Model Evaluation of GLAS

Report on Baseline Pollutant Losses

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Table of 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 provide a complementary intermediate between result and impact indicators by forecasting the potential long-term impact of GLAS management interventions in advance of long-term environmental monitoring for impact detection. 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)
- **MANNER** and **NARSES** – Ammonia (Chambers *et al*., 1999; Webb and Misselbrook, 2004)
- **FIO-FARM –** Faecal Indicator Organisms (Anthony and Morrow, 2011)
- **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 options considered in this project will be mapped to one or more mitigation actions.

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** and **efficiency**. Applicability 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. Efficiency measures physical or system limitations on an action. For example, the filtering effect of buffer strips is reduced on steep slopes. Measures of applicability and efficiency will be derived 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 will be 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).

Future work within this project will involve the development of a database of the targeting and effectiveness of mitigation actions, with which the impact of GLAS on pollutant losses can be derived.

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.

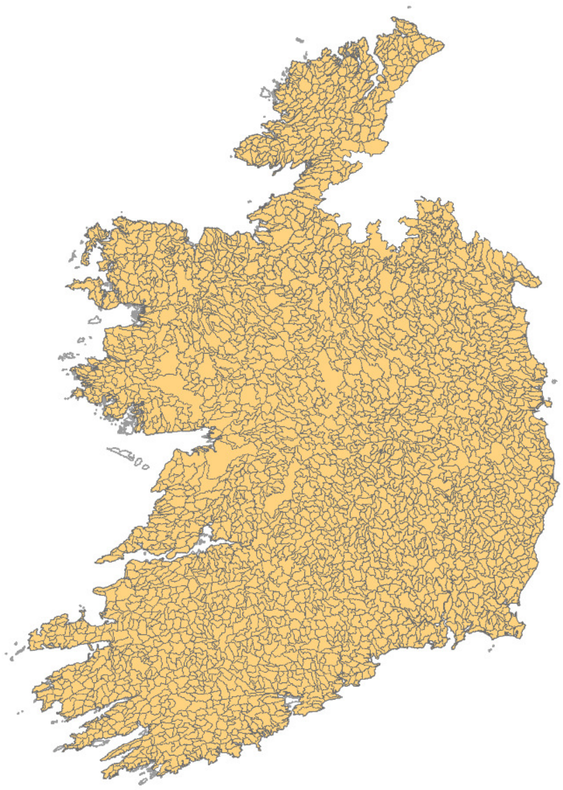


Figure 1-1 Catchments defined for Water Framework Directive reporting

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.

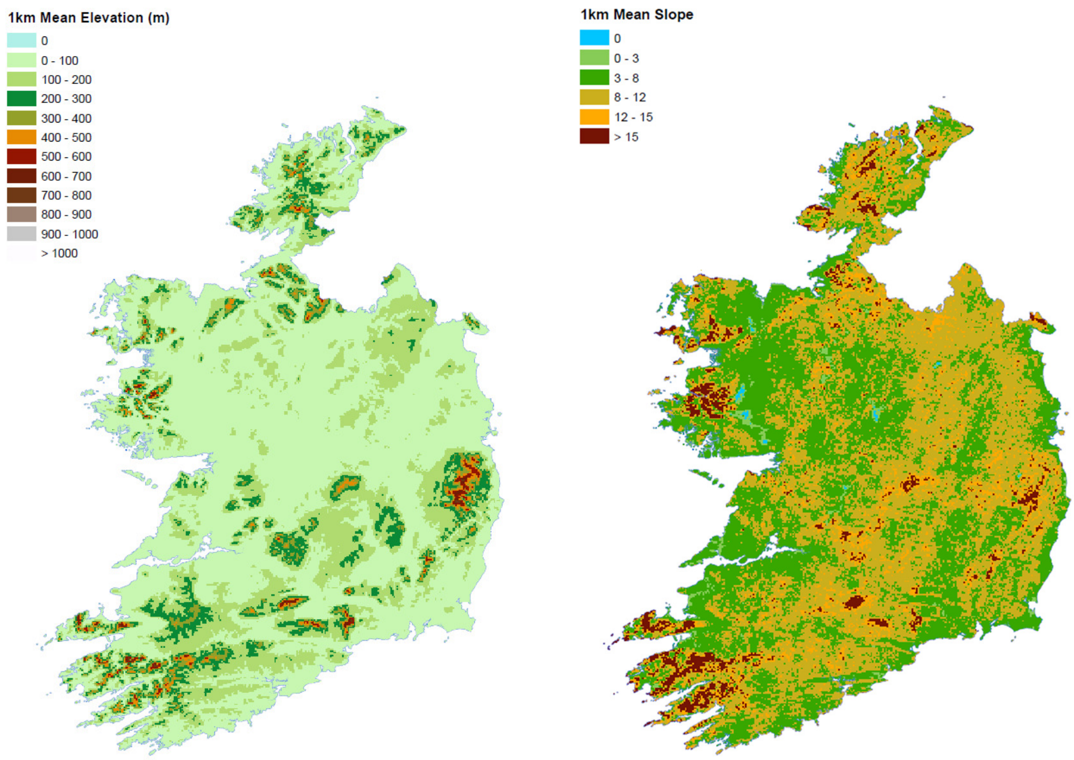


Figure 2-1 Mean elevation and slope calculated for each 1km^2 in Ireland.

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.

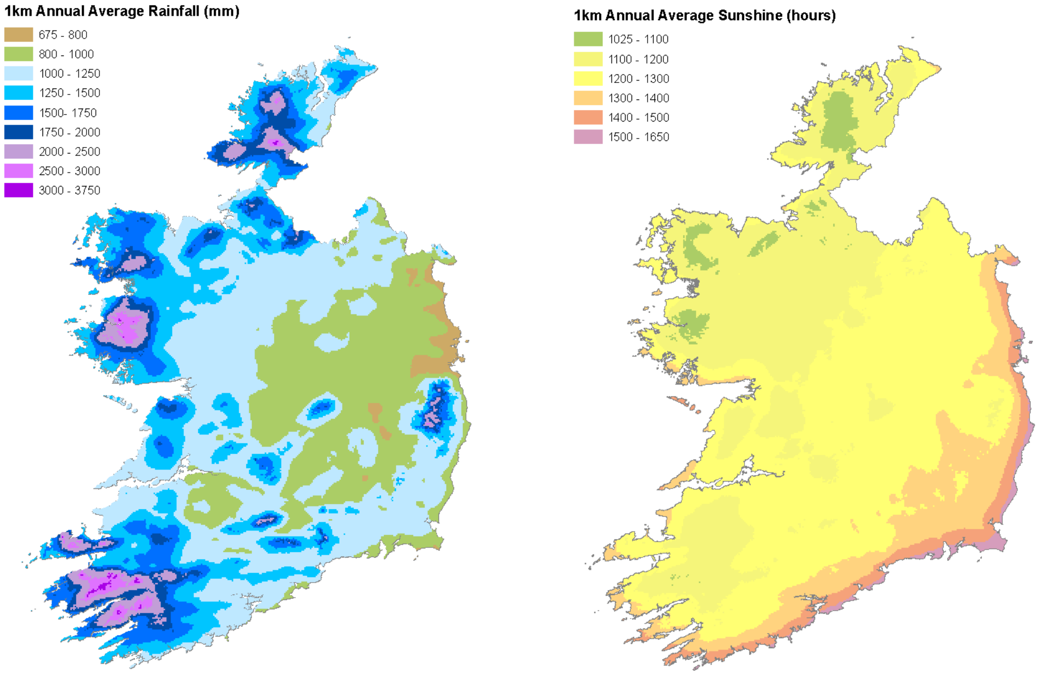


Figure 2-2 Annual average rainfall and sunshine hours for the period 1981-2010

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:

$$R_d = a + bR_M + c{R_M}^2$$

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

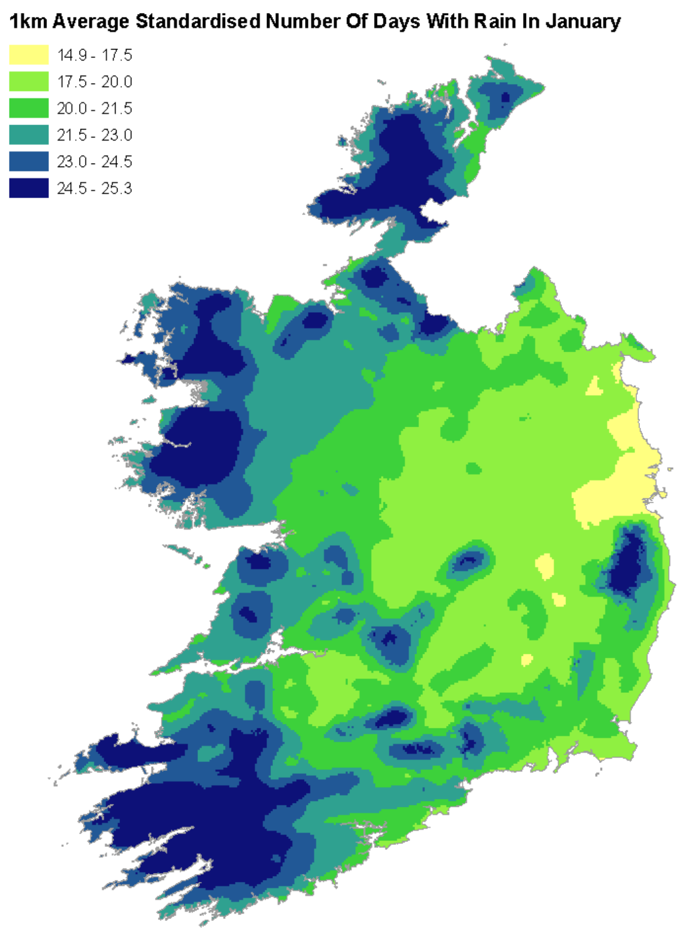


Figure 2-3 Rain days in January, dervied from data for 1986 – 2016.

