to the default IPCC methodology were made in the calculation of nitrous oxide emissions:

- The IPCC methodology assumes 20% of N excreted is volatilised. The farm workbooks explicitly calculate the volatilisation of excreta deposited in housing and on yards and volatilisation during the storage of manures, accounting for the time livestock spend in different locations and the amount of manure stored. Additional calculation of the volatilisation of excreta at grazing and during manure spreading allowed the IPCC assumption to be ignored and a more realistic mass flow approach to be adopted for both the direct nitrous oxide emissions and indirect emissions

following ammonia deposition, following the NARSES model (Webb and Misselbrook, 2004).

- The IPCC methodology calculates indirect losses of nitrous oxide from the denitrification of leached nitrate, but assumes the amount of leached nitrate is a fixed fraction of the applied organic and inorganic N. This assumption has been replaced by using the results of the nitrate leaching model (Section 3.3.3).
- The IPCC methodology ignores the impacts of soil compaction and poaching on nitrous oxide emission rates. Nitrous oxide emission rates are sensitive to soil aeration with the largest emissions occurring at or around field capacity (Davidson, 1991). The methodology was refined by introducing adjustments to represent the impact of soil damage on nitrous oxide emissions using empirical data as evidence (see Section 3.4).

3.3.2 Phosphorus and sediment

The diffuse sediment and phosphorus emissions from the representative farm types were calculated using the field scale version of the PSYCHIC model (Davison *et al.*, 2008; Stromqvist *et al*., 2008; Collins *et al*., 2007). This is a process based, monthly time-stepping, model with explicit representation of surface and drain flow hydrological pathways, particulate and solute mobilisation, and incidental losses associated with fertiliser and manure spreading. The model has previously been integrated with the soils, climate and agricultural census data held in the MAGPIE decision support system (Lord and Anthony, 2000) to calculate total phosphorus losses from all agricultural land, including rough grazing and runoff from hard-standings. The model calculations took account of landscape retention (see Section 2.4), and were the best available estimate of net delivery to lakes and rivers. The model output has been used previously to support phosphorus and sediment gap analyses for rivers and lakes in England and Wales (Anthony *et al.,* 2008; Anthony and Lyons, 2006; Anthony and Collins, 2007). Its application therefore ensured some consistency across a number of projects used to support government policy development.

The PSYCHIC model estimate soil phosphorus losses based upon Olsen P, with the Olsen's P value for a field calculated from soil texture and land use based upon unpublished data. The Irish National Soils Database (Fay *et al*., 2007) report average Morgan's P for grassland, tillage and rough grazing on mineral soils, Jordan *et al.,* (2012) has data from approximately 1,900 fields in 4 small catchments and the GLAS Nutrient Management Plans also provide P index values. The was no clear agreement between these datasets to justify replacing the default rules within PSYCHIC. The contribution of dissolved soil P to the national loss is approximately 13%, so any modification of the rules would not change overall loads significantly.

Table 3-15 Estimation of soil Olsen P (mg P kg^{-1}) from soil texture and land use

	Combinable Crops	Potatoes & Vegetables	Intensive grass	Extensive Grass
Sandy	42	45	25	7
Light	32	41	26	7
Medium & Heavy	27	30	22	7

PSYCHIC does not calculate agricultural losses from livestock manures at non-field areas, such as runoff farm hard standings, leaching from field manure heaps, runoff from farm tracks, or direct deposition into unfenced watercourses. Modifications were therefore made to estimate losses based on the proportion of time that livestock spent on each area or the volumes of excreta or manure handled, according to the data described in Section 3.1. Nicholson *et al.,* (2011) found that losses of phosphorus in leachate from manure heaps vary from 0.03 to 12.5% of the total P into the store. Based on Nicholson *et al.,* (2011) and other data cited within, a value of 2% was chosen for all situations. For manure heaps on yards, only a proportion of this potential would not be retained by bunding or similar. Slurry tanks and lagoons were assumed to be watertight and not lose any nutrients. Losses from excreta deposited on tracks and steadings were based on the FIO-Farm model (Anthony and Morrow, 2011).

3.3.3 Nitrate

Nitrate losses from the representative farm types were calculated using a combination of the field scale N-CYCLE, NITCAT and MANNER models (Lord, 1992; Scholefield *et al.,* 1991; Chambers *et al.,* 1999). The EDEN model (Gooday *et al.,* 2008) was also used to assess the proportion of nitrate losses by different pathways. To ensure a common hydrological basis between the water based pollutants, the combined nitrate models were linked to the PSYCHIC model (see above), so that it could use the output of that models water balance calculations. The selected nitrate models were sensitive to cropping history, fertiliser and manure nitrogen inputs and crop off-take, stocking density, and soil hydrology, and have previously been used to support the evaluation of Defra nitrates policy and the designation of the Nitrate Vulnerable Zones (Lord and Anthony, 2000).

The nitrate models used did not calculate agricultural losses from livestock manures at non-field areas, therefore separate calculations were performed, using the same approach as for phosphorus (Section 3.3.2).

3.4 Effect of soil compaction, poaching and waterlogging

Soil compaction, poaching and waterlogging can significantly increase diffuse pollutant emissions by increasing the risk of surface runoff and altering the aeration status of the soil. Some of the mitigation actions are designed to target these issues, so it was necessary that the Framework Model export coefficients were modified to explicitly represent their effects and thus potential mitigation.

The issues represented in the modelling were: machinery compaction; livestock trampling; seasonal poaching; poaching around livestock feeders; poaching around livestock water troughs and seasonal waterlogging. A specific component, 'Condition', was added to the source apportionment system for the representation of these issues. The initial results of the modelling were the 'default' condition. To represent e.g. livestock trampling, all of the relevant rules for pollutant losses occurring on grassland would be duplicated, with the duplicate coefficients being assigned the 'livestock trampling' condition. The value of the coefficients for the 'livestock trampling' component is altered from the 'default' value to reflect the cumulative effect of the area affected by livestock trampling and the increase in pollutant losses. The areas impacted by the different issues are listed in Table 3-16, Table 3-17 and Table 3-19 and the multipliers for each of these condition components relative to the 'default' condition component are in Table 3-18.

3.4.1 Machinery compaction

Soil compaction is associated with machinery wheelings and can affect a large fraction of a field area, especially on grassland where compaction is not removed by regular tillage.

Surveying farms in Wales, Anthony *et al.,* (2012) found that there was a significant increase in the frequency of reporting of compaction due to machinery on the dairy farms (25%) compared with upland cattle and sheep farms (10%) and lowland cattle and sheep farms (15%). An AIC Agronomist survey of soil quality on 146 farms (covering 56,000 ha) in England and Scotland reported that 10-15% of the cultivated land area was compacted. The farms surveyed were large arable farms, so this is taken as being representative of the intensive farm types. Therefore, it was assumed that 15% of fields on intensive farm types were compacted, with lower values on other farm types (Table 3-16). Where compaction was a problem, it was assumed to occupy 10% of the field area on both arable and grass fields (Table 3-17).

Table 3-16 Percentage of fields affected by soil compaction and poaching on the different farm types

Farm Type	Machinery Compaction	Livestock Trampling	Seasonal Poaching	Feeder Poaching	Trough Poaching
Mixed Crops	10	0	0	0	0
Mixed Crops & Livestock	15	20	20	100	100
Mixed Grazing Livestock	10	20	20	100	100
Specialist Beef	10	20	20	100	100
Specialist Dairy	15	20	20	40	100
Specialist Sheep	5	20	20	100	100
Specialist Tillage	15	0	0	0	0
Pigs	-	-	-	-	-

Table 3-17 Percentage area within affected fields which is affected by soil compaction and poaching on the different farm types

Land Use	Machinery Compaction	Livestock Trampling	Seasonal Poaching	Feeder Poaching	Trough Poaching
Arable	10	-	-	-	-
Grass	10	20	3	2	2
Rough	-	-	-	-	-

Emissions to water

Modifications to the HOST Standard Percentage Runoff (SPR) coefficient for compacted soils have been previously used to estimate an increase in rapid runoff of c. 30 to 35% for fields affected by compaction at landscape scale (Anthony *et al.,* 2012). Silgram *et al.,* (2007) reported increased runoff from compacted and repeatedly wheeled tramline plots of up to 10 times greater than without tramlines. Li *et al.,* (2007) measured runoff from controlled traffic plots of 90 m^2 (representing improved practice) that was 36% smaller than from single wheeled plots (representing conventional practice) for a heavy clay in Queensland, Australia. Robinson and Naghizadeh (1992) measured runoff from wheeled areas of calcareous silt loam plots on the South Downs, England, which was 1.3 to 6.6 times greater than from unwheeled areas. Assuming a threefold increase in runoff from the compacted area of a field, it was estimated that a 10 to 15% compacted area would generate the increase in runoff implied by the modified HOST model (hence Table 3-17). This is a small part of the total trafficked area (50 to 95%) under conventional or reduced tillage (see, for example, Kroulik *et al.,* 2009).

In a review of the effect of vehicle compaction on soil properties, Chamen (2006) reported that without wheel compaction, soil infiltration rates are increased by between 84 and 400%. The ADAS Infiltration Excess model was used to simulate runoff for a range of soils where the default hydraulic conductivity was reduced by 80%. The ADAS Infiltration Excess model is based on rainfall event based solutions to the Green and Ampt (1911) equation, where default soil hydraulic conductivity was calculated using the HYPRES pedo-transfer functions (Nemes *et al.*, 1999) and rainfall intensity is estimated from an analysis of the kinetic energy of rainfall (Davison *et al*., 2005). The Infiltration Excess model was the source of the surface run-off sub-model in PSYCHIC. Using this model, Gooday *et al*., (2016) predicted an increase in calculated runoff of between 90 and 370% for a wide range of soil textures and daily rainfall totals for representative sites across Scotland, with an average value of 230%, which has been used an initial value for the impacts of compaction in Ireland.

Based on the results of the HOST and Infiltration Excess models, a three-fold increase in surface runoff and entrained pollutant emissions was therefore used to represent the impacts of machinery compaction from the affected area in affected fields (Table 3-18).

Table 3-18 Relative increase for losses from areas affected by soil compaction, poaching and waterlogging

Condition	Source Apportionment Coordinates	SS	P	N	N_2O
Machinery Compaction	Arable | Grass Runoff | Preferential | Air	3	3	3	2
Livestock Trampling	Grass Runoff | Preferential | Air	2	2	2	5
Seasonal Poaching	Grass Runoff | Preferential | Air	3	3	3	10
Feeder Poaching	Grass Runoff | Preferential | Air	3	3	3	10
Trough Poaching	Grass Runoff | Preferential | Air	3	3	3	10
Seasonal Waterlogging	Arable | Grass | Rough Runoff | Air	3	3	3	10

Emissions to air

Sitaula *et al.,* (2000) measured a 44% increase in nitrous oxide emissions from compacted (wheeled) plots of typic udorthents (USDA soil classification) and 170% from plots that had been fertilised. Ball *et al.,* (1999) measured average nitrous oxide emission rates from a heavy compacted drained loam soil in Scotland, which were 30 to 95% greater than from zero and light compacted plots. Hansen (2009) measured average nitrous oxide emission rates that were 1.4 to 8.3 times greater than from an uncompacted sandy loam soil, for plots receiving fertiliser. Based on this limited data, it has been assumed that soil compaction results in a 2-fold increase in nitrous oxide emissions from nitrogen applied in fertiliser, organic manure and excreta (Table 3-18). Taking account of the compacted area (10 to 15%) the net impact is a 5% increase in emissions from an affected field.

Soil compaction can also reduce the ability of soils to act as sinks for methane (Sitaula *et al.*, 2000). Methane is removed from the atmosphere by microbial oxidation in surface soils. Dobbie and Smith (1996), for example, measured annual average rates of 2.9 kg CH4 ha-1 from a wheat field and 1.4 kg CH4 ha^{-1} from set-aside on a loam sand soil in Scotland. Dobbie *et al.,* (1996) reported methane uptake rates of 0.3 to 0.9 kg CH4 ha^{-1} yr^{-1} for arable sites in Poland and Denmark. Flessa *et al.,* (2002) reported methane uptake in the range 0.3 to 0.7 kg ha^{-1} yr^{-1} for arable crops on organic and conventional farms in southern Germany. Le Mer and Roger (2001) review and cite median methane uptake rates of 2.0 and 2.3 kg CH4 ha^{-1} for arable and grassland. Hansen *et al.,* (1993) reported that compaction of agricultural soils reduced methane uptake by *c.* 50%. However, the soil methane uptake rate is small compared to the total enteric and manure emissions of methane on the representative farm systems. As a consequence, any effect of soil compaction on methane emissions was not represented for machinery compaction or any other soil condition.

3.4.2 Livestock compaction and poaching

Soil compaction by livestock can be either poaching (where hooves penetrate the sward and plastically deform the soil) which occurs when the soil is wet, or treading which occurs in medium and dry soil conditions. It was assumed in this study that poaching was found on 20% of fields for all livestock farm types. All fields with poaching damage were assumed to have a seasonal visibly poached area of 3% on and around gates and camping areas, and a more widely spread permanent area (20%) of less visible compaction and sparse vegetation cover (Table 3-16).

All grassland fields on grazing livestock farms were also assumed to have feeders and troughs. As dairy animals are frequently fed whilst waiting to be milked, there were assumed to be fewer fields with feeders on specialist dairy farms (40%). The poached area around a livestock feeder or trough can typically extend up to 20 m away from the feeder or trough. Assuming a circular area around the trough in a field of 6 ha, this equates to 2% of the field area, thus giving a total damaged area of 27%.

Emissions to water

Heathwaite (1995) measured surface runoff under simulated rainfall of 12.5 mm hr^{-1} for 4 hours on a clay soil that was equivalent to 2% of rainfall for ungrazed temporary grass; 11 to 12% for lightly grazed permanent grass; and 25 to 28% for heavily grazed permanent grass. Alderfer and Robinson (1947) similarly measured surface runoff under simulated rainfall of 35 mm hr^{-1} from clay loam and sandy loam soils. Runoff was equivalent to less than 2% of rainfall for ungrazed permanent grass; 1 to 58% for lightly to moderately grazed; and 33 to 80% for heavily grazed grass. In each case, increased runoff was correlated with vegetation removal, soil compaction and a reduction in the rainfall infiltration rate. Reviews of the impact of grazing on infiltration rates have concluded that light and moderate grazing reduce infiltration capacity to 75% of the ungrazed condition, and heavy grazing results in a 50% reduction (Gifford and Hawkins, 1978; Trimble and Mendel, 1995). Application of the ADAS Infiltration Excess model, as per machinery compaction, with the hydraulic conductivity of the soil reduced by 50%, increased the calculated runoff by between 43 and 97%, with an average value of 75%.

As well as increasing surface runoff due to compaction, the congregation of livestock around feeders and troughs reduced the vegetation cover to intercept any runoff and the increased time spent in these areas will result in higher levels of excretal inputs. Therefore the impact of livestock trampling was assumed to be a 2-fold increase in emissions, but poaching around feeders and troughs was assumed to result in a 3-fold increase in emissions relative to the default process model outputs (Table 3-18).

Emissions to air

Oenema *et al.,* (1997) in a review of nitrous oxide emissions from grassland cite a 2 to 3.6 fold increase of emissions due to compacted grassland soil. Bhandral *et al.,* (2007) measured nitrous oxide emissions from compacted grassland soils that were 3.6 to 6.7 times greater than from non-compacted soils receiving urine, ammonium and urea; and up to 18 times greater for soils receiving nitrate. van Groenigen *et al.,* (2005) reported that nitrous oxide emissions of urine applied to a sandy soil increased 5 fold when the soil was compacted under moist conditions, which was comparable to a factor of 3.5 reported by Yamulki and Jarvis (2002). Matthews *et al.,* (2010) reported nitrous oxide emissions from gateways and poached land around water troughs that were 10 times greater than from neighbouring

managed pasture. Finally, Smith and Smith (2004) used a constant multiplier of 2 for fields grazed by cattle; and 1.3 for fields grazed by sheep for an improved calculation of nitrous oxide emissions for Scotland. This was a landscape scale multiplier against emissions from mineral fertiliser that is assumed to represent the net effect of poached and non-poached fields. Based on this evidence, a nitrous oxide emission multiplier of 5 was used for the wider damaged soil area and a multiplier of 10 for the visibly poached areas (Table 3-18). There is no evidence for soil compaction having an impact on methane emissions.

3.4.3 Seasonal waterlogging

An area of high-risk for diffuse pollution due to waterlogged soils within a field was defined as an area of comparatively frequent and rapid generation of surface runoff, within a short distance of a receiving watercourse. Runoff generation is more frequent than elsewhere within the field because the soils are close to saturation, perhaps within a topographic hollow, or because field drains are not operating efficiently to control the water-table. This type of high-risk area does not occur on freely draining soils.

Based on data for drain failure in England (Anthony *et al*., 2012) and grassland field conditions in England and Wales (Forbes *et al.,* 1980), it has initially been assumed that 2% of the arable tile drained area is affected by water-logging, 5% of improved grassland, and 10% of all rough grazing land (Table 3-19).

Table 3-19 Percentage of land use which is affected by seasonal waterlogging

Land Use	Percent Waterlogged
Arable	2
Grass	5
Rough	10

Emissions to water

The relative effect of water-logging has previously been calculated for Scotland (Gooday *et al.,* 2016) by modifying the PSYCHIC model so that soils remained at field capacity all year. This resulted in an average 3-fold increase in surface runoff losses.

Emissions to air

The impacts of waterlogged soils on nitrous oxide emissions were assumed to be analogous to the effects of intensive poaching and set at a 10-fold increase (see Section 3.4.2).

3.4.4 Overall impact of compaction, poaching and waterlogging

The results of the pollutant modelling are discussed more fully in Section 5 but Table 3-20 shows the percentage contribution to the total load resulting from the addition of compaction, poaching and waterlogging. The impacts on nitrate are relatively modest (only 2.3% of the total load results from the 6 affected areas), whereas for phosphorus and sediment, the overall impact is more noticeable (7% - 12% of the total). The impacts on nitrous oxide are very significant, resulting in the load increasing by over a third (so that over

30% of the load comes from the affected areas). The modifications have no impact on methane losses.

Table 3-20 Contribution to the modelled agricultural pollutant loads for Ireland from the default, compacted, poached and waterlogged areas.

	N (%)	P (%)	Z (%)	N_2O (%)	CH_4 (%)
Default	97.7	93	88.2	68.1	100
Machinery Compaction	0.4	1.1	2.1	0.6	0.0
Livestock Trampling	0.5	1.5	2.4	6.7	0.0
Livestock Seasonal Poaching	0.2	0.5	0.7	2.3	0.0
Livestock Feeder Poaching	0.4	1.2	2	5.6	0.0
Livestock Trough Poaching	0.5	1.5	2.4	7.6	0.0
Waterlogged	0.4	1.1	2.4	9.1	0.0

3.5 Calculation of mitigation action effects

The effect of mitigation actions to reduce diffuse pollution is represented by applying percentage reduction factors to the emissions from the relevant source apportionment coordinate. The reduction in emission is proportional to an action effectiveness value; the efficiency of the action; the applicability of the action; and the implementation of the action. The effectiveness of mitigation is calculated separately for each source apportionment coordinate. Literature data, modelling and expert interpretation have been used to estimate the effectiveness of mitigation actions applied alone, with the values for each GLAS action described in the following section.

Due to the range and uncertainty in the evidence for mitigation effectiveness, values chosen for the percentage reduction factors are generally taken from one of a set number of values, which are: 2, 10, 25 , 50, 80 or 100%.

The modelling framework used in this project calculates the impact of implementing multiple mitigation actions that might target the same source apportionment coordinate. To account for this, the pollution reduction *R* due to each mitigation action is first scaled in proportion to the efficiency *E* of implementation:

$$R = P \cdot E \qquad \text{Eq. 3-1}$$

where *P* is the expected effectiveness under optimal conditions. The efficiency *E* of implementation represented local environment conditions that would hinder the effectiveness of a mitigation action. For example, the efficiency of a riparian buffer strip was expected to decrease with increasing slope of field.

If the mitigation actions were applicable to the whole of a source area, such as the total area of grassland, then the net reduction *N* due to a suite of mitigation actions was calculated using a multiplicative model as:

$$N = 1 - \prod_{i=1}^{i=n} (1 - R_i) \qquad \text{Eq. 3-2}$$

where R_i is the reduction due to an individual action. In the circumstances that one or more actions were not implemented across the whole source area, an assumption of maximum overlap of action uptake was made:

$$N = \sum_{j=1}^{j=n} A_{1:j} \cdot \left(1 - \prod_{i=1}^{i=j} (1 - R_i) \right) \qquad \text{Eq. 3-3}$$

where $A_{1:j}$ is the proportion of the source area affected by action 1 to j and N is the net effect of all the actions. In this case, the proportions A are the product of the action applicability and implementation values. An applicability value was used to represent situations where an action was limited by environmental constraints. For example, the implementation of contour ploughing was restricted to a fraction of fields in areas with steep slopes. The implementation value was simply the proportion of farms practicing the mitigation action. This is in effect an area-weighted version of Eq. 3-1.

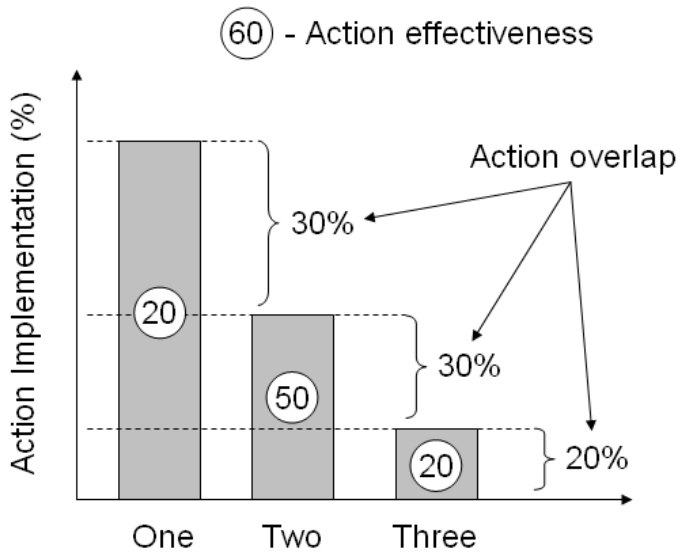


Figure 3-4 Schematic of the method of calculating the net effectiveness of multiple mitigation actions affecting the same pollutant source coordinate, assuming maximum overlap of action implementation.

Figure 3-4 makes this approach explicit. Three actions are shown on the figure of action implementation or uptake with values of 80%, 50% and 20%. They are ordered by decreasing uptake. The effectiveness values of the individual actions are 20%, 50% and 20%, respectively. The net effect is determined from the 30% of the farm source area impacted by only the first action with an effectiveness of 20%; the 30% of the farm source area impacted by the first and second actions with a combined effectiveness of 60%; and the 20% of the farm source area impacted by all three actions with a combined effectiveness of 68% (Eq. 3-2). The net effect is a weighted sum of these combined effectiveness values, where the weight is the area of overlap. In this case, the combined effectiveness of all three actions (including the part of the source area that is not impacted by any action) is 37.6% (Eq. 3-3).

The method of calculating the net effectiveness of multiple actions assumed that actions are acting on the same potential pollutant source. Therefore, the gain from additional actions targeting the same source apportionment coordinate decreased rapidly. This is not a perfect model but was thought to be better than the alternative additive model in which the pollutant source is quickly exhausted and the impact of multiple actions over-estimated. The explicit source apportionment coordinate system minimised the risk of erroneous competition between mitigation actions for effect. The calculation of mitigation effects also permitted an increase in pollutant emissions. For example, an increase in nitrate leaching following rapid incorporation of slurry and conservation of nitrogen that previously was emitted as ammonia.

4 GLAS

This chapter describes the structure of GLAS and the actions that are implemented as part of the scheme and explains how they are represented in the modelling framework.

4.1 Introduction

The Green Low Carbon Agri-Environment Scheme (GLAS) is a highly targeted scheme. Key to its design was the identification of a number of Priority Environmental Assets (PEAs) – primarily vulnerable landscapes (including Natura and uplands), species at risk (primarily endangered birds), and high-quality watercourses. All holdings with these physical assets were pre-identified and the areas of importance mapped at farm level. Presence of one or more of these assets on any farm guaranteed priority access to the scheme. Environmental assets and actions of lesser importance have also been identified, and these can be addressed once the over-riding objective of addressing the Priority Environmental Assets has been met.

GLAS has a three tier hierarchy and this structure is designed to ensure the targeted and prioritised delivery of environmental benefits.

Tier 1 is the most important Tier, comprising Tier 1(a) all the Priority Environmental Assets identified for support through GLAS, targeting vulnerable landscapes, species at risk and protection of high-status watercourses and Tier 1(b) a series of Priority Environmental Actions for intensive farmers, targeting climate mitigation and farmland birds. Organic farmers also receive priority access to the scheme under Tier 1 in their own right.

Tier 2 is the next most important tier and focuses in Tier 2(a) on water-quality, through protection of predetermined vulnerable water-courses, while also accepting proposals under Tier 2(b) from other farmers who are prepared to take on predetermined actions again targeting climate change mitigation and supporting farmland birds.

Tier 3 is largely a feeder menu of complementary environmental actions for applicants approved into Tiers 1 and 2. It consists of actions such as the protection of traditional hay meadows, species-rich pastures, important landscape features like archaeological monuments, hedgerows and stone-walls, as well as provision of bird, bat and bee nesting facilities and the planting of small groves of native trees.

Figure 4-1 shows WFD waterbody status for each waterbody in Ireland. High status waterbodies correspond to Tier 1, whilst those less than Good status are 'Vulnerable' and thus correspond to Tier 2.

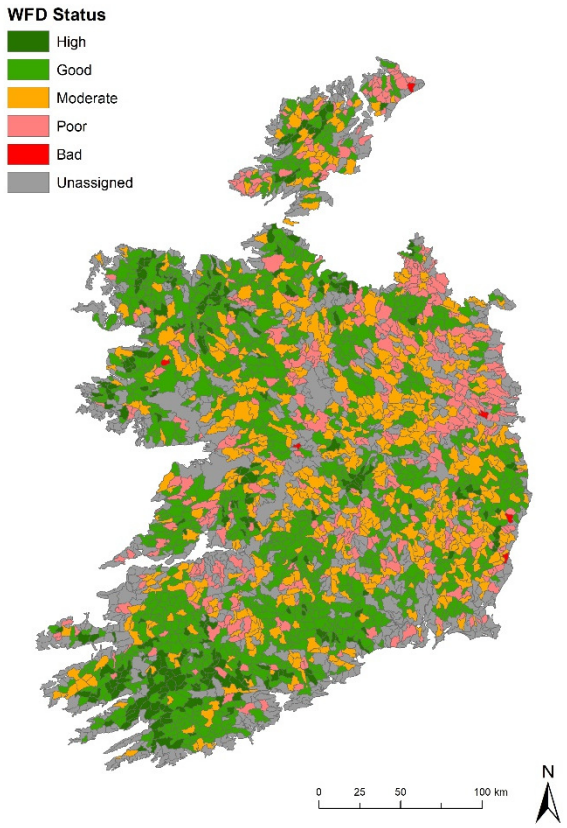


Figure 4-1 WFD Waterbody status, as used in the targeting of GLAS tiers.

ADAS were provided with the GLAS agreement data for tranche 1 and tranche 2 of the GLAS scheme, which closed for entry in to the scheme at the end of 2015. Approximately 25% of current GLAS participants (as of May 2019) are in tranche 3, which has not been included in this analysis. This tranche 1 and 2 dataset included 280,000 different actions on almost 40,000 different holdings. Herd and parcel identifiers were included to allow the actions to be linked to LPIS and agricultural census information so that they could be analysed spatially and by farm type.

There are over 20 actions within GLAS, some of which are mandatory depending upon the entry tier, the size of the farm or stocking density on the farm or if any of the priority environmental assets are applicable to the farm. The following is a list of all GLAS actions, with those that are assumed to have an impact on diffuse agricultural pollution underlined.

- Arable Grass Margin
- Bat Boxes
- Bird Boxes
- Conservation of Solitary Bees (Boxes)
- Conservation of Solitary Bees (Sand)
- Conservation of Farmland Birds
    1. Breeding Waders
    2. Chough
    3. Corncrake
    4. Geese and Swans
    5. Grey Partridge
    6. Hen Harrier

7. Twite

- Catch Crop
- Commonage Management Plan (CMP) and Commonage Farm Plan (CFP)
- Coppicing of Hedgerows
- Environmental Management of Fallow Land
- Farmland Habitat (Private Natura)
- Laying of Hedgerows
- Low-emission Slurry Spreading
- Low-input Permanent Pasture
- Minimum Tillage
- Planting a Grove of Native Trees
- Planting a New Hedgerow
- Protection and Maintenance of Archaeological Monuments
- Protection of Watercourses from Bovines
- Rare Breeds
- Riparian Margins
- Traditional Dry Stone Wall Maintenance
- Traditional Hay Meadow
- Traditional Orchards
- Wild Bird Cover

Alongside these actions, a Nutrient Management Plan (NMP) was seen as a tool to assist farmers to meet the objectives of GLAS in tandem with the delivery of their chosen actions and so was made a core requirement of the scheme.

In the following sub-sections, the GLAS actions that impact on diffuse pollution are outlined, along with evidence on typical impacts. In the modelling framework, actions are parameterised as a percentage reduction in one or more aspects of the pollutant coordinate system (Table 3-14) as described in Section 3.5, so each sub-section concludes with the apportionment targeting for that action along with a table showing the national level implementation. A summary of the representation of GLAS actions is included as an Annex to this report.

4.2 Representation of GLAS actions in the modelling framework

4.2.1 Nutrient Management Plans

Objective

To improve nutrient efficiency while at the same time achieving the environmental objectives of GLAS

Definition of action

All GLAS participants must have a NMP prepared by a GLAS advisor, with the plan prepared on the Teagasc NMP online system. The requirements of the NMP were that all land farmed must be soil sampled and it should outline the total inorganic N and P use figures for the whole farm, including any adjustments made for GLAS actions that have an inorganic N restriction. The NMP reports produced for each farm contain:

- Farm summary of soil fertility and fertiliser requirements
- Field and farm level lime report
- Field and farm level fertiliser plan
- Summary of land areas, cropping and max fertiliser allowances
- Summary of all livestock on the holding
- Summary of manure exports / imports
- Concentrate feed usage on the farm
- Soil sample results

Evidence for effect

Newell-Price et al. (2011) state that a fertiliser recommendation system could reduce fertiliser usage by around 5%, which would reduce losses associated with fertiliser inputs by up to 5%. Where manures are applied, reductions in fertiliser use could be 10-15% if full account is taken of the available N in manures.

Using data from fertiliser response trials, Sylvester-Bradley et al. (2008) found errors in recommended fertiliser rates of ±50 kg ha^{-1}, so use of a recommendation system may not reduce the excess fertiliser post-harvest which is vulnerable to leaching.

In an assessment of the impacts of the Nitrate Vulnerable Zone Action Programme in England and Wales, Lord et al. (2008) estimated changes in the quantity of fertiliser applied due to improved accounting for manure N supply had reduced overall nitrate losses by 4%. However, they state that there have been improvements in accounting for manures since the 2002 baseline used in this assessment and, more importantly, that the expected impact of this measure on grassland is negligible as most grassland receives much less than maximum recommended amounts.

Representation

Figure 4-2 shows farm level fertiliser and stocking rates from GLAS nutrient management plans, compared to national fertiliser survey data (Dillon *et al.*, 2018). Theoretically, fertiliser use on GLAS farms should be lower due to both NMPs and due to the fertiliser restrictions associated with a number of GLAS options, which occupy a large proportion of the grassland area of GLAS farms. However, there is insufficient evidence to draw a conclusion as to whether fertiliser rates are lower (for a given stocking density) which could indicate changes resulting from a NMP. The modelling framework does thus not include any impact associated with a NMP.

The NMPs contain information on soil P indices, but the GLAS scheme has not been running long enough for any potential impacts on P indices to have been realised that could be attributable to the scheme. There was also no definitive evidence available that P fertiliser use has changed, which might eventually result in changes in P indices.

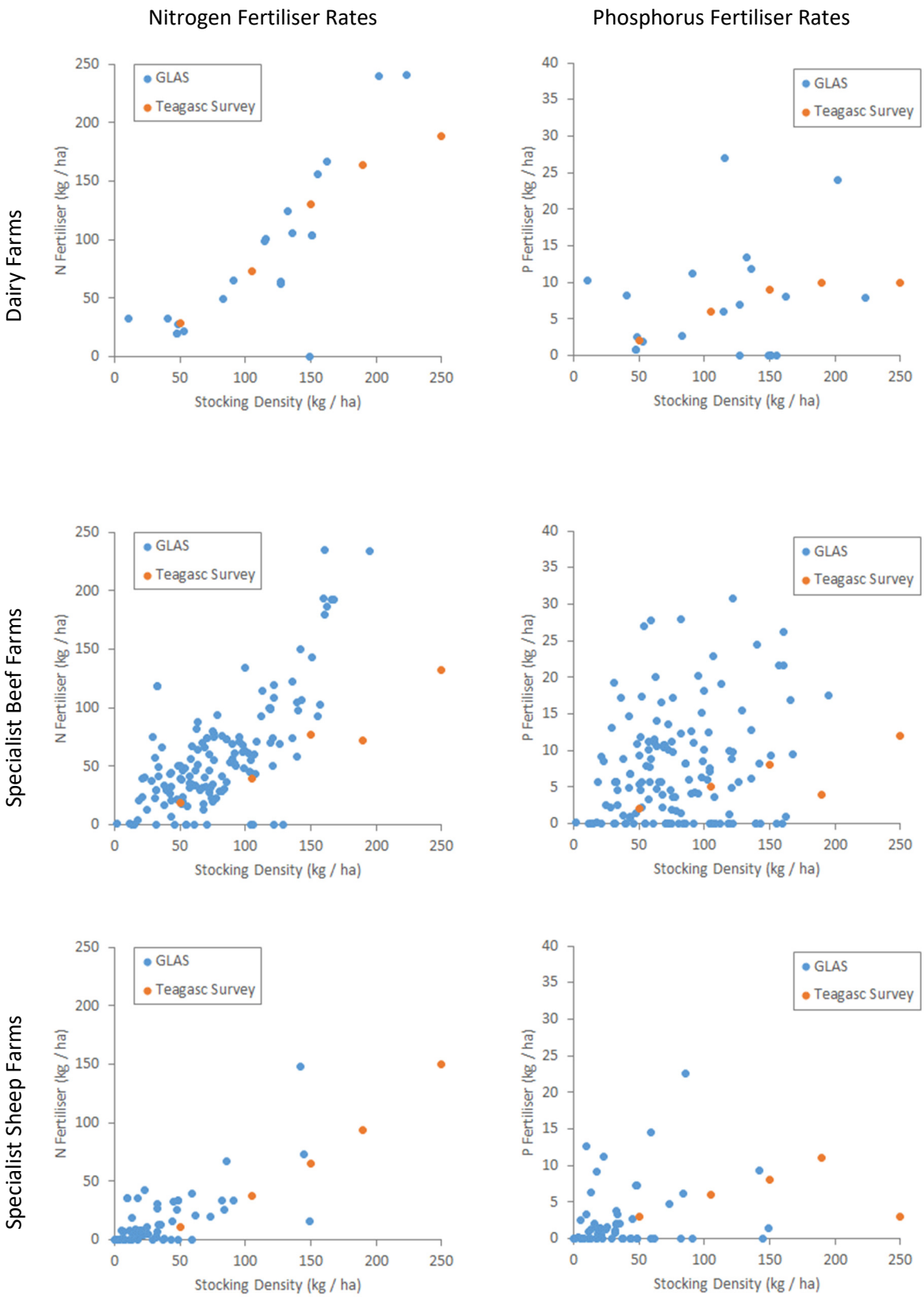


Figure 4-2 Comparison of nitrogen and phosphorus fertiliser application rates for farms in GLAS (using data from GLAS Nutrient Management Plans for 2016-2018) and average rates the whole of Ireland (Dillon *et al.*, 2018).

4.2.2 Minimum Tillage

Objective

To improve soil structure and increase soil organic matter.