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

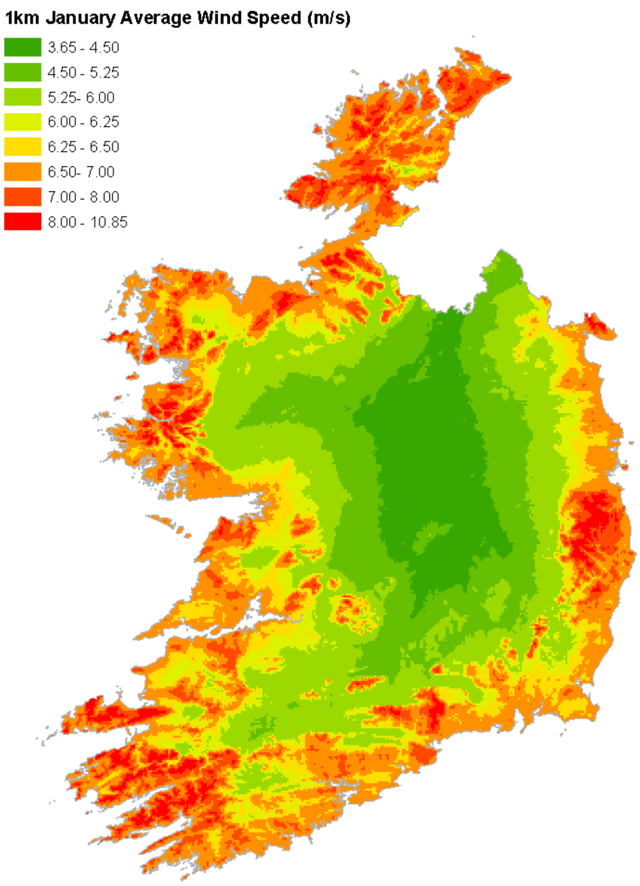


Figure 2-4 Wind speed in January, dervied from data for 1986 – 2016.

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).

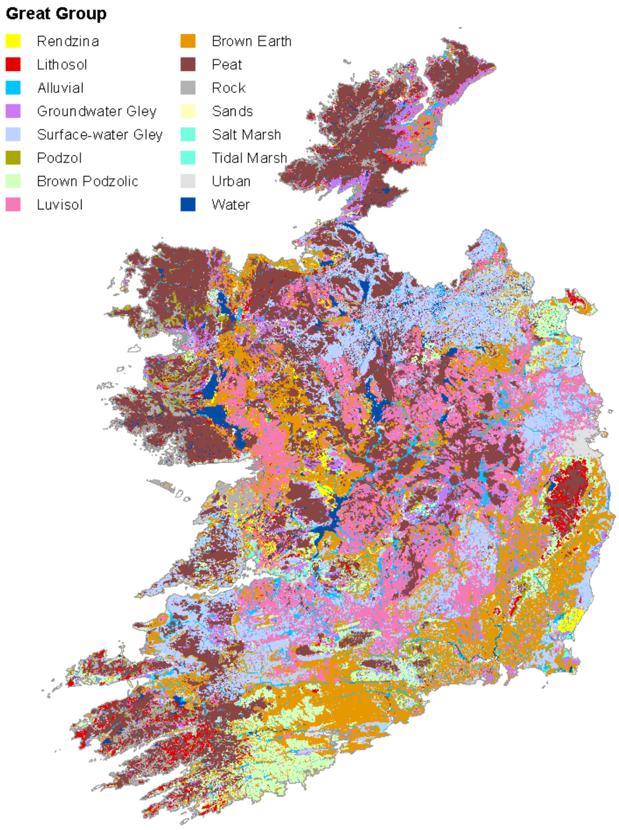


Figure 2-5 Great group assigned to each area of land

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.

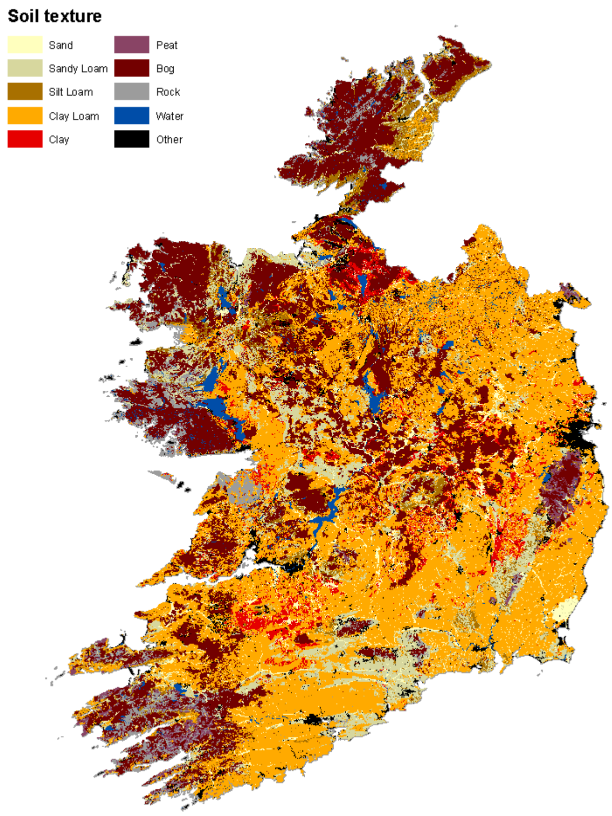


Figure 2-6 Top soil texture for each area of land

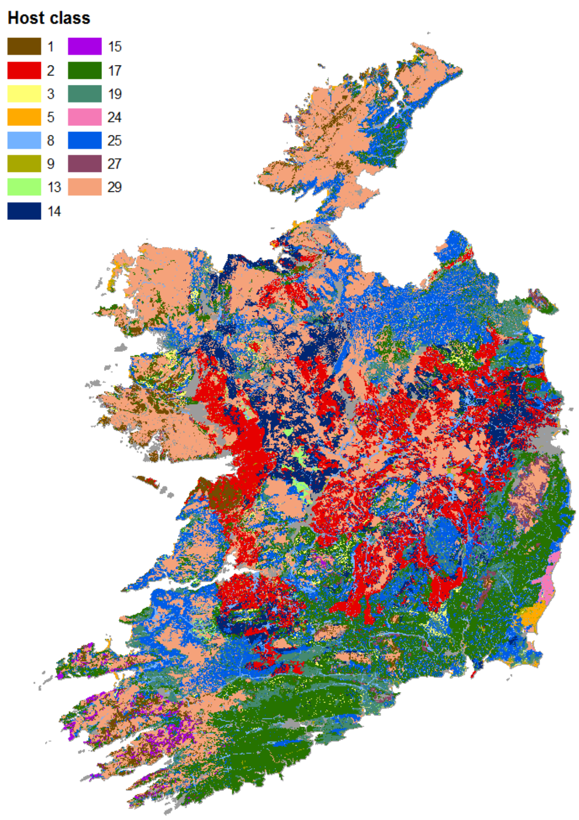


Figure 2-7 HOST category for each area of land

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.

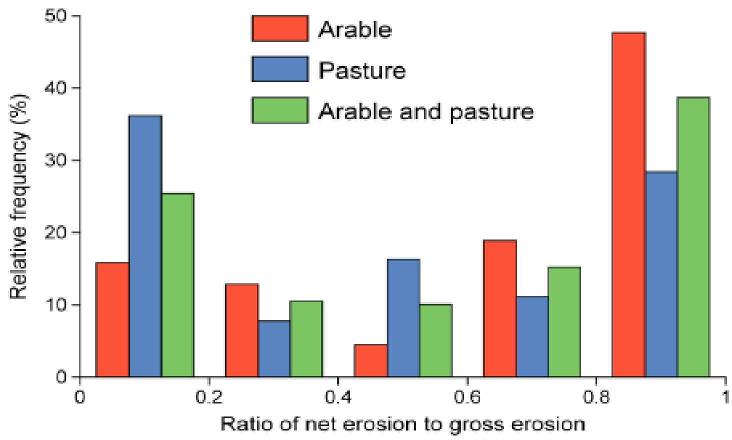


Figure 2-8 Frequency distribution for the ratio of net erosion to gross erosion for individual fields monitored using the Caesium-137 radiometric technique as part of Defra project SP0413 (Zhang *et al*., 2011).