Definition of action

Minimum tillage (which involves sowing a crop without inverting the soil) has many advantages for both the farmer and land as it is assumed to protect soil from erosion and minimise compaction, leading to reduced runoff. There are also secondary effects in terms of reduced time and fuels usage and potentially an increase in soil organic carbon. This measure also protects archaeological monuments within the topsoil and subsurface of the soil.

Evidence for effect

One of the major benefits of conservation/reduced tillage is the reduction of runoff carrying nutrients/sediment (Duiker & Myers, 2005; Busari et al. 2015). It has been reported in the USA that no tillage practices have resulted in a reduction of soil erosion on cropland of more than one third between 1983 and 1997 (from 3.1 billion tonnes to 1.9 billion tonnes) (Claassen, 2012). Catchment scale studies in the USA have shown conservation tillage to reduce runoff and sediment losses by 64 and 99% respectively (Clausen et al. 1996; Holland, 2004).

Crittenden et al. (2015) looked into the differences in physical soil properties between non-inversion tillage and conventional tillage. They found that soil structural parameters such as aggregate stability and penetration resistance were higher in non-inversion tillage trials, creating a denser and more stable soil without penalties on crop yield. Field-saturated hydraulic conductivity was also higher in the non-inversion tillage trials (in autumn measurements only) suggesting that reduced tillage could provide better soil water storage over the winter months.

Maetens et al. (2012) conducted a meta-analysis of a number of different soil conservation techniques, including minimum/reduced tillage in the Mediterranean and Europe. Reduced tillage reduced the amount of runoff by an average of around 8%, whilst no tillage reduced runoff by around 15% on average. Reduced tillage mitigated soil loss by an average of around 38%, and no tillage by an average of around 46%.

In a UK study, Deasy et al. (2009) demonstrated that TP losses under minimum tillage (compared to traditional ploughing) were reduced by 0.3 kg ha^{-1} TP over 5 site years. Deasy et al. (2010) reported a 4-81% reduction in runoff, a 37-98% reduction in suspended sediment loss, a 29-97% reduction in total phosphorus loss and a 26-94% reduction in total nitrogen loss when minimum tillage is used compared to conventional tillage.

Regan et al. (2012) noted that whilst reduced tillage can reduce levels of particulate P lost from a field, a build-up of P near the soil surface can occur in reduced tillage circumstances, increasing the risk of P loss in runoff.

Representation

Due to reduced stimulation of mineralisation, nitrate losses from soil organic matter are assumed to be reduced by 10%. Incidental losses of nitrate and dissolved phosphorus in surface runoff are reduced by 10%, whilst particulate phosphorus and sediment losses in surface runoff are reduced by 50%.

The action was applied to the proportion of the agricultural land where the minimum tillage action was applied (Table 4-1).

Table 4-1 Uptake of minimum tillage

Action	Area (ha)
Minimum Tillage (part of parcel)	107
Minimum Tillage (whole of parcel)	6,246

4.2.3 Low Emission Slurry Spreading

Objective

To improve the recycling of organic fertiliser and to contribute to reduced nitrous oxide emissions, ammonia emissions and odours.

Definition of action

The method and timing of slurry application are two important factors that determine the utilization efficiency of nutrients by the crop (grass or arable). Using Low-Emission technology improves the utilisation efficiency of slurry compared to the traditional splash-plate. Other benefits include, reduced phosphorus run-off, a wider window of opportunity to apply slurry, reduced tainting of the grazing sward and reduced smell from slurry spreading.

All of the slurry applied on the farm (produced and/or imported) must be spread by one or a combination of the following methods for each year of the contract and must be spread in compliance with the Nitrates Regulations.

a) Band Spreading
b) Injection Systems
c) Trailing Shoe

Evidence for effect

Although ammonia emissions are outside the scope of this project, the potential for reducing nitrate losses due to a reduced need for manufactured fertiliser inputs are directly related to savings in ammonia which result in more manure-N being available for crop uptake.

Webb et al. (2010) conducted a literature review into different methods of slurry application and their effect on ammonia and nitrous oxide abatement. They report abatement of ammonia emissions from slurry spreading via trailing shoe of 65%, and via injection of 70-80%. Trailing hose also has some effect of reducing emissions (35%). However, they report the greatest reduction in ammonia emissions (90%) when slurry is immediately incorporated, regardless of application method. It is noted that some techniques applied to reduce ammonia emissions may act to increase nitrous oxide emissions (Van der Zaag et al.

2011; Dosch & Gutser 1995; Perala et al. 2006). However, this is not a guaranteed effect and not thought to outweigh the benefit of the techniques. Rapid incorporation following trailing shoe mitigates this effect (Wulf et al. 2002ab), in some studies deep injection has been shown to reduce nitrous oxide emissions as well as ammonia (Webb et al. 2010). It is clear there is some degree of trade-off between ammonia and nitrous oxide emissions when considering different methods of slurry application.

Sharpe and Harper (1997) reported that 13% of total available N can be lost from swine lagoon effluent as it is sprayed through the air when applied by sprinkler method (Van der Zaag et al. 2011). Deep injection is generally considered to be the most efficient method at reducing emissions, reducing ammonia emissions by 90% when slurry is applied to arable land by deep injection compared to slurry being spread on the surface of bare soil (Rotz, 2004; Van Der Zaag et al. 2011). The best approach to reduce nitrous oxide emissions simultaneously with ammonia emissions is dependent on soil conditions and land use, with injection preferable on well drained arable fields due to reduced ammonia with little effect on nitrous oxide; band spreading with immediate incorporation preferable on moist soils; trailing shoe preferable on grassland (Van Der Zaag et al. 2011).

Huijsmans et al. (2001) reported mean cumulative volatilization for surface spread slurry to be 77% of TAN applied, 20% for bandspread and 6% for shallow injection. They also noted the influence that external factors such as weather and grass sward height can have on volatilization. Smith et al. (2000) reported ammonia losses of 40, 25, 23 and 17% from slurry spread by splash-plate, band spreading, trailing shoe and shallow injection respectively. This corresponded to a 39, 43 and 57% reduction in emissions by band spreading, trailing shoe and shallow injection respectively, relative to conventional surface broadcast application techniques.

Dowling et al. (2008) reported reductions of 28 and 40% (in different study years) in the emission of ammonia from trailing shoe application compared to splashplate application of slurry. Similarly, Bourdin et al. (2014) reported a 24% reduction in ammonia emissions via trailing shoe compared to band spreading.

A study commissioned by TEAGASC (2011) into developing strategies to control ammonia emissions from cattle slurry in Ireland found that on average 54% of TAN was lost when slurry was applied by splashplate, with emissions being influenced by weather conditions. Trailing shoe application was found to reduce emissions by an average of 36%.

McConnell et al. (2012) examined the effects of low emission slurry spreading techniques on phosphorus losses on grass and stubble. On stubble plots, DRP concentrations in runoff were 47 and 37% lower from injection and trailing shoe methods respectively compared to splash-plate, 2 days after slurry application. On the grass sward plots, application technique was found to have no effect of DRP concentrations in runoff.

Newell-Price et al. (2011) state that the reductions in ammonia from slurry band spreading and injection will increase the potential for nitrate leaching losses and direct and indirect nitrous oxide emissions, whilst the impact on phosphorus losses are likely to be minimal.

Representation

The primary aim of Low-Emission Slurry spreading is to reduce ammonia emissions. In theory, this conserved N should be available to the crop, allowing for a reduction in inorganic fertiliser inputs. However, as shown in Figure 4-2, there is limited evidence in a

change in nitrogen fertiliser applications between GLAS and all Irish farms. Thus on arable land, there is the potential for the ammonia saving to results in more nitrate being leached, but on grassland the sward would have sufficient capacity for nitrogen uptake to limit the potential increase in nitrogen losses.

Figure 4-2 shows that nitrogen fertiliser rates are approximately equal to stocking rates for Specialist Dairy and Specialist Beef systems. Assuming a stocking rate of 100 kg N ha^{-1} for a dairy farm, where approximately one third of the excreta would be managed as manure, results in 30 kg N ha^{-1} of manure (assuming no nitrogen-transformations in storage). Assuming that 60% of the manure is TAN, and the ammonia emissions reduce from 60% to 10% due to low emissions spreading, this would result in a 9 kg ha^{-1} saving, but as stated in Section 4.2.1, there is no evidence of a change in fertiliser use on GLAS farms.

This action has the potential to increase losses if the additional manure available N is not accounted for, or reduce them if less fertiliser is applied. Given the lack of evidence, a neutral position has been adopted in the modelling framework, with this action having no impact on the pollutants considered.

4.2.4 Catch Crops

Objective

To establish a catch crop that will absorb nutrients and prevent leaching in the autumn/winter period.

Definition of action

The primary aim for catch crops is for soil protection during fallow periods over the winter period as catch crops reduce soil erosion during heavy rainfall, provided better soil structure and aid the absorption of residual nitrogen, reducing nitrogen leaching from the soil. There is a reduction in soil erosion during heavy rainfall periods from reduced surface run-off and increased water infiltration. While protecting soil against exposure to the elements with foliage, cover crop roots break and condition the soil preventing slumping, thus ensuring easier cultivations and better soil tilth the following spring. Depending on the species, catch crops increase the absorption of residual nitrogen and reduce nitrogen leachate from soil.

Catch crops are to be established annually by the 15th September using light cultivation techniques and by either broadcasting or drilling as seed mixture that consists of a least 2 species from a specified list. The catch crop must remain in situ from the date of sowing until the 1st Dec unless the catch crop is sown to comply with the geese and swan actions at which point it must remain in place until 31st March annually. Catch crops can be grazed after the 1st December in fields where soil erosion is not considered to be an issue.

Evidence for effect

Randall et al. (2013) reviewed around 718 studies looking into the effectiveness of various agri-environmental measures on environmental impacts. 86% of the studies investigated the effectiveness of catch crops on reducing N (n=114), with 72% reported by authors be generally effective at reducing N (n=82). A meta-analysis was conducted on 10 studies suggested that catch crops are effective at reducing N loss compared to a fallow control. 14% of the catch crop studies looked into the effectiveness of catch crops on reducing sediment loss (n=19), and these indicated that catch crops were effective at reducing

sediment loss in 68% of the studies (n=13). For phosphate, 10% of the catch crop studies were relevant (n=14). Of these studies, only 3 reported an effect of catch crops reducing P loss.

Ground cover of between 70 and 80% is required to minimise the effects of soil runoff and loss, reducing surface runoff by between 10 and 30% and soil erosion by between 50 and 80%. The measured effects of purpose-sown cover crops best illustrate this. Schonning et al. (1995) reported that a rye grass cover crop before spring barley reduced the total soil loss by between 89 and 97% and the total phosphorus loss by between 91 and 92% relatively to a control treatment. Stevens and Quinton (2009) reported sediment reductions in the range 7 to 87% with an average value of 52%, and Novotny and Olem (1984) reported reductions in sediment and phosphorus losses in the range 30 to 60%.

As little as 1 t ha^{-1} of stubble can provide some protection to the soil from water erosion. The BTO (2002) measured over-winter ground cover by stubble and weeds at monthly intervals following harvest of crops with and without herbicide applications on 122 fields in eastern England. The stubble was not incorporated. On the majority of fields that were sprayed with glyphosphate, the average proportions of ground covered over the winter period ranged from 12% for winter wheat, 26% for barley and 34% for oilseed rape. On the fields that had not received any desiccant, the ground cover averaged an additional 10% for wheat, barley and maize crops. Based upon these data, stubble and weed growth could achieve a maximum of 50% cover and a similar proportion of rainfall energy intercepted.

A study by Plaza-Bonilla et al. (2015) showed that whilst increasing the number of cereal legume crops in a rotation increases the amount of nitrate leached and lost in drainage water, the effect can be mitigated by the establishment and incorporation of cover crops during fallow periods. Basche et al. (2014) conducted a meta-analysis of 26 papers looking into the effects of cover crops on nitrous oxide emissions. They found that cover crops increased nitrous oxide emissions from the soil surface in 60% of the studies, and decreased nitrous oxide emissions in the remaining 40% of studies.

A 3-year study by Premrov et al. (2012) showed a significant reduction in groundwater nitrate concentrations over the three year period after the establishment of a mustard cover crop, compared to no cover, the reduction was from 23.9mg/l to 18.0mg/l. The study also showed an increase in dissolved organic carbon concentrations under the mustard cover crop, which could potentially further aid the denitrification of groundwater.

The scheme specifications state that grazing of catch crops is permitted after the 1st December. If grazing happens extensively on catch crops, this may have some implications for how effective the measure can be. Whilst it is stated that grazing can only take place on parcels where an advisor considers there to be no issues with soil erosion, there are several negative impacts other than soil erosion associated with the grazing of catch crops. A report for the Welsh government (Jones et al. 2012) looking into the environmental sustainability of over winter grazing of forage crops identifies some of these issues, by assessing several field measurements in grazed and un grazed areas of 12 fields established with an over winter forage crop. There was a reduction in the amount of vegetation cover provided by the crop, from 87.3 and 91.8% to 11.4 and 10.4% in cattle and sheep grazed sites respectively. The soil physical characteristics were also negatively impacted, with an increase in soil compaction at all grazed sites. Water infiltration rates were reduced by 95% across all 12 sites. This could have implications for runoff rates and therefore increased transport of pollutants to

watercourses. Indeed, many of the sites studied showed increased concentrations of nitrate and ammonium, compounding the effect of increased rates of runoff.

Representation

Almost half of the respondents to the GLAS attitudinal survey (Cao, 2018) stated that they intended to graze their catch crops. The effectiveness values used are thus lower than some of those quoted above because of reduced vegetation cover and compaction resulting from livestock grazing the field and also the return of crop nitrogen uptake as excreta.

Catch crops were assumed to reduce over-winter losses of nitrogen by 25% and soil erosion (and associated particulate phosphorus) by 25%.

There is the potential to reduce fertiliser applied to the crop following the catch crop if adjustments in application rates are made to account for the greater soil mineral nitrogen in spring and the nitrogen in the catch crop residues. This could be in the range of 10-20 kg ha^{-1}, which is approximately 10% of the N applied to arable crops. This impact has not been considered in the modelling work.

The action was applied to the proportion of the spring cropping land where the catch crops were grown (Table 4-2).

Table 4-2 Uptake of Catch Crops

Action	Area (ha)
Catch Crops (part of parcel)	3,306
Catch Crops (whole of parcel)	15,825

4.2.5 Wild Bird Cover

Objective

To sow a seed crop mix that provides a food source and winter cover for farmland birds and other fauna.

Definition of action

This measure involves establishing a spring sown cereal mix which will remain unharvested, providing a food source for farmland birds over the winter period. The crop must remain in situ until the 15th March the following year, after which livestock may enter. Fertiliser can be applied at a maximum of half rate for a cereal crop.

Evidence for effect

As this action is primarily ensuring over-winter cover, the evidence from the impacts of this action are the same as those for the Catch Crop action.

Representation

Although the action allows fertiliser to be applied at a low rate, it was assumed that no fertiliser would be applied as it was thought unlikely that farmers would grow a low quality crop. Losses associated with fertiliser on this land were thus entirely negated.

The effectiveness of this action may be higher than some of the published evidence on catch crops as there will be no cultivation in the autumn to stimulate mineralisation, but there may be limited over-winter nitrogen uptake as the crop should be fully established. However, unlike the Catch Crop action, the land cannot be grazed over winter. Thus Wild Bird Cover was assumed to reduce over-winter losses of nitrogen by 25% and soil erosion (and associated particulate phosphorus) by 50%.

The action was applied to the proportion of the spring cropping land where the wild bird cover was grown (Table 4-3).

Table 4-3 Uptake of Wild Bird Cover

Action	Area (ha)
Wild Bird Cover (part of parcel)	8,431
Wild Bird Cover (whole of parcel)	5,283

4.2.6 Planting New Hedgerows

Objective

Plant new hedges along fence lines and use them to break-up the hydrological connectivity of the landscape.

Definition of action

Increasing the number of hedgerows can help to reduce sediment and associated nutrient loses by reducing surface runoff and trapping nutrients. Hedges can also help to reduce soil erosion due to wind. Installing hedges reduces the slope length and helps to prevent the delivery of pollutants in surface runoff by reducing the force of flow. Hedges also act as natural buffers strips and sediment traps and enables separate parts of the landscape to be managed in different ways.

The measure is applicable to most farming system but is likely to be more applicable to the arable sector where hedgerows have been removed.

Evidence for effect

Richet et al. (2017) conducted a study on the roles of vegetative barriers, seven fascines and three shrub hedges, in catchment management to reduce runoff and erosion effects using a field runoff simulator in Normandy. The ridge and leaf litter under the hedges was found to have a significant effect on the flow rates. The litter was kept in place but other vegetation in higher flows, for example, netters, brambles and small shrubs increasing the hydraulic roughness and reducing flow rate. The result given describe the optimal design of vegetation barriers and their efficiency when placed immediately downstream of erosion sources, across channels of concentrated runoff or immediately upstream of local assets at risk.

Van Vooren et al. (2017) quantitatively assessed the impact of hedgerows and grass strips bordering parcels with annual arable crops. After a systematic literature review, observations were made on 60 studies for modelling. Nitrogen interception from the surface flow was positively affected by hedgerows with the average interception being 69%. Hedge width was found to be a significant explanatory variable, a 2-m and 5-m wide hedge was found to have nitrogen interception of 42% and 72%. Average phosphorus interception was 67%, 3 studies however reported negative values due to the adverse effect from vegetation decomposition and sorption/desorption. Soil sediment interception was 91% and hedgerows are very effective under a wide range of circumstances.

Representation

The calculation of landscape connectivity (Section 2.4) includes factors for the retention caused by different boundary features. It was assumed that a hedgerow would be replacing an existing fence such that the connectivity for that field would be reduced by 75%. Pollutant losses in surface runoff were thus reduced by 75%.

The action was applied to the proportion of the arable and grassland fields where a new hedge had been implemented. The total length of new hedges is shown in Table 4-4. New hedges were located on almost 9000 field parcels.

Table 4-4 Uptake of Planting New Hedgerows

Action	Length (m)
Planting New Hedgerows	1,295,903

4.2.7 Protection of Watercourses from Bovines

Objective

To protect water quality by excluding bovines from watercourses.

Definition of action

Livestock grazing along a watercourse can lead to direct pollution of water with urine and faeces which could mean pathogens entering the water. This livestock grazing can destroy aquatic habitats and lower the quality of water that could potentially enter the water that humans use. Excluding bovines from watercourses will prevent the breakdown of vegetation on the banks of the watercourse. There will be direct effects on bank erosion and sediment loss as well as reduced deposition of excreta directly into the watercourse.

Fence must be located at least 1.5m from the top of the bank of the watercourse and the fencing must be stock-proof, fit for purpose and be undertaken with permanent stakes and wire. Livestock drinking points are not permitted, an alternative water supply must be provided.

Evidence for effect

Section 3.1.4 outlines the evidence on the amount of time cattle can spend in watercourses, and how much excreta and urine is deposited whilst there. Fencing should eliminate these losses entirely.

Bank erosion was outside the scope of this assessment and so it has not been considered. However, the introduction of fencing or alternative drinking water sources has the potential to reduce local bank retreat rates by between 50 and 80%. This estimate is based upon Collins et al (2010), who reported that erosion of channel banks in the south west of England fell by an average of 40% following the introduction of riparian fencing, and a small number of studies, most from the United States, which monitored the impacts of cattle grazing on bank retreat rates (Agourdis *et al*., (2005); Zaimes *et al*. (2004) Kauffman *et al*. (1983) Sheffield *et al*. (1997)). These studies indicated that bank retreat rates may be 2 to 5 times greater where cattle have unrestricted access to a watercourse, and depend on it for their water supply.

Representation

The modelling framework explicitly represents nutrient losses resulting from livestock having direct access to water. This action is assumed to entirely negate these losses.

The modelling framework does not represent sediment losses due to bank erosion and so it was not possible to quantify the impacts of bank stabilisation associated with fencing.

The action was applied to the proportion of the grassland fields next to water that were now fenced. The total length of fencing is shown in Table 4-5. The action was implemented on 44,000 field parcels.

Table 4-5 Uptake of Protection of Watercourses from Bovines

Action	Length (m)
Protection of Watercourses from Bovines	12,745,368

4.2.8 Arable Grass Margins

Objective

To provide a habitat for flora and fauna, increase biodiversity and help protect water quality.

Definition of action

The establishment of a 3, 4 or 6 metre arable grass margin along the full length of an existing field or LPIS parcel boundary will increase diversity on the farm. When established along a watercourse it acts as a buffer zone to intercept sediment and nutrients. The land for the margin will be taken out of arable production and so there will also be a reduction in inorganic nutrient inputs.

The parcel selected must remain as arable, and arable grass margins must remain in situ for the duration of the contract. An additional 2m of land unsown with arable crop must be in place between the watercourse and arable grass margin if running along a watercourse. Soil cultivation and fertilising or liming cannot be carried out once the margin has been

established, and pesticides may only be used for spot treatment of noxious or invasive weeds. The margin is to be sown with a grass seed mix containing at least 60% Cocksfoot and Timothy or a combination and must be mulched, mown or grazed at least once a year between 15th August and 1st March.

Evidence for effect

For sediment, the reported trapping efficiencies of buffer features are frequently in excess of 90%, and typically in excess of 50%, depending upon slope length, vegetation density, sediment size and the risk of channelization (Niebling and Alberts, 1979; Barfield and Albrecht, 1982; Hayes and Hairston, 1983; Hayes et al., 1984; Rose et al., 2002; Dorioz et al., 2006). Because pollutants, including phosphorus, are preferentially associated with fines and buffer zones preferentially trap coarser sediment particles, the nutrient trapping efficiency of buffer features is frequently less than the sediment retention efficiency (Dillaha et al., 1989; Hayes et al., 1984; Robinson et al., 1996).

For phosphorus, Dorioz et al. (2006) reviewed the literature on the effect of grass buffer strips on phosphorus dynamics. Riparian strips were reported to have a 49 to 93% reduction in total phosphorus, and 53 to 98% reduction in sediment. Dissolved phosphorus concentrations also were generally reduced by up to 89% but occasionally were increased during transfer across the buffer strip. Doriaz et al. (2006) concluded that the most common values for control are in the range 20 to 30%. Balana et al. (2012) also analysed the review dataset of Collins et al. (2009) to estimate phosphorus retention for 2 m buffer strips of 30% and for 8 to 20 m buffer strips of 75 to 97% for medium slopes (4 to 13°). Abu-Zreig et al. (2003) working in Ontario, Canada, concluded that the overall average phosphorus trapping efficiency of buffers comprising different grass species was 61%, with respective means of 31% and 89% for 2 m and 15 m wide filters. The authors concluded that filter width was the primary control on trapping efficiency, whilst rate of inflow, vegetation type and vegetation density represented important secondary factors. Also working under natural rainfall conditions, Patty et al. (1997) reported that filter widths of 6, 12 and 18 m removed, on average, 40%, 52% and 87% and 92%, 98% and 99% of the soluble phosphorus and sediment loads.

For nitrate, based on the review dataset in Collins et al. (2009), the average nitrate retention for buffers of 6 to 12 m wide is 61% (n 21; 17 to 100% range). White and Arnold (2009) similarly report retention of nitrate in the range 3 to 100% with an average value of 67% (n 35) based on data taken from separate 22 publications. Mayer et al. (2005) carried out a meta-analysis of published data and fitted an exponential regression model to predict total nitrogen retention as a function of buffer strip width. Retention was in the range 14 to 28% for surface runoff across buffer strip widths of 6 to 12 m.

Miller et al. (2015) carried out a study looking into the effect of mowing buffer strips. They found there to be no effect of mowing on the concentration of sediment, nutrients and bacteria. However, they did find that increasing the width of the buffer significantly increased the mass (not concentration) of sediment, nutrients and bacteria leaving the plot. Reduction ranged from 29 to 92% for total dissolved solids, 38 to 93% for total P and 23 to 92% for total N.

It is noticeable that literature values for the efficacy of filter strips rarely distinguish between plot level and whole field treatments. As the runoff volume and the risk of rilling is a factor in their efficacy, literature reviews may over estimate their effectiveness if they include results

for buffer strips that occupied the greater part of a field. Therefore, the review dataset published by White and Arnold (2009) was re-analysed to extract the retention efficiencies for a sub-set of buffer strips with more realistic field to buffer area ratios. The sediment trapping efficiency averaged 73% for buffer strips with realistic ratios in the range 10 to 40 (i.e. 2.5 to 10% of field length; n 17). Nitrate retention averaged 52% (n 9). Total phosphorus retention averaged 51% (n 9) and soluble phosphorus retention averaged 37% (n 9).

The effectiveness of vegetated buffer strips in filtering particulate and dissolved pollutants decreases within increasing field size and slope as a consequence of increased runoff velocity and likelihood of rill and gully formation. Dillhala et al. (1988; 1989), for example, reported 7 to 38% reductions in the trapping of suspended solids as slope increased from 6 to 11 degrees. Kronvang et al. (2003) also reported that the probability of sediment delivery across a buffer increased with rill erosion.

Liu et al. (2008) in a major review of the efficacy of vegetated buffers for sediment trapping fitted regression models to data from 80 studies to predict sediment removal as a function of buffer width. Efficacy increased from 65% for 2 m strips to 80% for 6 m and 90% for 12 m strips. In a previous model based assessment of the cost-effect of buffer strip placement in Scotland Balana et al., (2012) estimated the relative efficacy of 6 m strips to decline from 100% for slopes less than 4°, to 80% for slopes in the range 4 to 12°, and to 50% for slopes greater than 12°.

Zhang et al. (2010) also carried out a meta-analysis of the efficacy of vegetated buffer strips, using data from 73 studies. Fitted exponential regression models calculated efficacy for sediment to increase from 53 to 90% as buffer strip width increased from 2 to 12 m, and from 25 to 78% for total nitrogen, and from 24 to 76% for total phosphorus.

Donnison et al. (2013) conducted a review of 718 papers relating to the effectiveness of agri-environment measures on nitrate, sediment and phosphorus. They identified 225 studies relating to buffer strips. 61% of these looked into the effectiveness of buffer strips for reducing nitrate (n=139). 72% of these indicated that buffer strips were effective (n=100), but this varied for different forms. 44% of the studies looked at effects on sediment (n=98). It was generally indicated that buffer strips are effective at reducing sediment (87% of studies, n=85). 42% of the studies looked at phosphorus (n=94). 65% of the studies indicated that buffers were successful in reducing at least one type of P (n=61), but this did vary for different forms of P. Buffer strips seemed to be more effective at reducing total P than orthophosphate or soluble P.

Representation

The action allows for these grass margins to be grazed (which could result in pollution issues), but it was assumed that they were mown or mulched.

Given the uncertainty in the impacts of grass margins, the 3 available widths of grass margin were treated as comparable when considering their buffering impacts – all margins were assumed to reduce losses of nitrate and dissolved phosphorus in surface runoff by 10% and 25% respectively. Losses of particulate phosphorus and sediment in surface runoff were reduced by 50%. Note that these effectiveness values assume that the grass margins have been placed along a downslope field edge – this may not always be the case and so there may be an overestimation of the impacts of this action.

Grass margin width was accounted for when determining the arable land taken out of production, which would typically be about 1-2% of a field (assuming a 4m width margin along one side of a field of 200 – 400m length). Soil and fertiliser losses from this small part of the field were assumed to be completely negated.

The buffering aspect of this action was applied to the proportion of arable fields containing a grass margin. The land take aspect was applied to the proportion of the arable land taken out of production. The total length of the grass margins are shown in Table 4-6.

Table 4-6 Uptake of Arable Grass Margins

Action	Length (m)
Arable Grass Margins (3m)	112,867
Arable Grass Margins (4m)	65,116
Arable Grass Margins (6m)	179,403

4.2.9 Riparian Margins

Objective

Protect watercourses by creating linear buffer zones.

Definition of action

Livestock grazing in the riparian zone can lead to direct pollution of water with urine and faeces which can lead to pathogens entering the water. This can destroy aquatic habitats and lower the quality of water that could be used for human consumption. Riparian margins will stabilise riverbanks and intercept nutrients transported in overland flow.

Riparian margins require the establishment of a vegetated margin beside a watercourse, and the margin must be fenced off and inaccessible to livestock. The effect of this measure will be very similar to that of the protection of watercourses from bovines, but with an increased potential to reduce nutrient run off due to the presence of the vegetated margin which will act as a buffer strip. There will be direct effects on bank erosion, nutrient loss and sediment loss as well as the reduction of the direct deposition of livestock faeces into the watercourse.

Margins can be 3, 6, 10 or 30m wide and must be fenced off. Livestock are not permitted to graze the margin. The margin must be mowed or mulched once a year, but not between 1st March and 15th August. Fertilisers and pesticides are not permitted.

Evidence for effect

The evidence for this action is described under Protection of Watercourses from Bovines and Arable Grass Margins.

Representation

The impact of the fencing element of this action is assumed to be as per Protection of Watercourses from Bovines.

The buffer strips created by the Riparian Margin action are much wider than those from the Arable Grass Margin action, with the average Riparian Margin being 25 m wide (derived from Table 4-7). However, despite this greater width, these buffer strips are likely to be no more effective as their riparian location means the soils will be wetter so there is less potential for re-infiltration and the land occupied by the buffer strip would have had crop cover all year anyway rather than in the arable situation where the land may otherwise have been bare over winter or prior to crop establishment. Thus the representation of this action on losses of nutrients in surface runoff is the same as per the Arable Grass Margin.

Although no inorganic fertiliser can be applied to the grass margin, it is assumed that the total amount of fertiliser applied at field or farm scale is not altered to ensure that the same amount of forage is produced in order to support the existing amount of livestock. Because of this, and the comparative crop coverage of grassland and a grass margin, there is no impact of the land taken out of production, unlike for Arable Grass Margins.

The action was applied to the proportion of the grassland fields next to water with riparian margins. The total length of riparian margins is shown in Table 4-7. The action was implemented on 248 field parcels.

Table 4-7 Uptake of Riparian Margins

Action	Area (m)
Riparian Margins (3m)	4,960
Riparian Margins (6m)	1,967
Riparian Margins (10m)	6,140
Riparian Margins (30m)	48,705

4.2.10 Environmental Management of Fallow Land

Objective

To provide food and habitat for ground nesting birds other fauna and insects throughout the nesting season.

Definition of action

Fallow land in arable rotations has been a traditional feature across Europe for much of its agricultural history. However, changes in arable production during the latter half of the twentieth century along with technological improvements have led to reduced areas of fallow land. Fallow or set-aside land has multiple benefits for biodiversity including benefits for: breeding birds, wintering birds from crop stubbles and weed seeds; small mammal (and their predators) and insect and other invertebrates.

Participants must establish a fallow area through sowing a grass seed mix by the 31st May 2016. The area must be mowed or mulched at least once a year, but not between 1st March and 1st September, and off-takes are not allowed. Fertiliser or manure may not be applied and the area must be fenced off from livestock.

Evidence for effect

The IEEP (2008) was commissioned by DEFRA to produce a summary of the environmental benefits produced by set-aside in the EU, based on existing studies. Set-aside can help reduce pollution by nutrients in two ways. Through the reduced levels of inputs as levels of fertilizer are likely to be reduced on set-aside compared to conventional crops such as winter wheat and oil seed rape and through the actively buffering the land from surface runoff. A modelling study assessing nitrate leaching from different land use types found that long-term set-aside almost completely removed nitrate leaching on non-manured soils. It was estimated that the reversion of set-aside land to arable farming would increase N loss by 23-47kg ha^{-1} depending on soil type. It was also found that set-aside could reduce levels of total P loss and this would increase as set-aside became long-term. For one-year set aside the reduction in P loss was estimated at 15-20% increasing to 50% for long-term set-aside.

Newell-Price et al (2008) estimated that nitrate leaching losses were c.20 kg N ha^{-1} lower, when compared with winter cereal cropped land (typical leaching baseline loss of 40 kg N ha^{-1}) and that sediment and particulate phosphorus losses would be lower on under set-aside, particularly on sloping land or erosion-prone sandy and silty soils.

Representation

As no inorganic fertiliser is allowed, losses associated with nitrogen and phosphorus fertiliser were entirely removed.

Over-winter vegetation cover could be less than with a purposefully sown catch crop, but there are no establishment issues with the fallow land (as it is sown much earlier than a catch crop) and unlike catch crops in GLAS, the fallow land cannot be grazed. Losses of nitrogen from soil and crop residue mineralisation were reduced by 50% to account for reduced stimulation of mineralisation in the absence of cultivation and potentially some over-winter nitrogen uptake. Soil erosion (and associated particulate phosphorus) were assumed to be reduced by 25%.

Note that if the fallow land is ploughed out, there is likely to be an increase in nitrogen losses due to the mineralisation of the accumulated organic matter. This impact has not been considered.

The action was applied to the proportion of the arable land now managed as fallow land (Table 4-8).

Table 4-8 Uptake of Environmental Management of Fallow Land

Action	Area (ha)
Environmental Management of Fallow Land (part of parcel)	701
Environmental Management of Fallow Land (whole of parcel)	638

4.2.11 Farmland Habitat (Private Natura)

Objective

To avoid farming practices that cause environmental damage and protect vulnerable habitats such as wetlands, which in turn helps to safeguard animals and plants which occupy them.

Definition of action

Fallow or set-aside land has multiple benefits for biodiversity including benefits for; breeding birds, wintering birds from crop stubbles and weed seeds; small mammals (and their predators), insects and other invertebrates.

This measure involves producing a general sustainable management plan for any parcels which are part of a Natura site on the farm. This must address how the parcel is normally farmed (tillage/arable), where and when the parcel is grazed and by what livestock type, and set stocking levels that will avoid eutrophication, over grazing, under grazing and soil erosion.

Evidence for effect

For Natura sites on grassland, this action is assumed to be comparable to the Low Input Permanent Pasture, as low fertiliser rates would be required to avoid eutrophication and allow a moderate grazing rate. On arable land, it is assumed to be comparable to Fallow Land, as this more closely aligns to the objectives then minor modification of productive arable farming.

Representation

As per Low Input Permanent Pasture, the GLAS Attitudinal Survey (Cao, 2018) found that only a small proportion of farmers implementing this action (18% of 74 respondents) changed fertiliser use and even fewer (11%) changed stock densities due to this action.

Impacts were assumed to be identical to the Low Input Permanent Pasture and Environmental Management of Fallow Land actions.

The action was applied to the proportion of the grassland or arable land where the action was implemented (Table 4-9).

Table 4-9 Uptake of Farmland Habitat

Action	Land Use	Area (ha)
Farmland Habitat (private natura part of parcel)	Grass	2,655
Farmland Habitat (private natura whole of parcel)	Grass	86,993
Farmland Habitat (private natura part of parcel)	Arable	26
Farmland Habitat (private natura whole of parcel)	Arable	484

4.2.12 Low Input Permanent Pasture

Objective

To promote a grassland management system that through appropriate grazing levels and restriction on fertiliser and pesticide use results in a more diverse sward with an increase in flora and fauna.

Definition of action

Permanent pastures extensively grazed and managed with low inputs sustain a greater variety of plants and wildlife. The predominant impacts will be on nutrient leaching, with lower inputs, less nutrients will be available to be leached. Reduced stocking rates can potentially lead to a reduction in poaching, and hence reduce soil erosion.

The sward must be maintained by grazing and cannot be cut for hay or silage, and cannot be topped between the 15th March and the 15th July. The maximum chemical nitrogen that can be applied is 40 kg N ha^{-1} per year. There is no guidance on the use of organic fertilisers under this action.

Evidence for effect

Due to lower stocking rates and nitrogen fertiliser rates, nitrate losses from beef systems are generally expected to be lower than those from dairy systems (Ryan, 2002). A two year monitoring of nitrate leaching from steer grazing under different fertiliser N rates (57 kg N ha^{-1} and 216 kg N ha^{-1}) yielded leaching concentrations of 0.5 mg N l^{-1} and 3.9 mg N l^{-1} respectively (results averaged over the two year period of the experiment).

Richards et al. (2015) looked into the effect of REPS on nitrate leaching from a beef farming system in Ireland. They looked at an intensive farming system (stocking rate: 1.8 and 1.4 for bull and steer respectively; 2 silage harvests) and a representative REPS farming system (stocking rate: 1.4 and 1.1; 1 silage harvest). The REPS grazing treatments had significantly lower mean annual soil solution nitrate concentrations than the respective intensive treatments. Due to the reduced fertiliser inputs and stocking rates, nitrate leached was also significantly lower in the REPS systems. On average, over the three years of the study, 63.1kg N/ha was leached from the intensive plots compared to 17.3kg N/ha from the REPS plots. In addition, mean N inputs and surpluses were significantly lower in the REPS treatments than the intensive ones.