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.

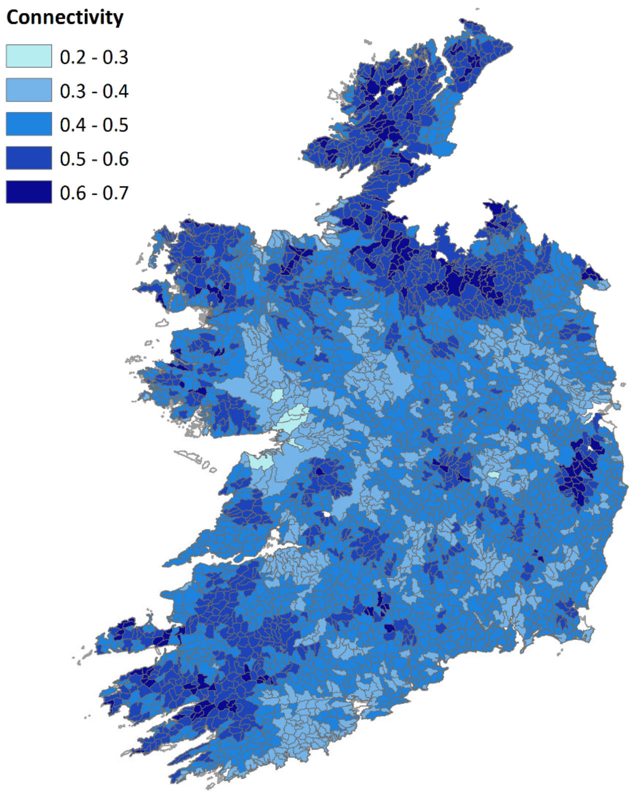


Figure 2-9 Average calculated connectivity for each WFD waterbody.

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. 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. Once 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

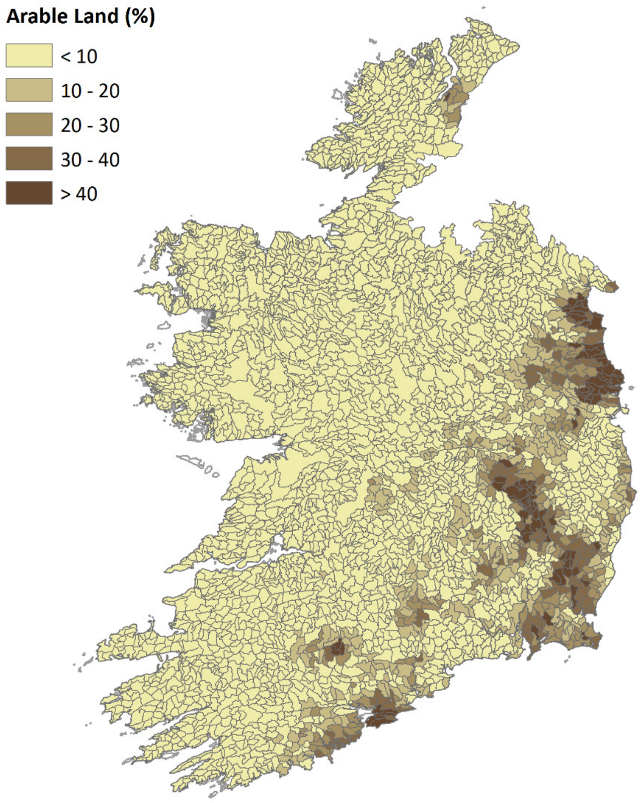


Figure 2-10 Percentage of each WFD waterbody catchment occupied by arable land.

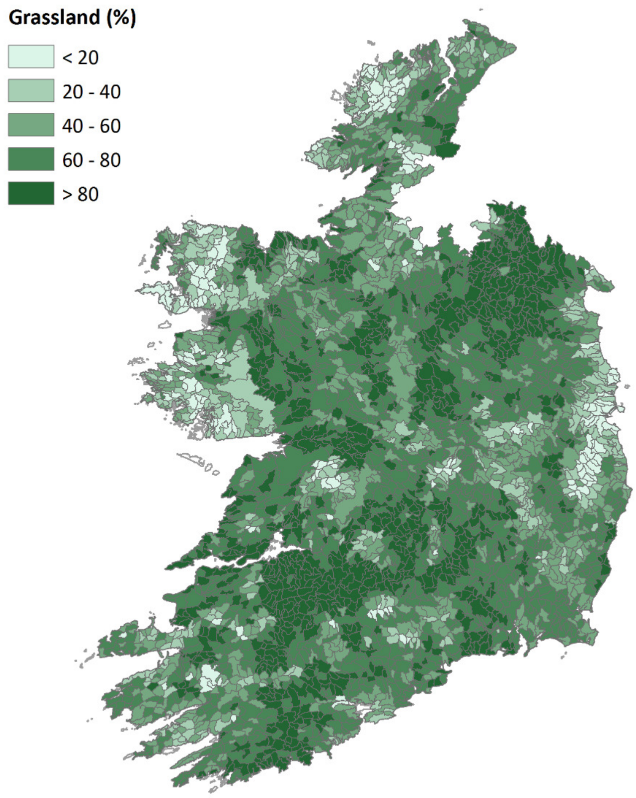


Figure 2-11 Percentage of each WFD waterbody catchment occupied by improved grassland.

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 therfore 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.

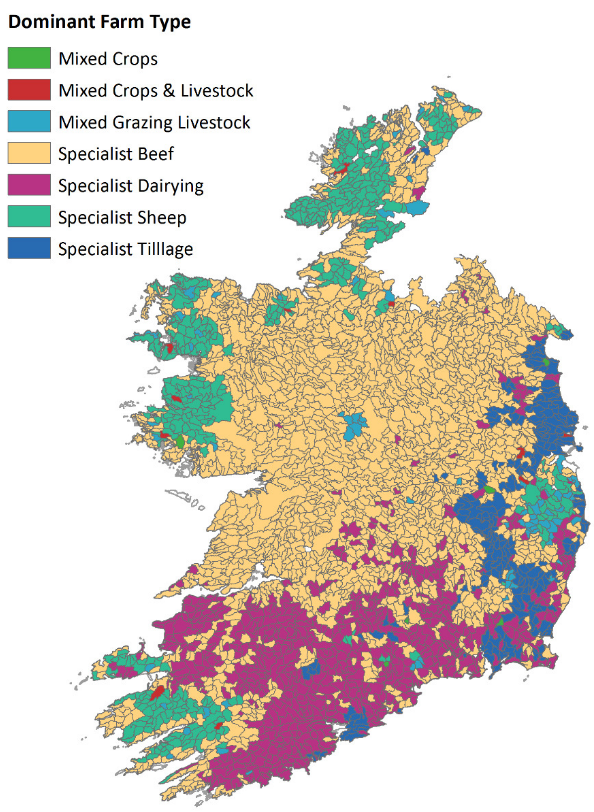


Figure 2-12 Dominant farm types, by farm area, in the WFD waterbodies

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 (inc. 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.*, 2013) 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)

Dataset / Parameter	Source
	English and Welsh data (Forbes *et al*., 1980)
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)
Use of animal waste management systesms	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 (Defra WQ0103) 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 2017
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 and manure management (Duffy *et al.*, 2016). Indirect nitrous oxide losses calculated from nitrate leaching losses.

Dataset / Parameter	Source
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 inc 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 inc 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 (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 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; 2001). 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.

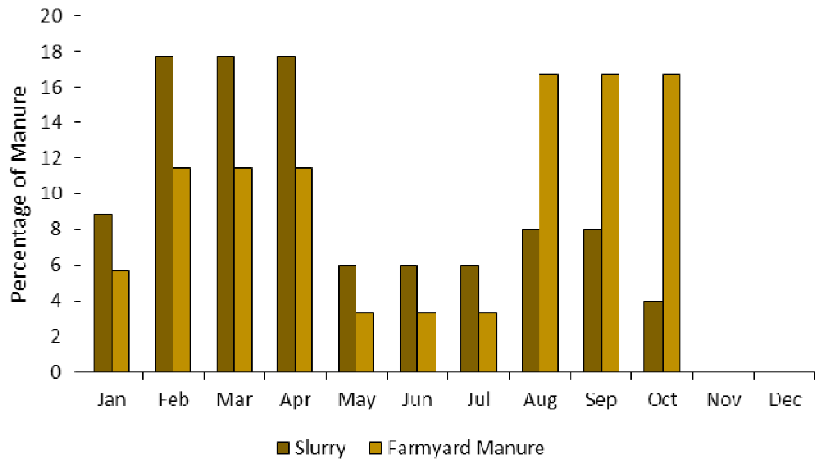
Table 3-9 Periods when application of fertilisers to land is prohibited (Government of Ireland, 2014).