McCarthy et al. (2014) did not find any impact of different stocking levels on total nitrogen loss from grazing land on a free draining dairy farm in Ireland, over three seasons. However, in this study, there was no change in fertiliser application between the different stocking rate treatments.

The mean nitrogen surplus on a sample of 21 beef farms across Ireland (studied over a three year period 2009-2011) was 175 kg/ha with a mean nitrogen use efficiency of 0.28 (Mihailescu et al. 2014a). This nitrogen surplus was lower than the overall mean surplus from six studies of northern/continental dairy farms (224 kg N/ha), showing that Irish farms tend to be lower input than other comparable farms across Europe. Mihailescu et al (2014b) also looked into Phosphorus balances and use efficiency on the same 21 farms. They found that the mean phosphorus surplus across all of the farms was 5.09kg/ha and the mean PUE was 0.7. It was noted that since the introduction of the Good Agricultural Practice (GAP)

regulations in 2006, P surplus had decreased by 74% and PUE increased by 48%, mainly due to reduced use of inorganic P fertiliser and improvements in P management, such as improved timing of manure applications (Mihailescu et al. 2014b).

Treacy et al. (2008) looked into the changes in farm gate nitrogen balances on the same 21 intensive dairy farms discussed above over a period of 4 years (2003-2006). The mean annual farm gate N surplus declined from 277 to 232 kg/ha, mainly due to a decline in the use of additional inorganic fertilisers and imported organic manures. The mean NUE efficiency tended to increase over this time period, from 0.18 in 2003 to 0.2 in 2006. Ruane et al. (2014) looked into farm gate phosphorus balances on the same farms over the same time period. The mean annual P balance per farm was 9.4 kg/ha (ranging from -3 to 47kg/ha) and the mean PUE was 0.71 (ranging from 0.24 to 1.37). There was also found to be a relationship between the farm P balance and soil P status, with farms having a deficit of P tending to have a sub optimal soil P level and vice versa.

Mondelaers et al. (2009) conducted a meta-analysis of the difference in environmental impacts between organic and conventional farming, where the organic farms would have (by definition) zero inorganic fertiliser use and likely lower stocking densities and thus could be used as a proxy for the impacts of Low Input Pasture. Based on 12 nitrate leaching studies, they found that the weighted average leaching of nitrate is 8.9 kg ha^{-1} for organic farming and 20.9 kg ha^{-1} for conventional farming.

Representation

The fertiliser application data from the GLAS NMPs was used to determine the proportion of the farmed area, by farm type, which was over the 40 kg N ha^{-1} fertiliser limit. The proportion was 80% for Specialist Dairy farms, 60% for Specialist Beef farms and only 2% for Specialist Sheep. Given that dairy farms are less likely to be in GLAS, this was deemed comparable to the results of the GLAS Attitudinal Survey (Cao, 2018), which found that only 34% of farmers made changes to fertiliser use as a result of this action (as they may already have been under the maximum limit before joining GLAS, and this action mainly prevents intensification in the future for sensitive areas). This analysis assumes that farm level fertiliser usage is indicative of field level usage, whereas in reality there will be fields on most farms which are less productive and receive less fertiliser and thus may be more likely to have been chosen for this action. The representation chosen is thus likely to be an optimistic assessment of the impact.

Using the fertiliser application data from the GLAS NMPs, Table 4-10 shows the average N fertiliser rate on the farmed area which is receiving over 40 kg ha^{-1}, and inferred from Teagasc fertiliser survey data (Figure 4-2), the stocking rate for that land. The Teagasc data could also be used to identify the stocking rate when 40 kg ha^{-1} of fertiliser is applied. The data in Table 4-10 was used to define the reductions in fertiliser use and stock numbers associated with the Low Input Permanent Pasture action. Two scalars were applied to this data to reflect the results of the attitudinal survey: one to account for the fact that not all farms changed their fertiliser use (80%, 60% or 2%, according to farm type), and one to account for the fact that not all farms changed their stock density. The second factor was set to 40%, 30% and 1% by farm type as the attitudinal survey reported half as many farmers making a change to stock density as those that did fertiliser use (changing stock density may not have been required if sufficient grazing or silage was available from other fields.

The action was applied to the proportion of the grassland where the action was implemented (Table 4-11).

Table 4-10 Fertiliser and stocking rate data derived from GLAS nutrient management plans and fertiliser survey data (Dillon *et al.*, 2018) for determining the impacts of restricting fertiliser inputs to 40 kg ha^{-1}.

Farm Type	Farmed area receiving > 40 kg N ha^{-1} (%)	Average fertiliser rate on that area (kg ha^{-1})	Stocking rate at that fertiliser rate (kg ha^{-1})	Stocking rate at 40 kg ha^{-1} fertiliser rate (kg ha^{-1})
Specialist Dairy	80	144	160	60
Specialist Beef	59	95	160	100
Specialist Sheep	2	84	180	100

Table 4-11 Uptake of Low Input Permanent Pasture

Action	Area (ha)
Low-input Permanent Pasture (part of parcel)	61,660
Low-input Permanent Pasture (whole of parcel)	173,724

4.2.13 Farmland bird actions

Objective

There are a number of different farmland bird actions, but the objectives are mainly to maintain and increase areas for breeding, shelter and foraging for the different target species.

Definition of action

The actions typically involve creation of suitable grass swards, with constraints on grazing and cutting, which will have limited impacts on pollutant emissions. The major impact on nutrient emissions will be associated with limits to fertiliser application rates, which are:

- Breeding Waders: 0 kg ha^{-1}
- Chough: 40 kg ha^{-1}
- Corncrake: 30 kg ha^{-1}
- Hen Harrier: 40 kg ha^{-1}
- Twite A: 35 kg ha^{-1}

Some of the bird actions create habitats in different ways, with the primary impact on diffuse pollution being:

- Geese and Swans: no relevant constraints (grassland), use of catch crops (arable)
- Grey Partridge: creation of 12 m grass margin
- Twite C: creation of fallow land

Evidence for effect

Evidence for the effects of these actions are described under the actions to which they are comparable (Low Input Permanent Pasture, Catch Crops, Riparian / Arable Grass Margins and Fallow Land).

Representation

For the grass sward actions, the Hen Harrier action is the most common, accounting for 75% of the area under these actions. The weighted fertiliser constraint is close to 40 kg ha^{-1} and so this action is considered identical to Low Input Permanent Pasture. As per Low Input Pasture, the GLAS Attitudinal Survey (Cao, 2018) found that only some farmers (21% of 245 respondents) changed their fertiliser use to meet the requirements of these actions and only half as many again changed stocking densities.

The Geese and Swan action on arable land was represented in the same way as the Catch Crop action.

Table 4-12 Uptake of Farmland Bird Actions

Action	Land Use	Area (ha)
Breeding Waders (part of parcel)	Grass	90
Breeding Waders (whole of parcel)	Grass	935
Chough (part of parcel)	Grass	521
Chough (whole of parcel)	Grass	9,464
Corncrake (part of parcel)	Grass	6
Corncrake (whole of parcel)	Grass	125
Hen Harrier (part of parcel)	Grass	9,798
Hen Harrier (whole of parcel)	Grass	28,208
Twite (option A part of parcel)	Grass	78
Twite (option A whole of parcel)	Grass	3,004
Geese and Swans (part of parcel)	Grass	712
Geese and Swans (whole of parcel)	Grass	10,593
Geese and Swans (part of parcel)	Arable	2
Geese and Swans (whole of parcel)	Arable	957
Twite (option C part of parcel)	Grass	2
Action		**Length (m)**
Grey Partridge	Grass	58,057
Grey Partridge	Arable	28,632

The Grey Partridge action was represented as either an Arable Grass Margin or a Riparian Margin depending upon the land use on which it was located.

The area of Twite C action was so small that it was ignored.

4.3 Calculation of uptake rates for GLAS actions

The GLAS agreement data included both a farm and field parcel identifier for each action in each agreement. By joining this dataset with the LPIS and agricultural census datasets, every instance of an action was assigned to both a farm type and a WFD waterbody. Where the parcel identifiers in the GLAS agreement data did not match with any parcel identifiers in the LPIS dataset (which was over 50% for some actions), but could be matched to a farm, actions were located based upon the waterbody that the majority of the farm was in. The area of an action for a waterbody-farm type combination could then be divided by the total area relevant to that action (e.g. all the grassland area for Farmland Habitat on grassland parcels) to calculate a percentage uptake rate for that waterbody-farm type combination for subsequent use in the modelling framework.

For some GLAS agreements, neither farm nor parcel identifiers matched those in the LPIS or agricultural census data and so these could not be spatially located – to account for this and reflect the full extent of GLAS, uptake rates were scaled so that the total areas mapped agreed with the totals in the initial GLAS agreement data. For most actions, only a few percent could not be joined by either the parcel or farm, but the figure was 15% for Farmland Habitat.

Table 4-13 shows the overall uptake rates for the GLAS actions. For this table, the uptake rates are expressed as a percentage of the applicable area relevant to that method in the whole of Ireland (not just land managed by farms in GLAS). Approximately one third of the agricultural land in Ireland is managed by farms in GLAS (see Section 6.1). Uptake rates for most of the actions are relatively small, particularly when the small proportion of agricultural land occupied by arable cropping is taken into consideration. The actions with the greatest uptake are: Protection of Watercourses from Bovines (12% of grassland next to water); Catch Crops (10% of spring Cropping); Wild Bird Cover (7% of Spring Cropping) and Low Input Permanent Pasture (6% of grassland). Uptake rates of the other methods are generally 2% or less. Given the low uptake rates, only a few GLAS actions are likely to be important for reducing pollutant losses at national scale, although other actions may be more important locally. The calculated impacts of the actions on pollutant emissions are shown in Section 6.

Table 4-13 Overall uptake rates used in the modelling work, derived from data provided for tranches 1 and 2 of GLAS, expressed as a percentage of the applicable area in the whole of Ireland, calculated for those GLAS actions that were assumed to impact on sediment and nutrient losses

Category	Description	Applicability	Applicable Area ('000 ha)	Action Area[4] ('000 ha)	Uptake (%)
Habitat Actions	Arable Grass Margins (buffering)	Arable	370	3.1	0.8
	Arable Grass Margins (land out of production)	Arable	370	0.1	0.04
	Environmental Management of Fallow Land	Arable	370	1.3	0.4
	Farmland Habitat (Natura)	Arable	370	0.5	0.1
	Farmland Habitat (Natura)	Grassland	3,764	89.6	2.4
	Low-input Permanent Pasture	Grassland	3,764	235.4	6.3
Pollutant Abatement Actions	Protection of Watercourses from Bovines[1]	Grassland next to water	2,009	249	12.4
	Riparian Margins	Grassland next to water	2,009	0.9	0.05
	Catch Crops	Spring Crops	200	19.1	9.5
	Minimum Tillage	Arable	370	6.3	1.7
	Low-emission slurry spreading[5]	-	-	-	-
Landscape Actions	Planting New Hedgerows - (Suspended under Tranche 2 & 3)	Arable	370	3.2	0.9
	Planting New Hedgerows - (Suspended under Tranche 2 & 3)	Grassland	3,764	55.8	1.5
Species Actions	Breeding Waders[2]	Grassland	3,764	1.0	0.03
	Chough[2]	Grassland	3,764	10.0	0.3
	Corncrake[2]	Grassland	3,764	0.0	0.003
	Curlew[2]	Grassland	3,764	2.0	0.1
	Geese and Swans[3]	Spring Crops	200	1.0	0.5
	Grey Partridge	Arable	370	0.4	0.1
	Grey Partridge	Grassland	3,764	0.8	0.02
	Hen Harrier[2]	Grassland	3,764	36.0	1.0
	Twite A[2]	Grassland	3,764	3.0	0.1
	Twite C	Grassland	3,764	0.0	0.0
	Wild Bird Cover	Spring Crops	200	13.7	6.8

1 Includes the fencing aspect of Riparian Management
2 Represented as LIPP in modelling
3 Represented as Catch Crop in modelling

4 Some actions measured in length (e.g. protection of watercourses from bovines) have been converted to area of fields impacted, using the LPIS field area where each action was located
5 No evidence for changes in nutrient use, so no impacts calculated. Would have impacts on ammonia losses, but these were outside the scope of the modelling project.

5 Baseline Agricultural Pollutant Losses

This chapter shows the results of the first phase of the modelling work, which are the agricultural pollutant loads before the impacts of GLAS have been accounted for. The results include apportionment of pollutant loads, spatial mapping of loads and verification of model predictions against available measured datasets.

5.1 Baseline losses and source apportionment

National pollutant losses and footprints (losses expressed per hectare of agricultural land) are shown in Table 5-1. This shows that, for example, the total nitrate load from agriculture is 128 kT, which equals 29.3 kg ha^{-1}. The calculated loads are long term annual average losses delivered to water.

Agricultural pollutant footprints are summarised by farm type in Table 5-2. The definition of the pig farm did not include the land which received the manure generated, so it was not possible to calculate a footprint. The specialist dairy farm has the highest footprint for both nitrate, phosphorus nitrous oxide and methane, reflecting the high stocking densities on this farm type and the associated high use of fertilisers compared with other livestock farms. The lowest footprints for nitrate, phosphorus and nitrous oxide were on the specialist sheep farm and the mixed crops and livestock farm, which both have lower stocking densities and larger areas of rough grazing. The highest sediment losses are found on the specialist tillage farm – due to the larger area of arable land on these farms, there is more bare soil over winter when land is more susceptible to sediment loss.

Figure 5-1 summarises the proportions of the national pollutant emissions by representative farm type, along with the proportion of the agricultural area in Ireland occupied by each farm type. The specialist dairy and beef farm types contribute approximately 70% of the nitrate, phosphorus and nitrous oxide emissions and 80% of the methane emissions, reflecting the high pollutant footprint of the dairy farm and the larger number of specialist beef farms. Even though the specialist tillage farm has by far the highest footprint for sediment, it only contributes 20% of the total sediment load.

Table 5-3 shows the pollutant footprints of the different land uses, and here it can be seen that the results for nitrate, phosphorus and sediment are highest on arable land, lower on grassland and, lower still on rough grazing, which reflects the relative intensity of production on these land uses and for arable land its susceptibility to erosion and pollutant loss due to greater drainage and periods of bare soil. Methane emissions on the different land uses reflect differences in livestock density. For all pollutants except methane, the losses from non-field areas (i.e. manure storage, losses from yards and housing, excretion on tracks and in fords) is generally small. Apportionment of the total pollutant load by area (Figure 5-2), shows that grassland is the major source of losses for all pollutants except methane, reflecting the fact that grassland is the dominant agricultural land use. Methane emissions are mostly in proportion to the location of the livestock, and are thus high on non-field areas as they reflect the significant portion of the year that cattle spend away from the fields.

The modelling framework also allows apportionment of pollutant emissions by source type (Figure 5-3) and delivery pathway (Figure 5-4; for water borne pollutants only). For nitrate, the soil, fertiliser, manure and excreta sources are all relatively important (15% to 30% of

the total). Soil and excreta are the most important sources for phosphorus emissions (c. 40% each). The dominance of grassland (and associated livestock) in Irish agriculture makes the soil source less significant in areas where more arable land is found. The majority of nitrous oxide emissions come from either fertiliser (30%) or excreta at grazing (50%). For methane, the majority of emissions are enteric (83%). The majority of nitrate is lost through leaching to groundwater (75%), with surface runoff relatively unimportant (5%). For phosphorus, the contributions from surface runoff is greater (15%) but the main pathways are preferential flow (through drains) and direct excreta to water (i.e. livestock paddling whilst grazing or traveling to the yard). Preferential flow is the dominant pathway for sediment transport (68%) with the remainder transport through surface runoff and no losses due to leaching. The importance of the preferential pathway for phosphorus and sediment means that emissions are concentrated in areas where field drains have been installed.

The results from the modelling framework can also be apportioned in other ways, but these are typically important for only one pollutant. For nitrous oxide, 13% occurred indirectly as a result of nitrate leaching and 2% occurred due to volatilisation and deposition of ammonia. Nitrate emissions were explicitly separated into short term or incidental emissions (16%), emissions that occurred during the winter following manure or fertiliser application and from the mineralisation of crop residues (52%), and emissions that occurred over many years due to the effect of livestock excretion and repeated manure applications on the build-up and mineralisation of the soil organic nitrogen supply (32%).

Table 5-4 and Table 5-5 show a detailed breakdown of the source and pathway apportionment by land use for each pollutant (combining Figure 5-2, Figure 5-3 and Figure 5-4). These tables show, for example, how nitrate loss on arable land is dominantly from the soil source (62%), but for grassland, fertiliser (20%) and slurry (28%), and excreta at grazing (31%) are all more important than the soil contribution (12%). The greater proportion of arable land that is drained compared to grassland is shown by phosphorus and sediment emissions in preferential flow contributing over 80% of the total on arable land, but just under 50% on grassland.

The spatial variations in pollutant losses are shown in Figure 5-5 and Figure 5-6. Pollutant losses are generally highest in southern Ireland, where dairying is the dominant agriculture and so land is managed more intensively. Sediment losses are high in western areas where rainfall is higher and in eastern areas where there is more arable land.

Table 5-1 Baseline pollutant loads and pollutant footprints (load per hectare of agricultural land) for the whole of Ireland

	N	P	Z	N_2O	CH_4
Load (kT)	128.1	2.43	672	36.0	526.7
Footprint (kg ha^{-1})	29.3	0.56	154	8.3	120.7

Table 5-2 Baseline pollutant emission footprints (expressed per hectare of agricultural land) for the different farm types, summarised for the whole of Ireland. Note that the pig farm did not include and land, so no footprint is given.

	N (kg ha^{-1})	P (kg ha^{-1})	Z (kg ha^{-1})	N_2O (kg ha^{-1})	CH_4 (kg ha^{-1})
Mixed Crops	27.4	0.50	288	6.0	67.6
Mixed Crops & Livestock	10.1	0.25	153	2.4	-
Mixed Grazing Livestock	44.4	0.60	127	8.7	117.0
Specialist Beef	20.6	0.43	121	7.1	103.0
Specialist Dairy	50.1	0.96	139	14.6	251.9
Specialist Sheep	11.0	0.32	137	3.7	33.6
Specialist Tillage	26.7	0.51	406	5.0	0.0
Pig	-	-	-	-	-

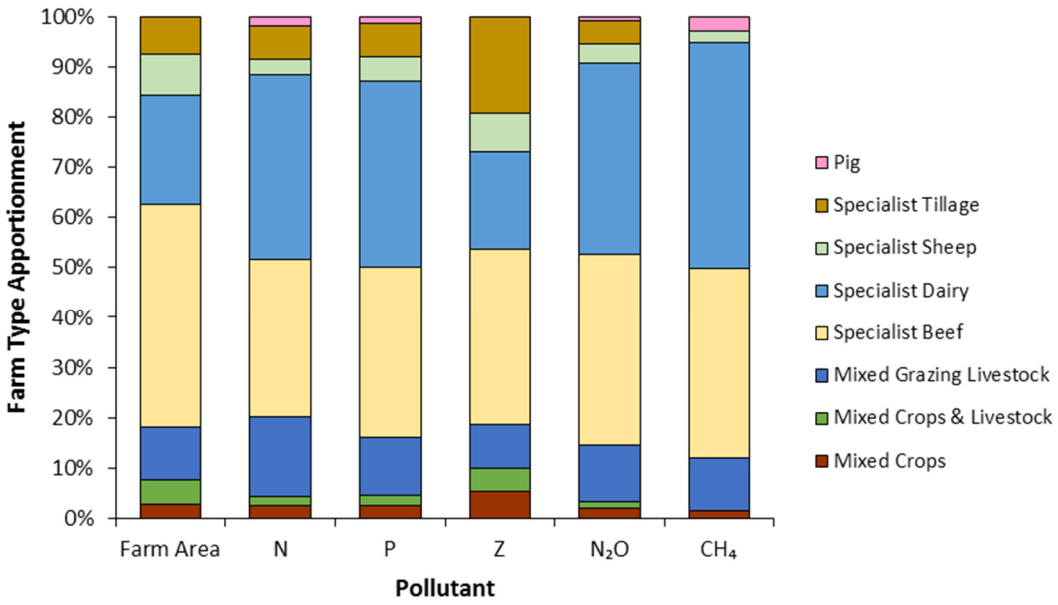


Figure 5-1 Apportionment of national agricultural pollutant losses by farm type.

Table 5-3 Baseline pollutant emission footprints for the different source areas, summarised for the whole of Ireland. Losses from 'Other' areas (steadings, fords, tracks and manure storage) are expressed per ha of all agricultural land.

	N (kg ha^{-1})	P (kg ha^{-1})	Z (kg ha^{-1})	N_2O (kg ha^{-1})	CH_4 (kg ha^{-1})
Arable	33.3	0.61	496	6.3	0.0
Grass	29.3	0.48	127	8.4	69.4
Rough	7.5	0.08	45	1.2	18.4
Other	0.8	0.09	-	0.4	59.6

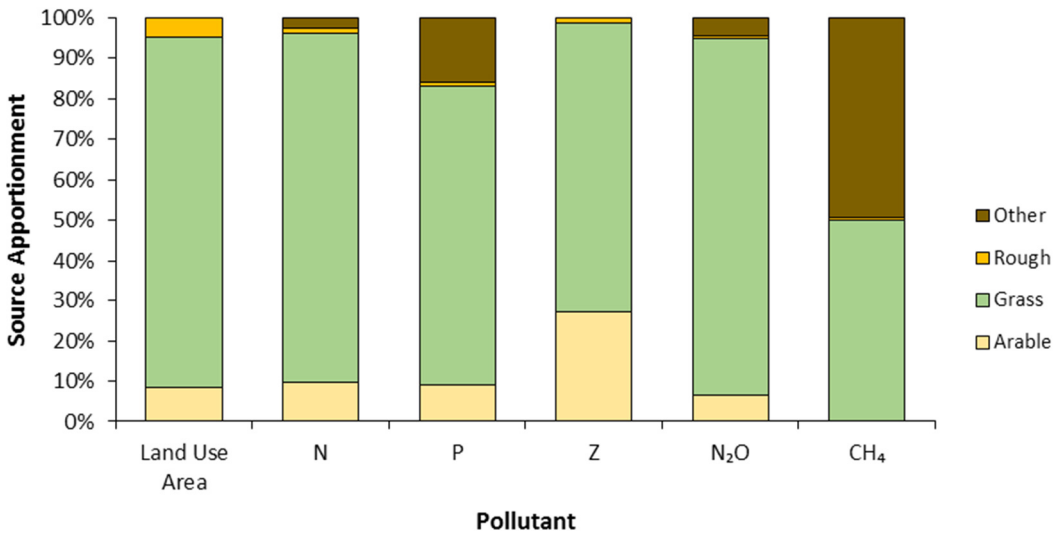


Figure 5-2 Apportionment of national agricultural pollutant losses by source area.

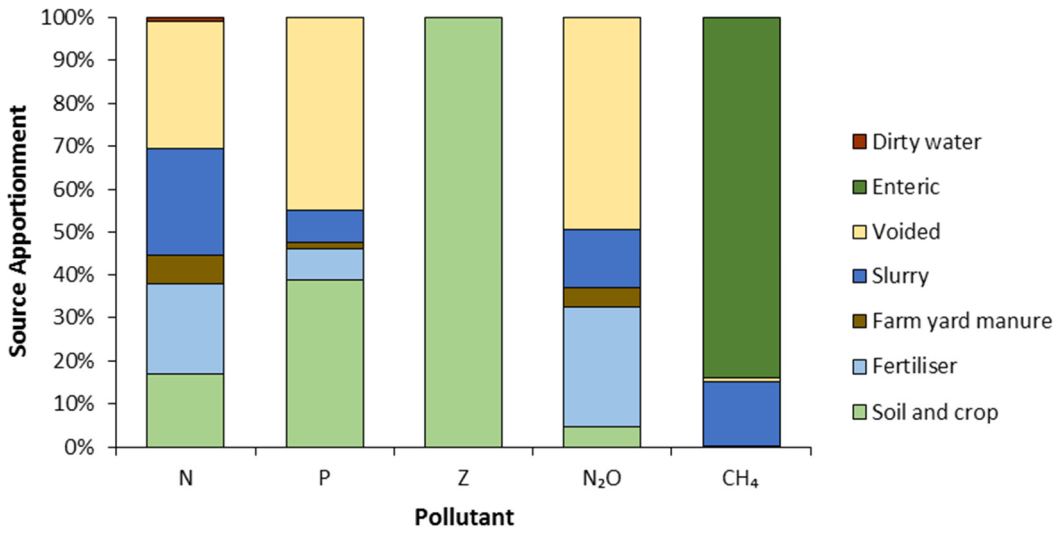


Figure 5-3 Apportionment of national agricultural pollutant losses by source type.

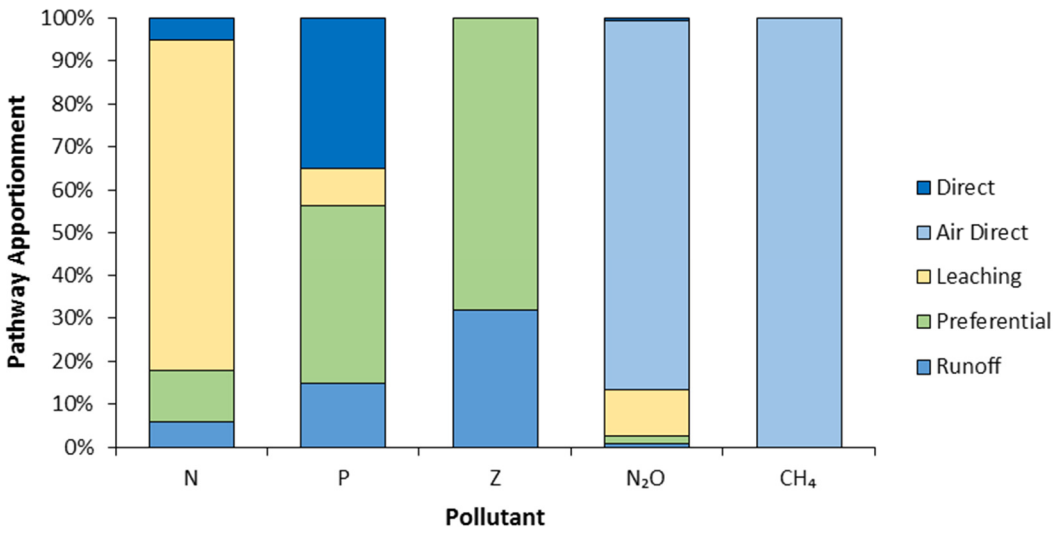


Figure 5-4 Apportionment of national agricultural pollutant losses by pathway.

Table 5-4 Source and pathway apportionment for agricultural emissions, summarised for all of Ireland, for nitrate, phosphorus and sediment.

Nitrate	Total Loss (kg ha^{-1})	Source Apportionment (%)						Pathway Apportionment (%)				
		Soil	Fertiliser	Manure	Slurry	Voided	Enteric	Surface	Preferential	Leaching	Direct	Air
Arable	33.3	62	31	1	6	-	-	2	21	77	-	-
Grass	29.3	12	20	8	28	31	-	6	12	79	3	-
Rough	7.5	66	-	-	-	34	-	10	-	90	-	-
Yards & housing	0.0	-	-	-	-	100	-	100	-	-	-	-
Tracks & fords	0.7	-	-	-	-	100	-	6	-	-	94	-
Manure storage	0.0	-	-	100	-	-	-	100	-	-	-	-

Phosphorus	Total Loss (kg ha^{-1})	Source Apportionment (%)						Pathway Apportionment (%)				
		Soil	Fertiliser	Manure	Slurry	Voided	Enteric	Surface	Preferential	Leaching	Direct	Air
Arable	0.61	93	6	-	-	-	-	12	81	7	-	-
Grass	0.48	40	9	2	11	39	-	17	46	10	27	-
Rough	0.08	83	-	-	-	17	-	68	-	32	-	-
Yards & housing	0.00	-	-	-	-	100	-	100	-	-	-	-
Tracks & fords	0.09	-	-	-	-	100	-	2	-	-	98	-
Manure storage	0.00	-	-	100	-	-	-	100	-	-	-	-

Sediment	Total Loss (kg ha^{-1})	Source Apportionment (%)						Pathway Apportionment (%)				
		Soil	Fertiliser	Manure	Slurry	Voided	Enteric	Surface	Preferential	Leaching	Direct	Air
Arable	496	100	-	-	-	-	-	18	82	-	-	-
Grass	127	100	-	-	-	-	-	36	64	-	-	-
Rough	45	100	-	-	-	-	-	100	-	-	-	-
Yards & housing	-	-	-	-	-	-	-	-	-	-	-	-
Tracks & fords	-	-	-	-	-	-	-	-	-	-	-	-
Manure storage	-	-	-	-	-	-	-	-	-	-	-	-

Table 5-5 Source and pathway apportionment for agricultural emissions, summarised for all of Ireland, for Nitrous Oxide and Methane.

Nitrous Oxide	Total Loss (kg ha^{-1})	Source Apportionment (%)						Pathway Apportionment (%)				
		Soil	Fertiliser	Manure	Slurry	Voided	Enteric	Surface	Preferential	Leaching	Direct	Air
Arable	6.3	47	51	-	2	-	-	-	2	16	-	79
Grass	8.4	-	29	3	12	56	-	1	4	11	-	86
Rough	1.2	-	-	-	-	100	-	2	-	22	-	76
Yards & housing	0.1	-	-	-	-	100	-	-	-	-	-	100
Tracks & fords	0.1	-	-	-	-	100	-	2	-	-	35	63
Manure storage	0.2	-	-	62	38	-	-	1	-	-	-	99

Methane	Total Loss (kg ha^{-1})	Source Apportionment (%)						Pathway Apportionment (%)				
		Soil	Fertiliser	Manure	Slurry	Voided	Enteric	Surface	Preferential	Leaching	Direct	Air
Arable	0.0	-	-	8	92	-	-	-	-	-	-	100
Grass	69.4	-	-	-	-	2	98	-	-	-	-	100
Rough	18.4	-	-	-	-	3	97	-	-	-	-	100
Yards & housing	39.6	-	-	-	-	-	100	-	-	-	-	100
Tracks & fords	1.8	-	-	-	-	1	99	-	-	-	-	100
Manure storage	18.3	-	-	-	100	-	-	-	-	-	-	100

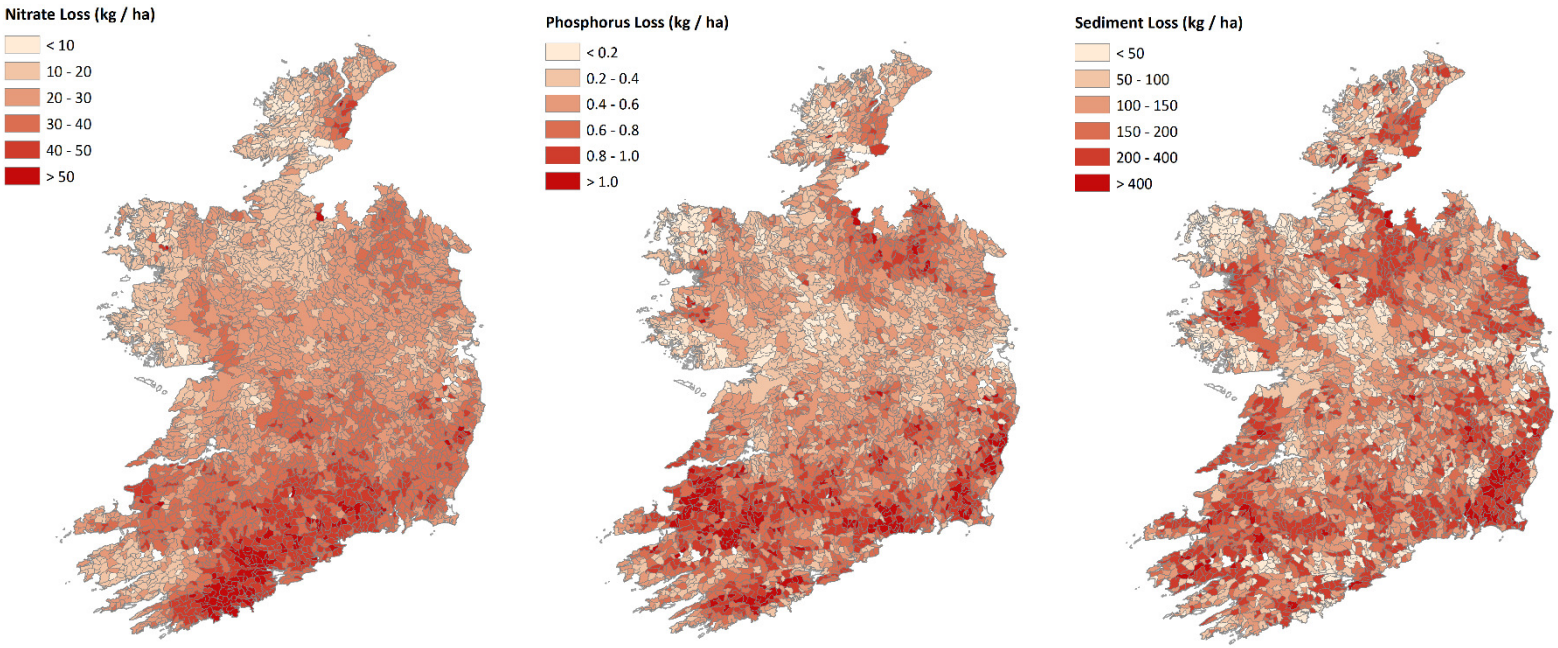


Figure 5-5 Annual average agricultural pollutant losses of nitrate, phosphorus and sediment for each WFD waterbody, expressed per hectare of agricultural land.

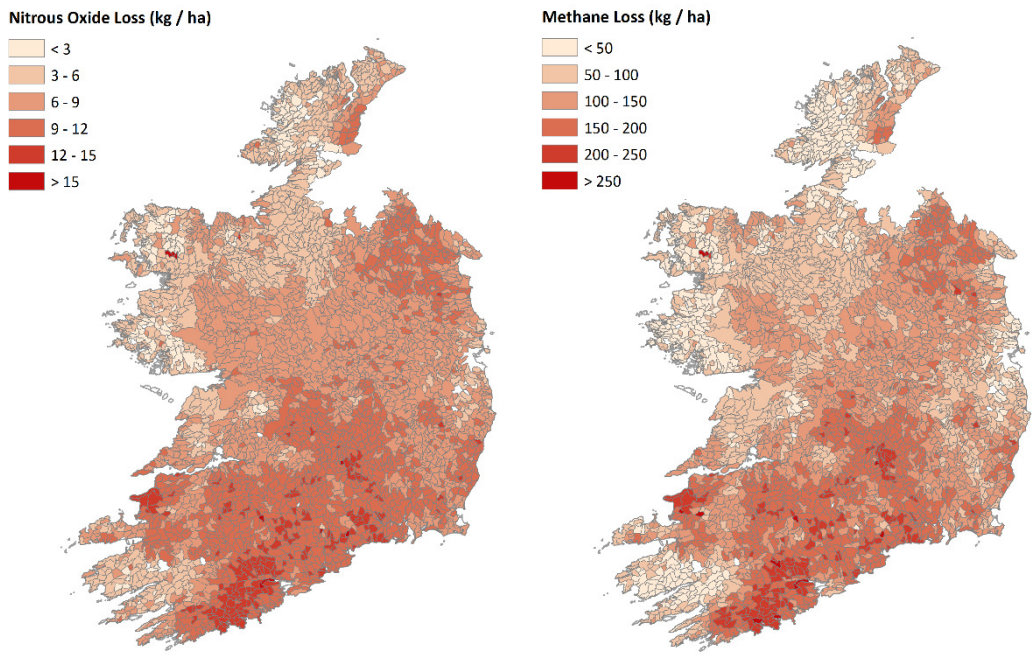


Figure 5-6 Annual average agricultural pollutant losses of nitrous oxide and methane for each WFD waterbody, expressed per hectare of agricultural land.