		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

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

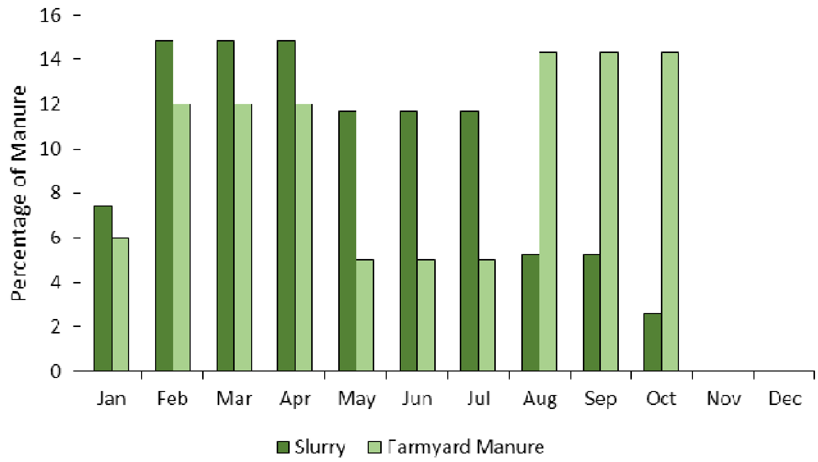


Figure 3-1 Manure application timings to different crop types, based on data from Manures-GIS (Defra Project WQ0103) constrained by closed periods for manure application in Ireland and surveyed data available in Hennessy *et al.*, (2011).

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. 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 (Teagasc 2017). 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.

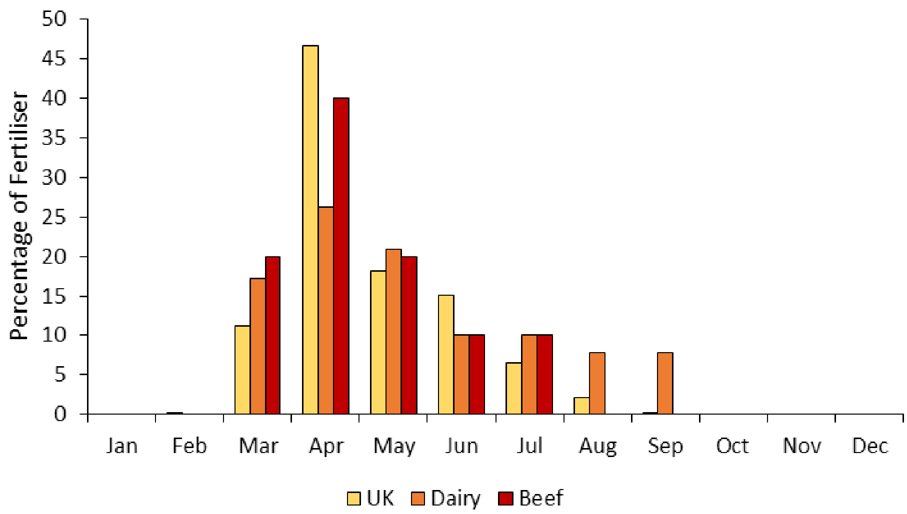
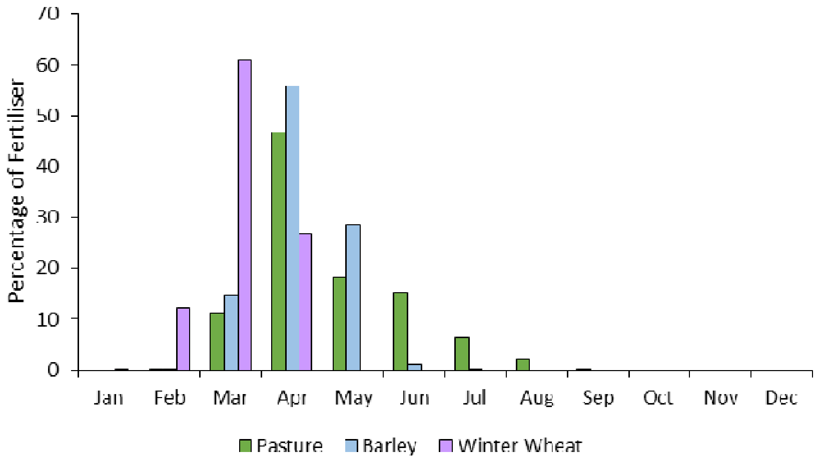


Figure 3-2 Nitrogen fertiliser timing data for grassland based on Teagasc advice for Dairy and Beef farms stocked at the national average stocking rate, and for the UK from the British Survey of Fertiliser Practice.

N



P_2O_5

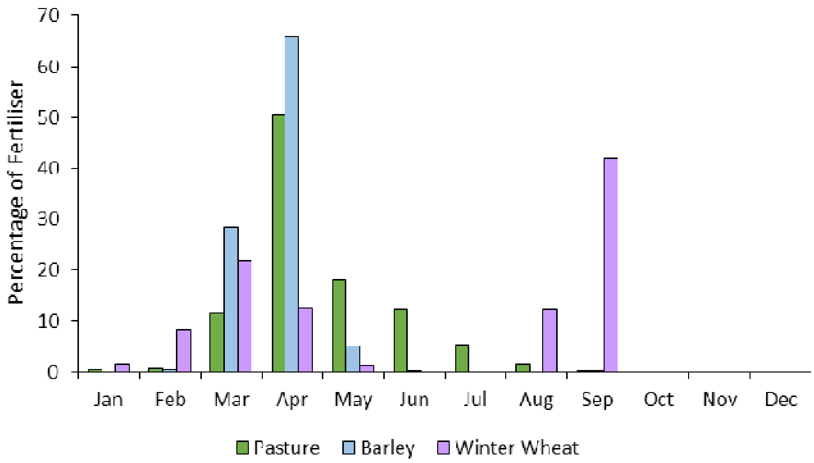


Figure 3-3 Fertiliser timings for common crop types based on British Survey of Fertiliser Practice data for 2008 to 2010, constrained by fertiliser closed periods in Ireland.

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 of 10.1, 11.0 and 5.6 mg kg^{-1} respectively. Converting these values to Olsen's P (using data in Foy *et al*., 1997 and accounting for bulk density) suggests that PSYCHIC under predicts dissolved phosphorus loss from soils in Irish conditions by up to 50%. Comparison of the PSYCHIC rules with data from approximately 1,900 fields in 4 small catchments in Jordan *et al.,* (2012) produces a much closer fit. This will be reviewed in the next iteration of the modelling framework within this project. The contribution of dissolved soil P to the national loss is approximately 13%, so any modification of the rules should not change overall loads significantly.

Table 3-15 Estimation of soil Olsen P (mg P kg^{-1}) from soil text 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.,* (2013) 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.,* (2006) 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 4 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), with waterlogged areas the main contributor to this pollution (3.3% of the total sediment load; 4.7% of the total FIO load). The impacts on nitrous oxide are very significant, resulting in the load increasing by over a third (so that over 25% 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 will be used to estimate the effectiveness of mitigation actions applied alone.

The modelling framework used in this project will by necessity calculate 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.

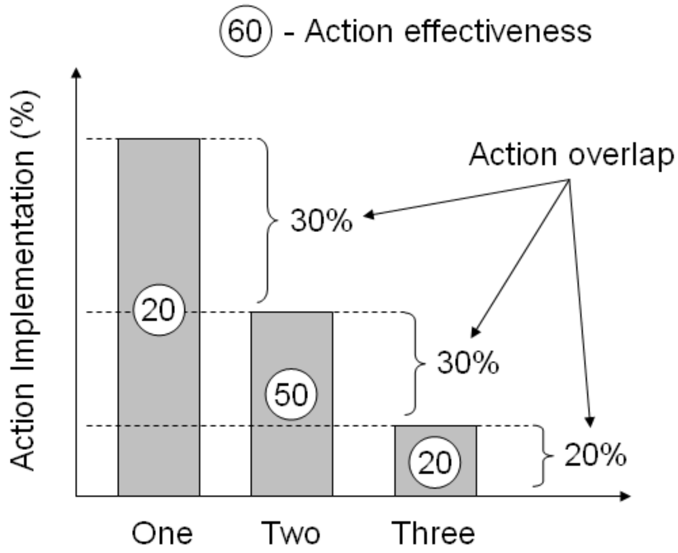


Figure 3-4 Schematic of the method of calculating the net effectiveness of multiple mitigation actions affecting the same pollutant source coordinate, assuming maximum overlap of action implementation.