5.2 Verification of Pollutant Loads

For the 16 monitoring sites used for OSPAR reporting, it was possible to compare modelled predictions of nitrate and phosphorus loads with observed loads for the period 2011-2013 (O'Boyle *et al*., 2016). The OSPAR catchments vary in size between 128 km^2 (Tolka) and 11,115 km^2 (Shannon). The observed data includes contributions from non-agricultural sources, and will also include the impacts of in-channel retention. Non-agricultural losses of nitrate are typically less than 10% of the total load, but for phosphorus it was more significant, averaging around 50% (Ní Longphuirt *et al*., 2016). Figure 5-7 shows that there is a good agreement between the N loads (r^2 of 0.67), although the observed loads are lower due to in-river processes. The agreement between modelled and observed phosphorus loads is lower (r^2 of 0.42) - this can be explained by the greater variation in the contributions from non-agricultural sources and the potentially large impact of retention in the catchments with large lakes.

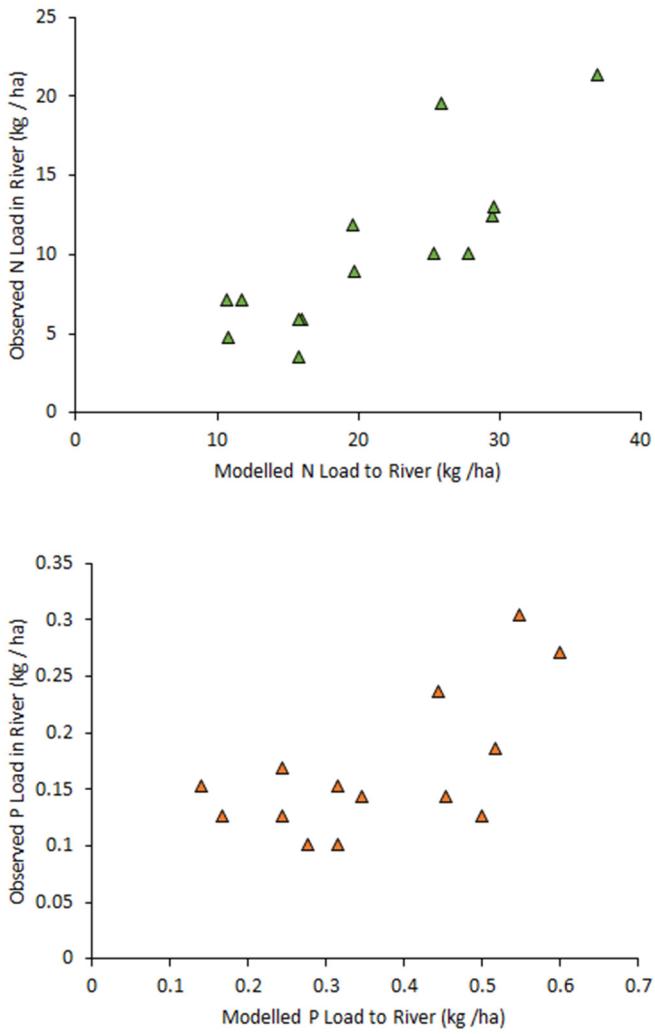


Figure 5-7 Comparison of modelled agricultural loads (predicted to river) with observed loads measured in rivers for 2011-2013 (O'Boyle *et al*., 2016). Observed loads were available for 16 OSPAR monitoring sites. Loads are expressed per hectare of all land within the catchments. The modelled loads do not include non-agricultural sources or the impacts of in-river processes.

McGuckin *et al*., (1999) calculated phosphorus export coefficients for different land cover types in Ireland, based upon fortnightly sampling of 30 small streams. They calculated export coefficients for managed grassland of 0.8 kg ha^{-1}, which are slightly greater than average load calculated in this project of 0.48 kg ha^{-1}, although there have been reductions in phosphorus fertiliser usage and soil P indices since their report. Lewis *et al*., (2013) measured annual P losses in 2002 for three nested grassland catchments in southern Ireland of 1.6, 2.5 and 2.6 kg ha^{-1}. Values of over 1 kg ha^{-1} were predicted in this project for some waterbodies in southern Ireland (Figure 5-5).

The Agricultural Catchments Programme has involved the monitoring of nutrient losses in four grassland and two arable catchments, representative of the different conditions across Ireland, since 2008 (Shortle and Jordan, 2017). The catchments range in size from 760 to 3,000 ha, typically with 90% of the land use agricultural. The modelling approach used in this project uses national survey data to provide nationally representative input data and so will not reflect the intricacies of management, and thus pollutant losses, within specific catchments. However, the range in the observed data for these catchments should be comparable to the range in the modelled outputs. Sherriff *et al*., (2015) reports suspended sediment loads in 5 of these catchments between 2009 and 2012. Annual losses ranged between 4 and 50 t km^{-2}, whilst the range in losses calculated in this project is between 0.2 and 70 t km^{-2} (of agricultural land). The low values in this project reflect upland catchments where losses would be expected to be lower than in the monitored grassland and arable catchments. Shore *et al*., (2017) reported annual average total phosphorus loads (2010 – 2014) of between 0.03 and 1.17 kg ha^{-1}, which are comparable to the losses presented in this project (Figure 5-5). The proportion of the observed total phosphorus load that was soluble varied between 38% and 65% for the different catchments, with the higher values found in the grassland catchments. The modelling framework predicted national average soluble fractions of 20% on arable land and 75% on grassland, which are slightly outside the observed ranges but they do not account for a mix of land uses within a catchment. Across all the waterbodies, the framework predicted soluble fractions of between 44 and 96%, with highest values typically in the more intensively stocked catchments. Mellander *et al*., (2014) reported annual nitrate losses in two catchment in the south and south east of Ireland of between 20 and 48 kg ha^{-1} between 2010 and 2012. The predicted annual average loss for the whole of the Ireland was 29 kg ha^{-1}, ranging from 10 kg ha-1 in the north of Ireland to over 50 kg ha-1 in the south.

6 Impacts of GLAS

This chapter shows the proportion of the agricultural land and calculated pollutant loads in Ireland on farms in GLAS and then the calculated impact of GLAS on these agricultural pollutants.

6.1 Coverage of GLAS

Figure 6-1 shows the proportion of all agricultural land managed by farms in GLAS, which can be used to estimate the total proportion of the national pollutant load that could in theory be controlled by GLAS, although it must be considered that GLAS options are not located on all land on these farms in GLAS, and not all options have any impact on diffuse pollution. Approximately one third of agricultural land is managed by farms in GLAS (Table 6-1), with the proportions roughly comparable for all farm types except specialist dairying which is noticeably lower at only 13% and specialist sheep farming which is higher at 47%. This explains the pattern in Figure 6-1, where uptake of GLAS is lowest in dairying areas such as the south.

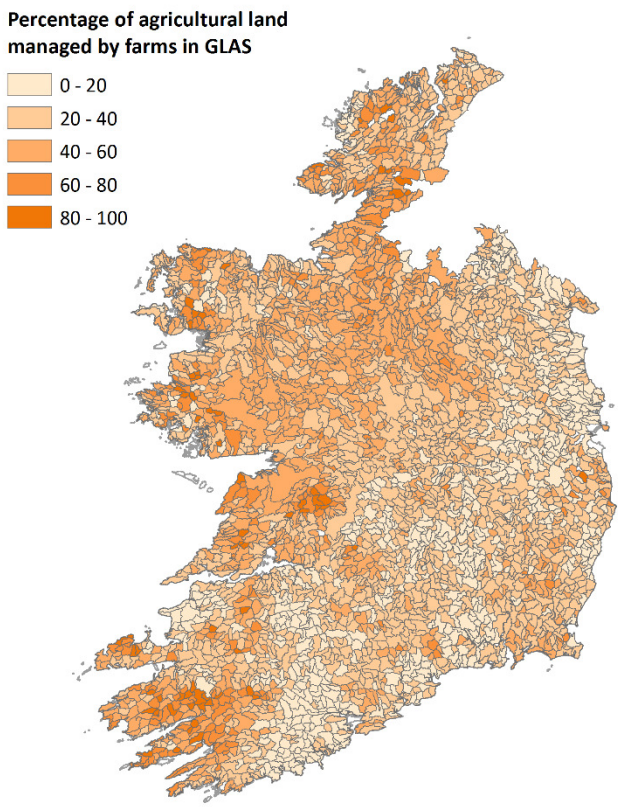


Figure 6-1 Percentage of all agricultural land managed by farms in GLAS, summarised by WFD waterbody.

Although approximately 32% of all agricultural land is managed by farms in GLAS, the percentage of the national pollutant load occurring from this land varies between 33% for sediment to 23% for methane (Table 6-2). The values are lower than the proportion of land (i.e. 32%) for most pollutants because dairy farms, which typically have the highest pollutant

footprints (see Table 5-2), are less likely to be in GLAS. By farm type, the proportion of the pollutant load from farms in GLAS is generally proportional to the area of that farm type in GLAS.

Table 6-1 Percentage of land for each farm type which is managed by farms in GLAS, summarised by land use.

	All Agricultural Land	Arable	Grass	Rough
Mixed Crops	23	27	23	25
Mixed Crops & Livestock	36	34	37	41
Mixed Grazing Livestock	37	28	36	50
Specialist Beef	39	35	39	46
Specialist Dairy	13	11	13	24
Specialist Sheep	47	39	47	47
Specialist Tillage	31	31	30	40
Pig	-	-	-	-
Total	**32**	**30**	**32**	**44**

Table 6-2 Percentage of the agricultural pollutant load for each farm type (and nationally), which is from farms in GLAS.

Farm Type	N	P	Z	N_2O	CH_4
Mixed Crops	24	24	24	23	-
Mixed Crops & Livestock	35	36	37	36	36
Mixed Grazing Livestock	34	34	37	34	33
Specialist Beef	38	38	40	38	37
Specialist Dairy	13	13	14	13	12
Specialist Sheep	46	48	48	45	44
Specialist Tillage	31	33	33	31	-
Pig	-	-	-	-	-
National Total	**27**	**28**	**33**	**27**	**25**

GLAS was designed as a targeted scheme, focussing on High Status and Vulnerable waterbodies (Figure 4-1). Table 6-3 shows the breakdown of the land manged by farms in GLAS by farm type, for the different targeted areas. The major difference between the areas is that in the High Status waterbodies there is a greater proportion of Specialist Sheep farms (28% of the land managed by farm in GLAS) than there is in other areas (9 – 13%). The extra land for Specialist Sheep farms comes at the expense of the Mixed Crops and Livestock, Specialist Beef and Specialist Tillage farm types, which are all more likely to be found outside of the High Status waterbodies.

Table 6-3 Percentage of the land managed by farms in GLAS, summarised by farm type, for catchments of different status as targeted by GLAS.

	High	Vulnerable	Neither	Unassigned
Mixed Crops	4.3	3.3	3.0	3.8
Mixed Crops & Livestock	0.6	4.1	2.5	3.3
Mixed Grazing Livestock	15.5	11.4	12.1	11.8
Specialist Beef	41.4	54.6	55.4	53.2
Specialist Dairy	8.9	8.3	9.1	9.0
Specialist Sheep	27.6	9.3	12.6	10.8
Specialist Tillage	1.7	9.1	5.3	8.1
Pig	-	-	-	-
Total	**100**	**100**	**100**	**100**

6.2 Impacts of GLAS

6.2.1 Overall impacts

The overall impact of GLAS, on the long term annual average pollutant loads from land in GLAS, are shown in Table 6-4. For the pollutants modelled (nitrate, phosphorus, sediment nitrous oxide and methane), the calculated reductions are 3-10%. There are significant differences by farm type, with larger (3-15%) reductions on Specialist Dairy farms and small reductions (<1%) on Specialist Sheep farms. The major causes of these reductions are the Low Input Permanent Pasture action (and the comparable Natura Habitat and Farmland Bird actions), which has the highest uptake rate of all the actions (Table 4-13) and which has been assumed to reduce fertiliser inputs and livestock (particularly on Specialist Dairy farms), thus controlling pollutant losses at source rather than trying to mitigate mobilisation or delivery and the Watercourse fencing action (preventing livestock excreting directly to water). Methane emissions only occur from livestock and none of the GLAS actions impact on emissions except where stock numbers are reduced, so the fact that methane emissions have been estimated to have been reduced by 3% effectively means there has been a 3% reduction in stock numbers. The overall impact on sediment is much higher on the farm types with more arable land (e.g. Specialist Tillage at 9%) where the Catch Crop and Wild Bird Cover actions have greater applicability as most of the actions relevant to grassland have a limited impact on sediment.

Table 6-4 Percentage reduction in the agricultural pollutant load from farms in GLAS, due to GLAS actions, by farm type and for all GLAS farms.

Farm Type	N	P	Z	N_2O	CH_4
Mixed Crops	9.7	4.2	6.4	12.1	-
Mixed Crops & Livestock	7.9	9.9	9.9	4.6	1.6
Mixed Grazing Livestock	4.7	8.4	2.6	4.8	3.1
Specialist Beef	5.8	10.5	1.4	5.1	3.2
Specialist Dairy	6.2	14.1	2.7	4.8	3.1
Specialist Sheep	0.4	0.5	0.8	0.4	0.2
Specialist Tillage	8.6	7.6	9.1	4.5	-
Pig	-	-	-	-	-
All	5.7	9.6	3.7	4.7	3.0

Table 6-5 shows the reductions in the agricultural pollutant loads when the impacts of GLAS are diluted by those farms not in GLAS. National reductions are reduced by approximately over two thirds to 2-3%, reflecting the proportion of the pollutant load arising from farms not in GLAS (Table 6-2). The two-thirds change is similar for most farm types, except for Specialist Dairy, where the reduction drops more as only 13% of land / pollutant loads are in GLAS.

Table 6-5 Percentage reduction in the agricultural pollutant load from all farms, due to GLAS actions, by farm type and nationally.

Farm Type	N	P	Z	N_2O	CH_4
Mixed Crops	2.3	1.0	1.6	2.8	-
Mixed Crops & Livestock	2.8	3.6	3.6	1.6	0.6
Mixed Grazing Livestock	1.6	2.9	1.0	1.6	1.0
Specialist Beef	2.2	4.0	0.6	1.9	1.2
Specialist Dairy	0.8	1.9	0.4	0.6	0.4
Specialist Sheep	0.2	0.2	0.4	0.2	0.1
Specialist Tillage	2.7	2.6	3.1	1.4	-
Pig	0.0	0.0	-	0.0	0.0
National	1.5	2.7	1.2	1.3	0.7

Figure 6-2 and Figure 6-3 show the local spatial variation in the impacts of GLAS. The data in these figures are reductions in the total load from all farms, as ultimately it is the impact at catchment scale that is important for water quality and WFD status. For nitrate, nitrous oxide and methane, reductions are greatest in the western catchments which correspond to areas of high GLAS uptake. Sediment impacts are greatest along the Eastern coast, where arable land and Specialist Tillage farms are more common. Phosphorus impacts are more

evenly distributed across the country, reflecting the importance of both livestock excreta and sediment-bound phosphorus as sources.

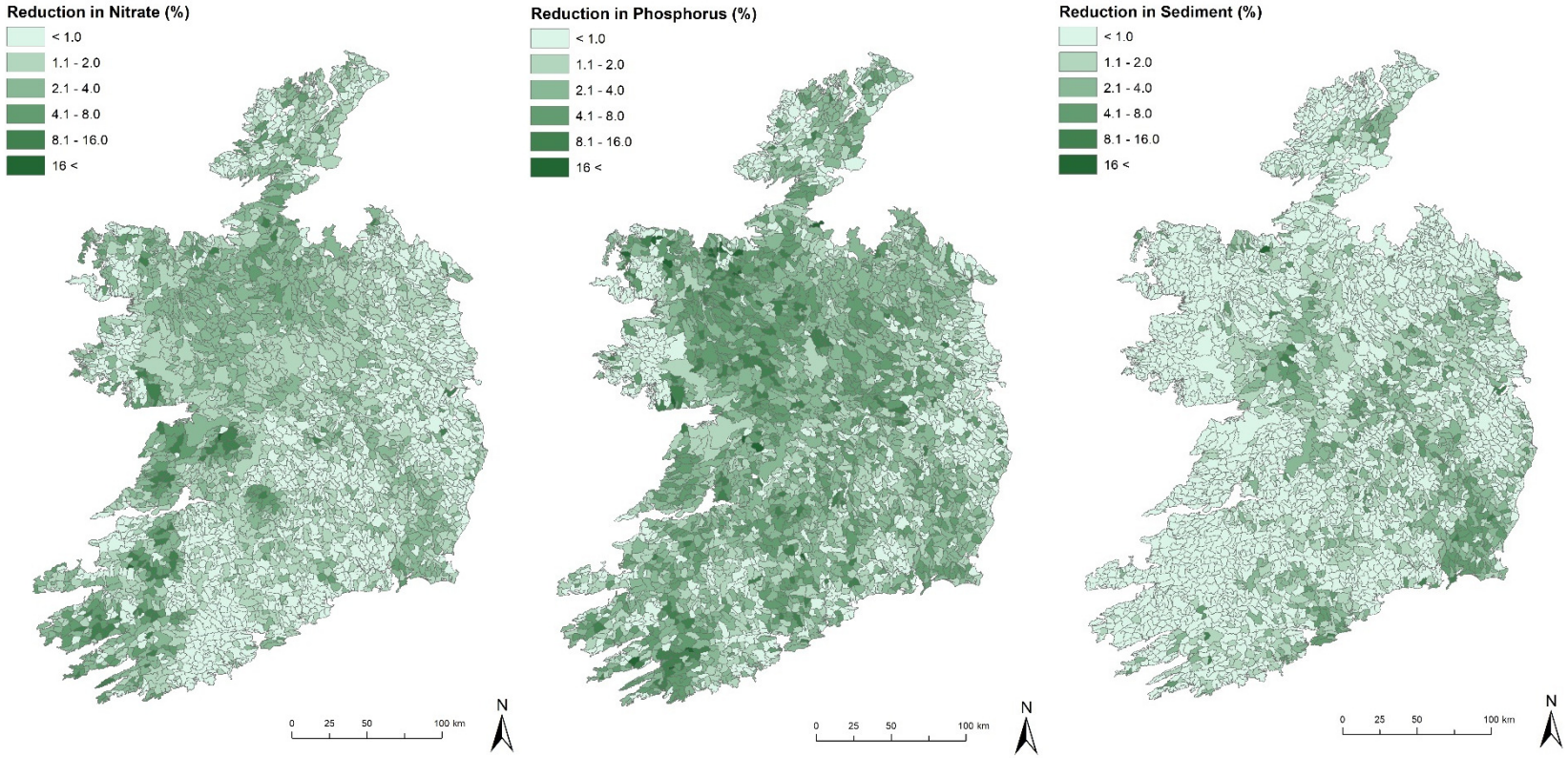


Figure 6-2 Reductions in annual average agricultural pollutant losses of nitrate, phosphorus and sediment due to GLAS. Results are expressed as a proportion of the load from GLAS and non-GLAS farms for each WFD waterbody.

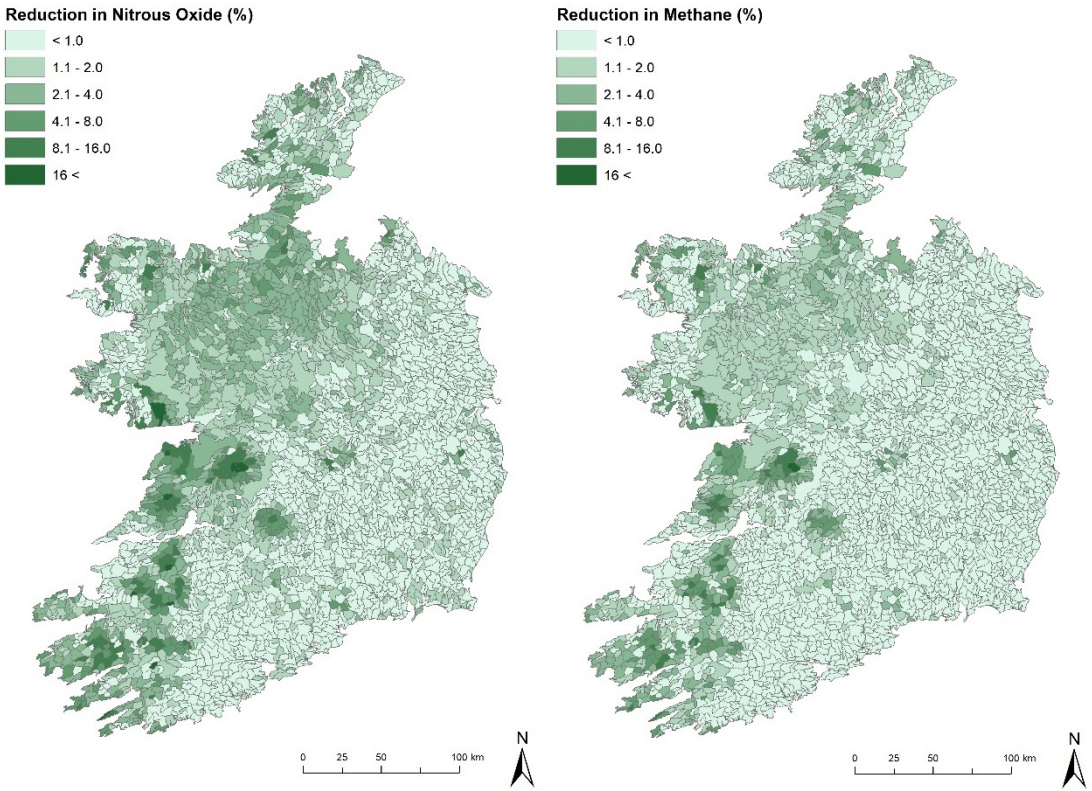


Figure 6-3 Reductions in annual average agricultural pollutant losses of nitrous oxide and methane due to GLAS. Results are expressed as a proportion of the load from GLAS and non-GLAS farms for each WFD waterbody.

Figure 6-4 shows the pollutant reductions plotted against the percentage of the agricultural land area. Reductions in phosphorus are more evenly distributed than the reductions in other pollutants. The highest reductions in nitrate, sediment, nitrous oxide and methane are focussed in approximately 15% of the land area.

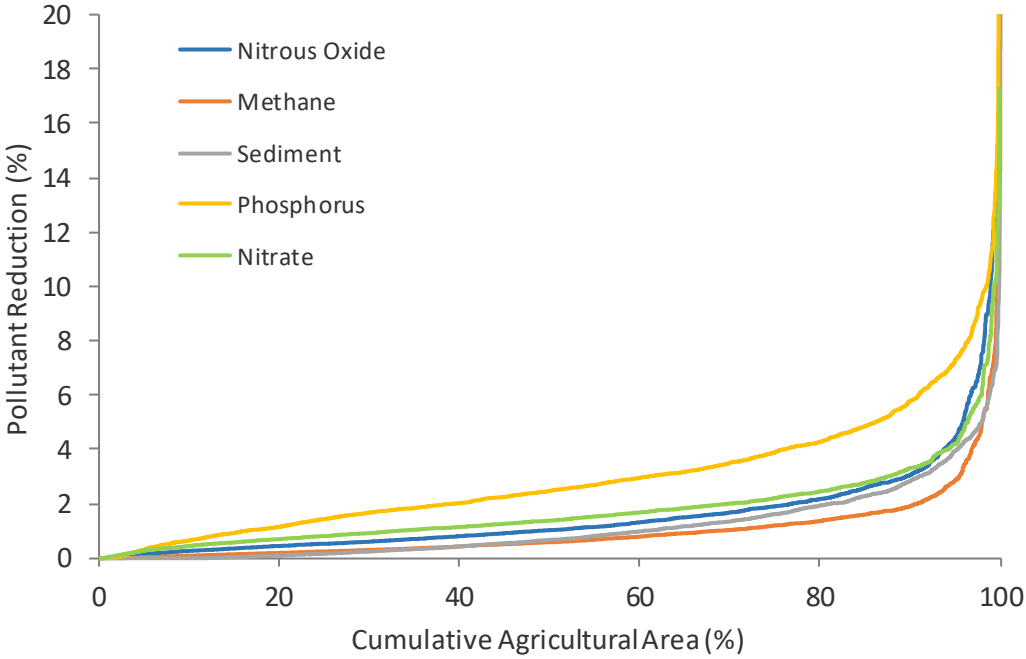


Figure 6-4 Reductions in agricultural pollutant loads due to GLAS plotted against cumulative agricultural area.

6.2.2 Impacts by waterbody targeting

Table 6-6 shows the reductions in pollutant loads from GLAS farms by the waterbody classification used in the Tier-based targeting in GLAS. For nitrate, nitrous oxide and methane, the reductions are all slightly highest in High Status waterbodies but lower in Vulnerable than other categories. GLAS was designed to target High Status waterbodies, but the greater prevalence of Specialist Sheep farms in these areas (Table 6-3) limits the potential for greater impacts – on sheep farms initial pollutant loads are lower and the impacts of the action which has been estimated as the most effective (Low Input Permanent Pasture) are minimal due to the low fertiliser use on these farms meaning that little change is necessary to comply with the action requirements. Sediment impacts are lowest in High Status waterbodies, reflecting the facts that the most effective GLAS actions to reduce sediment losses are mostly found on tillage land, which is less common in the High Status waterbodies.

At catchment scale, these pollutant reductions occurring on the GLAS farms will be diluted by the agricultural land (and associated pollutant loads) not in GLAS. Across all the waterbodies in Ireland within a classification, typically only 25-35% of the pollutant loads is

from GLAS farms (Table 6-7), with slightly higher values in the High-Status waterbodies. Thus the impacts achieved on the GLAS farms are reduced by roughly 65-75% at catchment scale (Table 6-8), down to the 1-3% figures quoted for the whole of Ireland.

Table 6-6 Percentage reduction in the agricultural pollutant load from farms in GLAS, due to GLAS actions, by WFD waterbody classification.

WFD Classification	N	P	Z	N_2O	CH_4
High	6.3	10.0	1.5	5.7	3.6
Vulnerable	5.5	10.3	4.4	4.2	2.7
Neither	6.0	9.8	3.2	5.2	3.3
Unassigned	5.5	8.5	4.2	4.6	2.8
Total	**5.7**	**9.6**	**3.7**	**4.7**	**3.0**

Table 6-7 Percentage of the agricultural pollutant load from farms in GLAS, by WFD waterbody classification.

WFD Classification	N	P	Z	N_2O	CH_4
High	30	34	41	31	27
Vulnerable	26	27	32	26	24
Neither	28	29	33	28	26
Unassigned	26	27	32	26	24
Total	**27**	**28**	**33**	**27**	**25**

Table 6-8 Percentage reduction in the agricultural pollutant load from all farms, due to GLAS actions, by WFD waterbody classification.

WFD Classification	N	P	Z	N_2O	CH_4
High	1.9	3.4	0.6	1.8	1.0
Vulnerable	1.4	2.8	1.4	1.1	0.6
Neither	1.6	2.8	1.1	1.5	0.9
Unassigned	1.4	2.3	1.4	1.2	0.7
Total	**1.5**	**2.7**	**1.2**	**1.3**	**0.7**

7 Conclusions

A number of key spatial environmental datasets have been created to enable agricultural pollutant modelling across the whole of Ireland. These datasets include monthly annual average climate variables, soil series and land cover. Data on soil series properties were also tabulated, and additional properties such as bulk density derived using pedo-transfer functions appropriate for Irish conditions.

In order to create the agricultural input data required for the pollutant models, representative farm systems have been created and populated with activity data (i.e. livestock, manure and fertiliser management data) for Ireland. Where possible, this activity data was derived from surveys which provided data by farm type. The holding level agricultural census data was used to determine the farm type for each holding, allowing for both the creation of crop and livestock statistics for each farm type and the creation of farm type crop and livestock numbers by WFD waterbody.

All these datasets were used to run a suite of agricultural pollutant models in order to produce annual average loads of nitrate, phosphorus, sediment, nitrous oxide and methane. The pollutant loads were produced at WFD waterbody scale, and the results could be disaggregated by farm type and the other coordinates of the source apportionment system (e.g. by flow pathway, or source area). The calculated pollutant loads are comparable in size to modelled and observed pollutant loads in the literature, and reflect the intrinsic risks associated with the underlying environmental data as well as the local agricultural pressures.

Applications for GLAS were invited over 3 tranches with almost 75% of current scheme participants (as of May 2019) drawn from tranches 1 and 2. GLAS agreement data was provided for the almost 40,000 farmers who joined the scheme under tranches 1 and 2 (which closed for entry at the end of 2015). Tranche 3 applications were not included in this analysis.

For the assessment of GLAS, the project firstly determined the following output and result indicators recognised by the CMEF:

- Areas of scheme participation, by farm type and by WFD waterbody
- Baseline pollutant loss from farms in scheme (and the proportion of WFD waterbody and national totals)

Nationally, the percentage of agricultural land managed by farmers in GLAS is 32%. The percentage of the national agricultural pollutant load from land managed by farmers in GLAS varies from 25% for methane to 33% for sediment. The fact that the percentage of the load is lower than the percentage of the area reflects the fact that less intensively managed farms are more likely to have joined GLAS.

The baseline losses are explicitly disaggregated by source, source area, method of mobilisation and delivery pathway allowing a transparent evaluation of the limits to pollution control under GLAS.

For each action within GLAS, spatial data on uptake was combined with evidence on how these actions attempt to control diffuse pollution and this was utilised within the modelling framework to assess the impacts of GLAS agreements on agricultural pollution. This has

allowed for the calculation of impact indicators demonstrating the levels of pollutant reduction that have occurred, both on land in scheme and at whole catchment / national level when diluted with the pollution occurring from farms not in scheme. This provides data which can be used to help answer the following Common Evaluation Questions under the Rural Development Programme (2014-2020):

- FA-4B – To what extent have the RDP interventions supported the improvement of water management, including fertiliser and pesticide management?
- FA-4C – To what extent have RDP interventions supported the prevention of soil erosion and improvement of soil management?
- FA-5D – To what extent have the RDP interventions contributed to reducing greenhouse gas and ammonia emissions from agriculture?

The actions with the greatest uptake are: Protection of Watercourses from Bovines (12% of grassland next to water); Catch Crops (10% of spring Cropping); Wild Bird Cover (7% of Spring Cropping) and Low Input Permanent Pasture (6% of grassland). Uptake rates of the other actions are generally 2% or less. Given the low uptake rates, only a few GLAS actions are likely to be important for reducing pollutant losses at national scale, although other actions may be more important locally.

Nationally, the estimated reductions in long term annual average pollutant loads from farms in GLAS are 10% for phosphorus, 6% for nitrate, 5% for nitrous oxide, 3% for methane and 4% for sediment. When diluted by the unaffected loads from non-scheme farms, which occupy over two thirds of the agricultural area, the national impacts are 3% or less. There are significant spatial variations in the reductions achieved, with reductions in some catchments estimated to be over 15%, even when the contributions from agricultural land not in GLAS are accounted for. These national scale impacts are broadly comparable to model-based estimates of national reductions in nitrate, phosphorus and sediment of ~2% achieved by the agri-environment schemes Tir Cynnal and Tir Gofal in Wales (Iwan-Jones et al., 2016) and 1-2% achieved by Environmental Stewardship, Countryside Stewardship and Catchment Sensitive Farming schemes in England (Elliot et al., 2019).

With the exception of sediment, which is mainly impacted by a couple of actions which change arable cropping practices, the reductions in pollutant losses are mainly a result of the Low Input Permanent Pasture action, which limits fertiliser use on the fields where the action is implemented and potentially impacts on stock numbers if less forage is available on farm. The contribution of this one action to the overall reduction occurs despite the calculations accounting for the results of the GLAS Attitudinal Survey (Cao, 2018), which stated that only 34% of farmers changed their fertiliser usage and only 15% changed stocking density due to this action (as they may already have been under the fertiliser restriction limit on those fields or have sufficient grazing land or silage to maintain stock numbers).

GLAS is a spatially targeted system, with Tier 1 designed to protect priority assets including High Status WFD waterbodies. Whilst the scheme may be successful in achieving this aim, the fact that Low Input Permanent Pasture is the single most effective action (both due to uptake and method of impact) results in reductions in pollutant loads not being much greater in High Status waterbodies than elsewhere as these areas typically already had lower fertiliser use and stocking rates before joining the scheme and so the potential impact of this action is lower.

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Annex 1 GLAS Action Summary

Action	Uptake	Expected Impact
Minimum Tillage	6,353 ha	Due to reduced stimulation of mineralisation, nitrate losses from soil organic matter are assumed to be reduced by 10%. Incidental losses of nitrate and dissolved phosphorus in surface runoff are reduced by 10%, whilst particulate phosphorus and sediment losses in surface runoff are reduced by 50%.
Catch Crops	19,131 ha	Assumed to reduce over-winter losses of nitrogen by 25% and soil erosion (and associated particulate phosphorus) by 25%. The values are lower than some literature evidence due to the proportion of catch crops used for grazing, resulting in reduced vegetation cover and compaction and the return of crop nitrogen uptake as excreta.
Wild Bird Cover	13,714 ha	Assumed to reduce over-winter losses of nitrogen by 25% and soil erosion (and associated particulate phosphorus) by 50%. There will be no cultivation in the autumn to stimulate mineralisation, but there may be limited over-winter nitrogen uptake as the crop should be fully established, so the nitrogen impact is assumed comparable to Catch Crops. However, unlike the Catch Crop action, the land cannot be grazed over winter and so soil erosion impacts should be greater.
Planting New Hedgerows	1,295,903 m	Assumed that a hedgerow would be replacing an existing fence, resulting in reduced connectivity for that field. Pollutant losses in surface runoff are reduced by 75%.
Protection of Watercourses from Bovines	12,745,368m	Nutrient losses resulting from livestock having direct access to water are reduced by 100%. Although this action might impact on sediment losses due to bank erosion, the modelling framework does not represent the impacts of bank stabilisation associated with fencing.
Arable Margins	3m Width:	All margins are assumed to reduce losses of

Action	Uptake	Expected Impact
	112,867 m 4m Width: 65,116 m 6m Width: 179,403 m	nitrate and dissolved phosphorus in surface runoff by 10% and 25% respectively. Losses of sediment (and associated particulate phosphorus) in surface runoff are reduced by 50%. Soil and fertiliser losses from the 1-2% of the field occupied by the margin were assumed to be completely negated.
Riparian Management	3m Width: 4,960 6m Width: 1,967 10m Width: 6,140 30m Width: 48,705	All margins are assumed to reduce losses of nitrate and dissolved phosphorus in surface runoff by 10% and 25% respectively. Losses of sediment (and associated particulate phosphorus) in surface runoff are reduced by 50%. These values are the same as for an arable margin as although the grass margins are wider, they are not replacing bare land over winter. Fencing is required so nutrient losses resulting from livestock having direct access to water are reduced by 100%. There is no impact of land going out of production as it is assumed farm level fertiliser use would be maintained to main forage production.
Environmental Management of Fallow Land	1,339 ha	As no inorganic fertiliser is allowed, losses associated with nitrogen and phosphorus fertiliser are reduced by 100%. Losses of nitrogen from soil and crop residue mineralisation were reduced by 50% to account for reduced stimulation of mineralisation in the absence of cultivation and potentially some over-winter nitrogen uptake. Soil erosion (and associated particulate phosphorus) were assumed to be reduced by 25%. Unlike a catch crop, there are no establishment issues with the fallow land (as it is sown much earlier than a catch crop) and the fallow land cannot be grazed.
Farmland Habitat	Grassland : 89,648 ha Arable: 510 ha	Impacts assumed to be identical to the Low Input Permanent Pasture and Environmental Management of Fallow Land actions.

Action	Uptake	Expected Impact
Low Input Permanent Pasture	235,384 ha	Nitrogen fertiliser losses reduced by ~70% on dairy farms and ~50% on beef and sheep farms. Losses from livestock excreta and manure reduced by ~60% on dairy farms and ~40% on beef and sheep farms. These are scaled back to reflect the proportion of surveyed farms actually making changes (from 80% on dairy farms to 1% on sheep farms).
Farmland Bird Actions	Breeding Waders: 1,025 ha Chough: 9,985 ha Corncrake: 131 ha Hen Harrier: 38,006 ha Twite A: 3,082 ha Geese and Swans: 12,265 ha Twite C: 2 ha Grey Partridge: 86,689 m	Geese and Swan action on arable land assumed to be as per a Catch Crop Grey Partridge action assumed to be as an Arable Grass Margin or a Riparian Margin depending upon the land use. Other actions assumed to be as per Low Input Permanent Pasture (area-weighted fertiliser constraint of these actions is close to 40 kg ha^{-1}, as required by Low Input Permanent Pasture).