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 Agricultural Pollutant Losses

4.1 Baseline losses and source apportionment

National pollutant losses and footprints (losses expressed per hectare of agricultural land) are shown in Table 4-1. 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 4-2. 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 4-1 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 4-3 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 4-2), 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 4-3) and delivery pathway (Figure 4-4; 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 4-4 and Table 4-5 show a detailed breakdown of the source and pathway apportionment by land use for each pollutant (combining Figure 4-2, Figure 4-3 and Figure 4-4). 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 4-5 and Figure 4-6. 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 4-1 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 4-2 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	P	Z	N_2O	CH_4
	(kg ha^{-1})	(kg ha^{-1})	(kg ha^{-1})	(kg ha^{-1})	(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	-	-	-	-	-

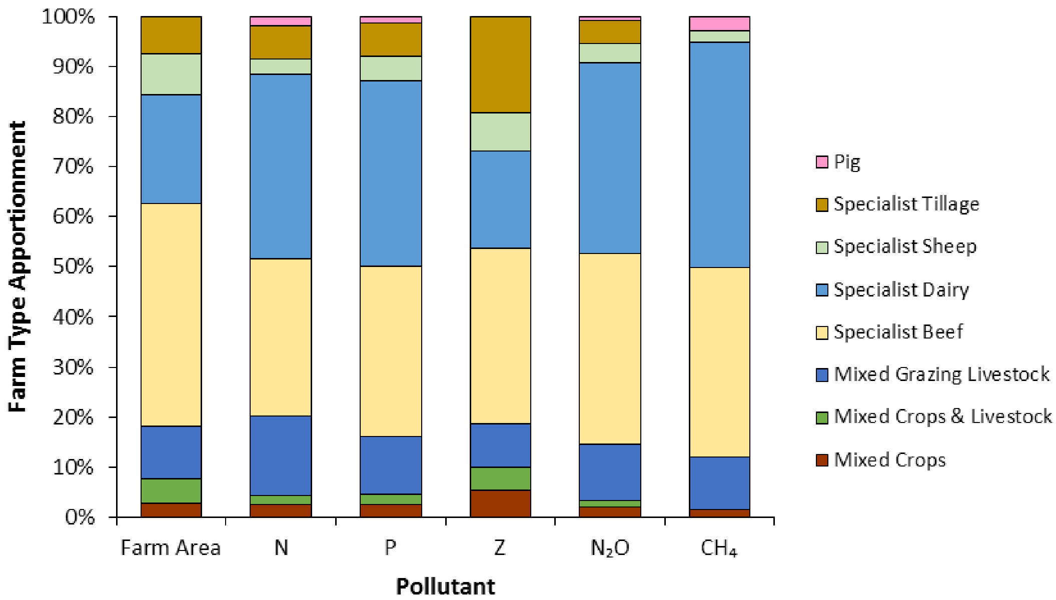


Figure 4-1 Apportionment of national agricultural pollutant losses by farm type.

Table 4-3 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

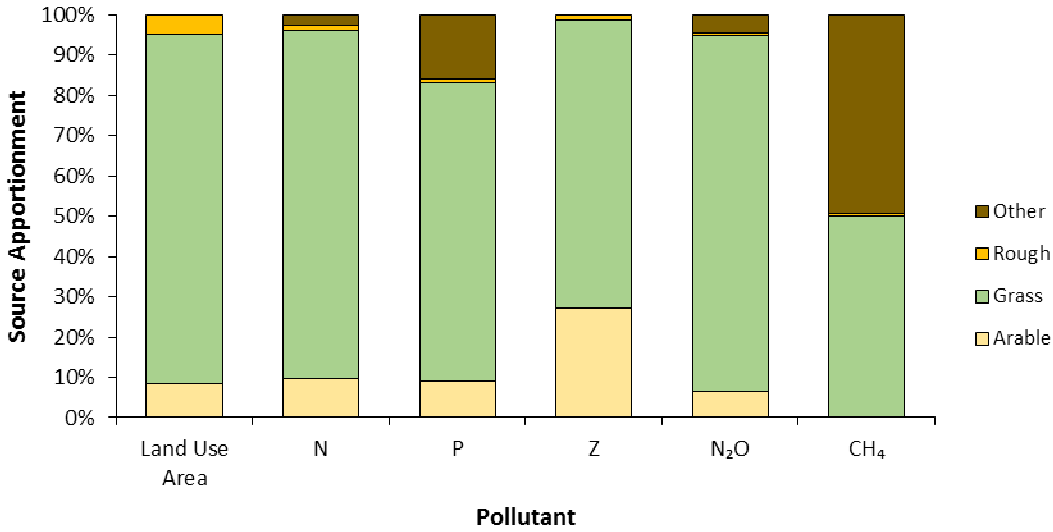


Figure 4-2 Apportionment of national agricultural pollutant losses by source area.

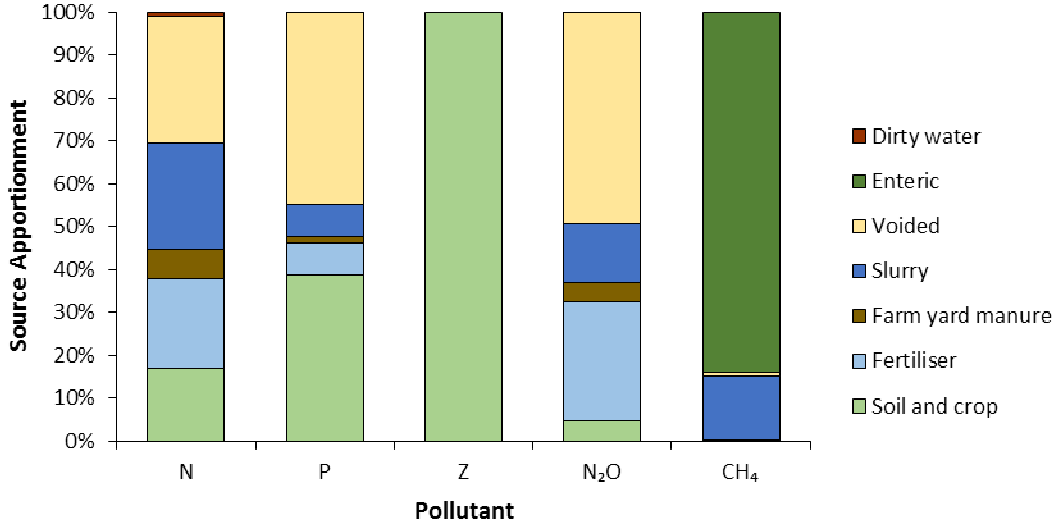


Figure 4-3 Apportionment of national agricultural pollutant losses by source type.

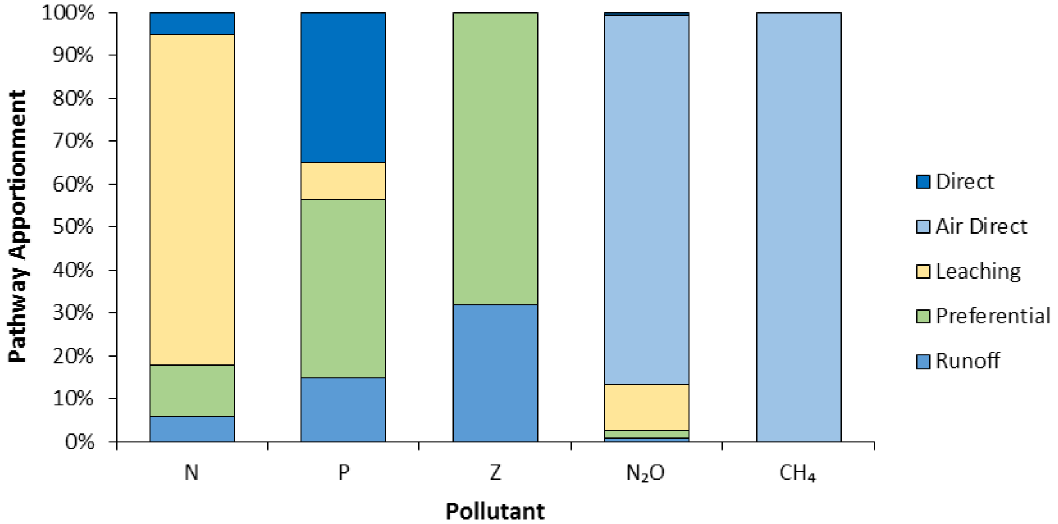


Figure 4-4 Apportionment of national agricultural pollutant losses by pathway.

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

Nitrate	Total Loss	Source Apportionment (%)						Pathway Apportionment (%)				
	(kg ha^{-1})	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	Source Apportionment (%)						Pathway Apportionment (%)				
	(kg ha^{-1})	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	Source Apportionment (%)						Pathway Apportionment (%)				
	(kg ha^{-1})	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 4-5 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

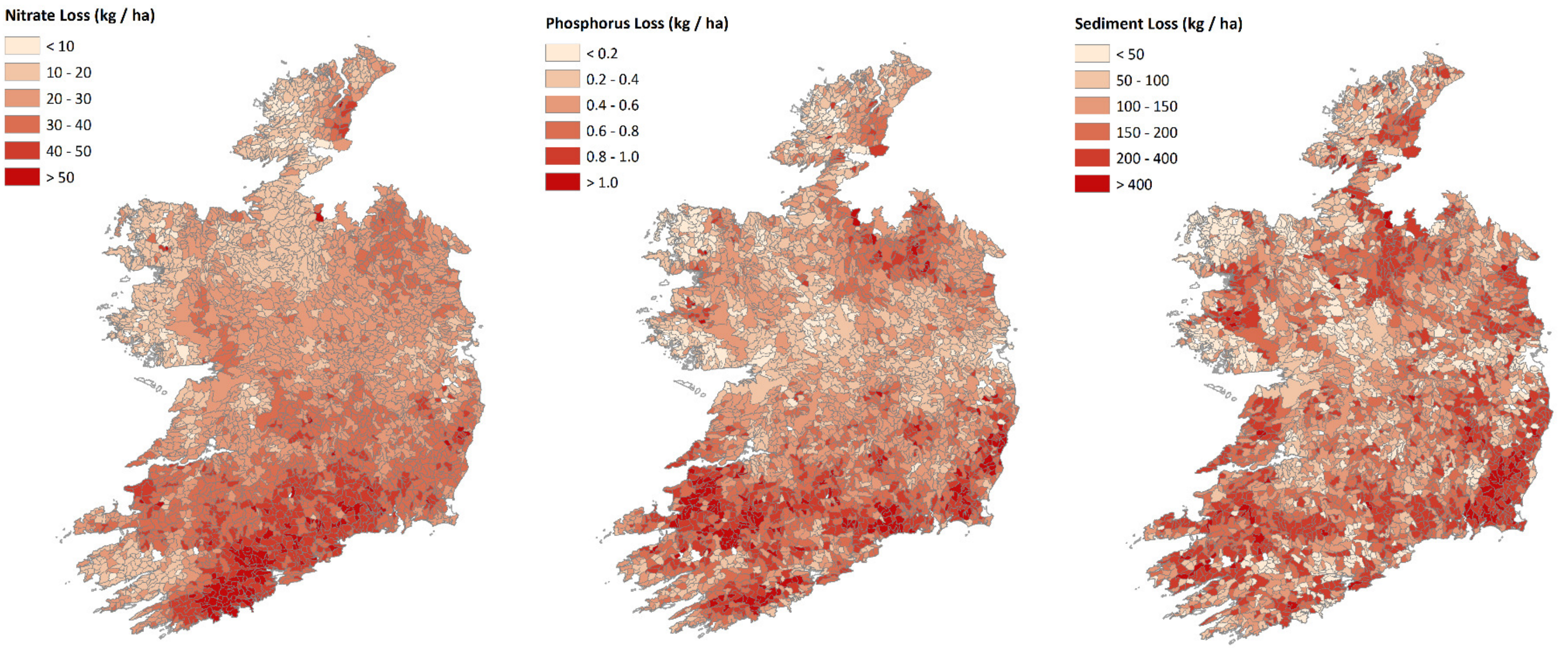


Figure 4-5 Annual average agricultural pollutant losses of nitrate, phosphorus and sediment for each WFD waterbody, expressed per hectare of agricultural land.

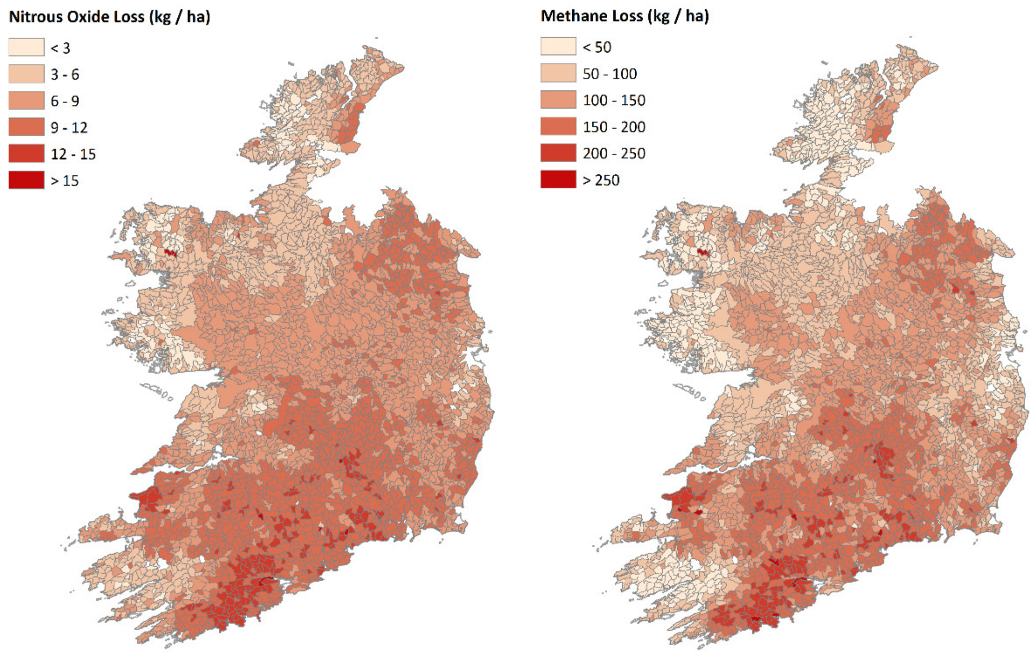
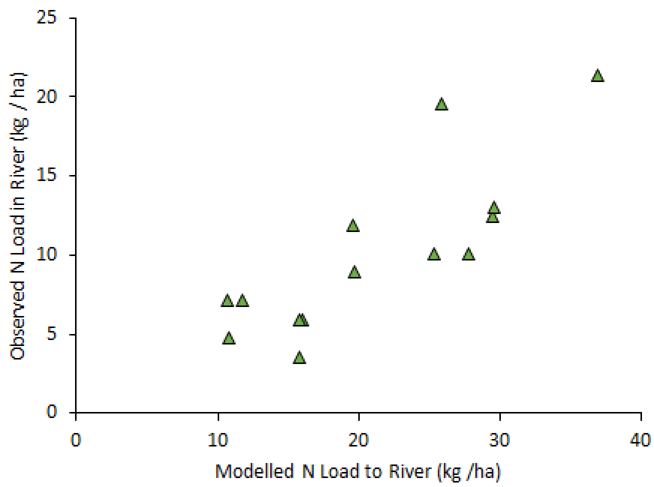


Figure 4-6 Annual average agricultural pollutant losses of nitrous oxide and methane for each WFD waterbody, expressed per hectare of agricultural land.

4.2 Initial 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 4-7 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.



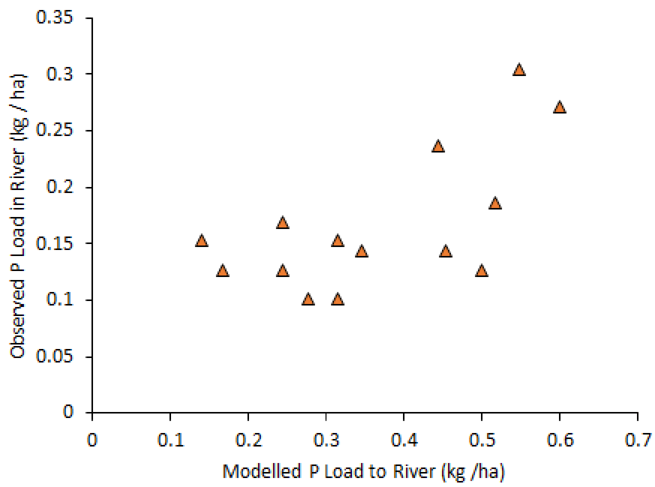


Figure 4-7 Comparison of modelled agricultural loads (predicted to river) with observed loads measured in rivers for 2011-2013 (O'Boyle *et al.*, 2016). Observed loads were available for 16 OSPAR monitoring sites. Loads are expressed per hectare of all land within the catchments. The modelled loads do 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 4-5).

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 4-5). 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.

5 Impacts of GLAS

This section describes the proportion of land and associated pollutant loads managed by farms in GLAS, provides a summary of the different GLAS actions that will be modelled in a subsequent report and then provides an example of how the actions will be modelled by quantifying the impacts of cover crops sown as a result of GLAS on sediment losses.

5.1 Land in GLAS

Figure 5-1 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 5-1), 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 5-1, where uptake of GLAS is lowest in dairying areas such as the south.

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. 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 4-2), are less likely to be in GLAS.

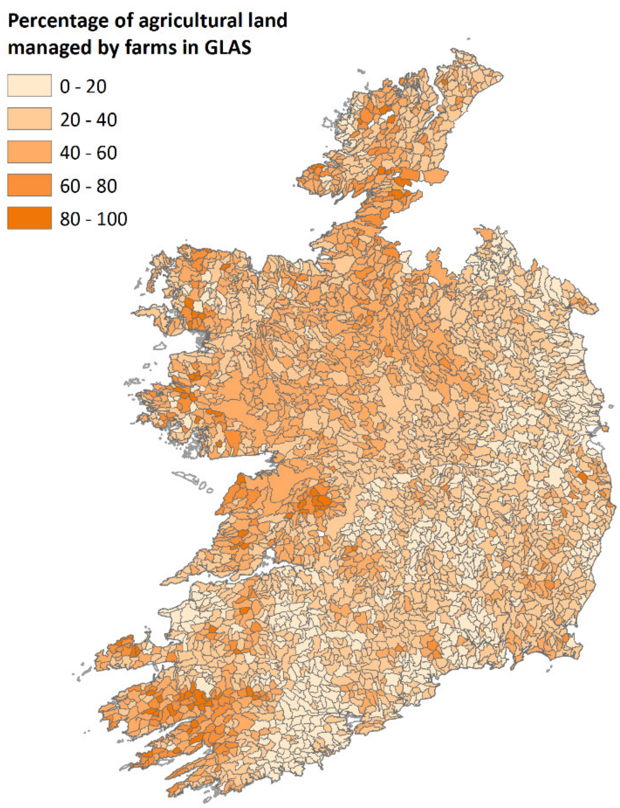


Figure 5-1 Percentage of all agricultural land managed by farms in GLAS, summarised by WFD waterbody.

Table 5-1 Percentage of land managed by farms in GLAS, summarised by land use and by farm type.

	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
Total	**32**	**30**	**32**	**44**

Table 5-2 Percentage of the national pollutant load from farms in GLAS

	N	P	Z	N_2O	CH_4
Percentage of national Load from farms in GLAS	27	28	33	27	23

5.2 Representation of GLAS actions in the modelling framework

Actions within GLAS have been examined to identify those methods that we will explicitly represent within the modelling work. These actions, together with a brief description of how their impacts will be represented in the modelling work, are described below. Where an action is listed as having an impact on nitrate, there will also be an impact on indirect emissions of nitrous oxide.

5.2.1 Arable Grass Margins

This will be represented by a reduction in pollutant delivery from field to river, resulting in lower emissions. The amount of land taken out of arable production, even for the 30 m margin, is likely to be negligible at catchment scale.

Pollutants affected: nitrate, phosphorus and sediment.

5.2.2 Catch Crops

Catch crops reduce soil erosion during heavy rainfall, provide better soil structure and aid the absorption of residual nitrogen, reducing nitrogen leaching from the soil.

Pollutants affected: nitrate, phosphorus and sediment.

5.2.3 Environmental Management of Fallow Land

Although predominantly aimed at increasing biodiversity by providing habitat and food for birds, this action may also have a number of positive impacts on soil and water quality. The actions require reduced fertiliser inputs onto the land parcel, and the presence of grass cover will improve soil quality and help prevent runoff and soil erosion, reducing losses of sediment and nutrients. There is also a secondary effect in terms of taking land out of production, where the action is applied to land previously cultivated.

Pollutants affected: nitrate, phosphorus and sediment.

5.2.4 Farmland Habitat (private natura)

This action relies on the production of a management plan to set stocking density and management of the parcel. Where this has resulted in a change in stocking density or in a reduction of inputs to the parcel, this will be represented in the modelling framework, with the reductions in stocking density and inputs derived from the responses to the survey by farmers who have implemented this option.

Pollutants affected: nitrate, phosphorus, sediment, nitrous oxide and methane.

5.2.5 Low Emission Slurry Spreading

The method of slurry application is an important factor that determines the potential utilisation efficiency of nutrients by the crop (grass or arable). Low emission spreading reduces the vulnerability of nutrients in manure to being lost in surface runoff and reduces emissions of ammonia. Reduced ammonia emissions increase the potential for nitrate losses if adjustments are not made to nitrogen fertiliser rates to account for the improved efficiency, but we would assume such adjustments are made.

Pollutants affected: nitrate and phosphorus.

5.2.6 Low Input Permanent Pasture

Parcels with this action have reduced fertiliser inputs and restrictions on stocking rates to achieve a more diverse sward with an increased flora and fauna. The predominant impacts will be on nutrient losses due to the reduced inputs. Reduced stocking rates can potentially lead to a reduction in poaching, reducing surface runoff and soil erosion. The modelling framework will utilise information on reductions in fertiliser usage and changes in stocking rates to represent this action.

Pollutants affected: nitrate, phosphorus, sediment, nitrous oxide and methane.

5.2.7 Minimum Tillage

Minimum tillage 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 fuel usage and potentially an increase in soil organic carbon, although these will not be modelled.

Pollutants affected: nitrate, phosphorus and sediment.

5.2.8 Planting New Hedgerows

The planting of new hedgerows will potentially reduce the connectivity between fields for runoff, but as the measures include fencing, there is also the potential to reduce livestock ingress to watercourses. The modelling framework will represent the effect of this action only in parcels adjacent to watercourses, as this is where the predominant effect will occur.

Pollutants affected: nitrate, phosphorus and sediment.

5.2.9 Protection of Watercourses from Bovines

Livestock grazing along a watercourse can lead to direct pollution of water with urine and faeces, resulting in nutrients and pathogens entering the water. This can destroy aquatic habitats and lower the quality of water potentially used for human consumption. Excluding bovines from watercourses will prevent the breakdown of vegetation on the banks of the watercourse and thus reduce bank erosion. There may also be a reduction in runoff and pollutant losses due to the creation of a form of buffer strip between the fence and the watercourse.

Pollutants affected: nitrate, phosphorus and sediment.

5.2.10 Riparian Margins

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 (no direct excretion to water and reduced bank erosion), but with an increased potential to reduce nutrient and sediment runoff due to the presence of the vegetated margin which will act as a buffer strip.

Pollutants affected: nitrate, phosphorus and sediment.

5.2.11 Wild Bird Cover

This action is intended to provide habitat and food for birds, but may also have a number of positive impacts on soil and water quality. The action limits fertiliser inputs onto the land parcel, but since the crop cannot be harvested, fertiliser use is likely to be nil. The presence of crop cover over winter will help prevent runoff and soil erosion, reducing losses of sediment and nutrients.

Pollutants affected: nitrate, phosphorus and sediment.

5.2.12 Farmland bird actions

In addition to the key actions described above, we will represent the effects of other actions focussed on farmland birds as one or more blocks, depending on their primary impact. For example, those actions that have a prescription on fertiliser used for grassland fields (breeding waders, chough, corncrake, twite) will be modelled together as a reduction in the fertiliser inputs to the parcel.

5.2.13 Other actions

The following actions have not been represented as we believe that these actions will have no significant effect on water quality:

- Bat boxes
- Bird boxes
- Conservation of solitary bees
- Coppicing of hedgerows
- Laying of hedgerows
- Protection of archaeological monuments
- Traditional dry stone wall maintenance

5.3 Potential impacts of cover crops on sediment losses

The following section describes the calculation of the impacts of catch crop options within GLAS on sediment losses, as an example of how the impacts of mitigation can be represented with the framework using the source apportionment system and GLAS scheme data. Impacts of cover crops on other pollutants will be considered in the subsequent report.

The net effect of the catch crop intervention is calculated as the product of the percentage uptake, targeting and efficiency terms. Uptake is the percentage of the relevant crop area on which the intervention is implemented, and is derived from scheme records of payments to land managers. Targeting is the percentage of the total baseline pollutant loss that occurs from the source area, method of pollutant mobilisation and delivery pathway affected by the intervention. Efficiency is the percentage reduction in pollutant loss from the target and is derived from published reviews of field experimentation.

In the case of the catch crop, the uptake is derived from the option areas in the scheme payments database provided by the Irish Government. The uptake area was reported for individual farms and aggregated as a percentage of the arable spring sown crop area within each water body, by representative farm type. Overall, 17% of the arable crop area on farms participating in GLAS was sown with a catch crop. Uptake ranged from less than 8% on specialist dairy farms to 19% on the specialist cereals farm.

The targeting was based on a conceptual model of how the catch crop worked to control pollutant loss. In this case, it acts to firstly reduce the splash detachment of soil by absorbing the kinetic energy of rain drops before they hit the ground, and secondly to slow runoff and cause detached soil that was entrained in runoff to be redeposited before leaving a field. The intervention therefore reduces the loss of sediment in both surface runoff from fields, and in under-ground drain flow resulting from the infiltration of impacting rainfall laden with detached sediment that enters the top soil by way of cracks and macropores. Over-winter growth of the catch crop will also take up nitrate from the soil, reducing the risk of surplus nitrate being leached before a bare soil is sown in the spring. Note that the terms catch and cover crops are used interchangeably. Cover crop is normally used when referring to the protection of the soil from rainfall detachment, and catch crop is used when referring to the over-winter uptake of surplus nutrient remaining from the previous growing season. However, the multiple effects of the intervention are not calculated in this example and we consider only the protection of the soil from rainfall detachment.

The efficiency of the intervention was sourced from literature review. 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 in field experiments 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 of bare soil. 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 losses in the range 30 to 60%. Based on these reviews, we have assumed that the effect of the catch crop is to reduce incidental losses of sediment in surface runoff and drain flow by 50% and 25% respectively, on the fields affected by the intervention.

By spatial integration of the uptake, targeting and efficiency terms, we calculated that the catch crop intervention would reduce total sediment losses from farms participating in GLAS by 9%. Together with information on the proportions of land in scheme (i.e. approximately 30% as shown in Table 4-6), the net effect is a 2% reduction in sediment loss from all agricultural land on all farms in Ireland. Spatial variation in uptake and the importance of the surface runoff pathway will result in a range of local effectiveness values for each waterbody.

6 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.

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

- Areas of scheme participation
- Input loads controlled by farms in scheme (and the proportion of regional and national totals)
- Baseline pollutant loss from farms in scheme (and the proportion of regional and national totals)

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.

These datasets will now be used to assess the impacts of the current uptake of GLAS agreements on agricultural pollution from farms in GLAS. This will allow 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.

7 References

Alderfer, R. B. and Robinson, R. R. (1947) Runoff from pastures in relation to grazing intensity and soil compaction. Agronomy Journal, 39, 948-958.

Anthony, S. and Collins, A. (2006) Sediment gap analysis to support Water Framework Directive. Defra project WQ0106, Final Report, 136 pp.

Anthony, S. and Lyons, H. L. (2006) Identifying the gap to meet WFD and best policies to meet the gap. Interim Project Report to Defra, Project WT0719CSF, ADAS UK Ltd, 72 pp.

Anthony, S., Turner, T., Roberts, A., Harris, D., Hawley, J., Collins, A., and Withers, P. (2008) Evaluating the extent of agricultural phosphorus losses across Wales. Defra project WT0743CSF, Final Report, ADAS UK Ltd, 281 pp.

Anthony, S., Duethman, D., Gooday, R., Harris, D., Newell-Price, P., Chadwick, D. and Misselbrook, T. (2009) Quantitative Assessment of Scenarios for Managing Trade-Off between the Economic Performance of Agriculture and the Environment and Between Different Environmental Media. Final Report, Defra Project WQ0106 (Module 6), 95 pp.

Anthony, S. G. and Morrow, K. (2011) Prototype Farm Scale Faecal Indicator Budget Model. Final report, Defra project WQ0111 – Faecal Indicator Organism Losses from Farming Systems (FIO-FARM), 89 pp.

Anthony, S., Jones, I., Naden, P., Newell-Price, P, Jones, D., Taylor, R., Gooday, R., Hughes, G., Zhang, Y., Fawcett, L., Simpson, D., Turner, A., Fawcett, C., Turner, D., Murphy, J., Arnold, A., Blackburn, J., Duerdoth, C., Hawczak, A., Pretty, J., Scarlett, P., Laize, C., Douthwright, T., Lathwood, T., Jones, M., Peers, D., Kingston, H., Chauhan, M., Williams, D., Rollett, A., Roberts, J., Old, G., Roberts, C., Newman, J., Ingram, W., Harman, M., Wetherall, J. and Edwards-Jones, G. (2012) Contribution of the Welsh agri-environment schemes to the maintenance and improvement of soil and water quality, and to the mitigation of climate change. Welsh Government, Agri-Environment Monitoring and Technical Services Contract Lot 3: Soil, Water and Climate Change (Ecosystems), No. 183/2007/08, Final Report, 477 pp + Appendices.

ADAS (2008) The National Inventory and Map of Livestock Manure Loadings to Agricultural Land: MANURES-GIS. Final Report for Defra Project WQ0103. ADAS, Wolverhampton, UK.

Baggott, S., Brown, L., Cardenas, L., Downes, M., Garnett, E., Hobson, M., Jackson, J., Milne, R., Mobbs, D., Passant, N., Thistlethwaite, G., Thomson, A. and Watterson, J. (2006) United Kingdom greenhouse gas inventory, 1990 to 2004. AEA Technology Ltd, 468 pp.

Bagshaw, C. (2002) Factors influencing direct deposition of cattle faecal material in riparian zones. MAF Technical Paper, No. 2002/19, New Zealand, 20 pp.

Ball, B., Scott, A., and Parker, J. (1999) Field nitrous oxide, carbon dioxide and methane fluxes in relation to tillage, compaction and soil quality in Scotland. Soil and Tillage Research, 53, 29-39.

Bhandral, R., Saggar, S., Bolan, N. and Hedley, M. (2007) Transformation of nitrogen and nitrous oxide emission from grassland soils as affected by compaction. Soil and Tillage Research, 94, 482-492.

Boorman, D., Hollis, J. and Lilly, A. (1995) Hydrology of soil types: a hydrologically-based classification of the soils of the United Kingdom. Institute of Hydrology. Project No.126. Institute of Hydrology, Wallingford,137pp.

British Survey of Fertiliser Practice (2004-2011) Available from: https://www.gov.uk/government/collections/fertiliser-usage [accessed 1st April 2014]

Brogan, J., Crowe, M. and Carty, G. (2001) Developing a national phosphorus balance for agriculture in Ireland: A discussion document. Environmental Protection Agency, Johnstown Castle, Co. Wexford, Ireland.

Chambers, B., Lord, E., Nicholson, F. and Smith, K. (1999) Predicting nitrogen availability and loses following applications of manures to arable land: MANNER. Soil Use and Management, 15, 137-143.

Chamen, T. (2006) Controlled traffic farming: literature review and appraisal of potential use in the UK. HGCA Research Review No. 59, 61pp.

Collins, A.L., Stromqvist, J., Davison, P. and Lord, E. (2007) Appraisal of phosphorus and sediment transfer in three pilot areas identified for catchment sensitive farming initiative in England – application of the prototype PSYCHIC model. Soil Use and Management, 23, 117-132.

Connolly, J. and Holden, N.M. (2009) Mapping peat soils in Ireland: updating the derived Irish peat map. Irish Geography, 42, 3, 342-352.

CSO (2010) Census of Agriculture 2010 – Final Results. Central Statistics Office, Cork, Ireland.

D'Hour, P., Hauwuy, A., Coulon, J.B., and Garel, J.P. (1994) Walking and dairy cattle performance. Annales de Zootechnie, 43, 369-378.

Davidson, E. (1991) Fluxes of nitrous oxide and nitric oxide from terrestrial ecosystems. In Rogers, J. and Whitman, W. (Editors) Microbial production and consumption of greenhouse gases: methane, nitrogen oxides and halomethanes. American Society of Microbiology, Washington DC, 219-235.

Davies-Colley, R., Nagels, J., Smith, R., Young, R. and Phillips, C. (2004) Water quality impact of a dairy herd crossing a stream. New Zealand Journal of Marine and Freshwater Research, 38, 569-576.

Davison, P., Hutchins, P.G., Anthony, S.G., Betson, M., Johnson, C. and Lord, E.I. (2005) The relationship between potentially erosive storm energy and daily rainfall quantity in England and Wales. Science of the Total Environment, 344, 1-3, 15-25.

Davison, P., Withers, P., Lord, E., Betson, M. and Stromqvist, J. (2008) PSYCHIC – A process based model of phosphorus and sediment mobilisation and delivery within agricultural catchments. Part 1 – Model description and parameterisation. Journal of Hydrology, 350, 290-302.

Defra (2006) Compendium of UK Organic Standards. Defra Organic Branch, London, UK. 104pp.

Defra (2006) Nutrient management decision support system (PLANET). Final report for DEFRA project KT0113. Defra, London, UK.

DEFRA (2008) Major upgrade of SEISMIC to extend to the whole of Great Britain. Final report for DEFRA project PS2225A. Defra, London, UK.

Demal, L. (1982) An intensive water quality survey of stream cattle access sites. Stratford Avon River Environment Management Project, Technical Report R-19,Upper Thames River Conservation Authority, Ontario, 36 pp.

Dobbie, K. and Smith, K. (1996) Comparison of methane oxidation rates in woodland, arable and set-aside soils. Soil Biology and Biochemistry, 28, 1357-1365.

Duffy, P., Hanley, E., Black, K., O'Brien, P., Hyde, B., Ponzi, J. and Alam, S. (2016) Nation Inventory Report 2016: Greenhouse gas emissions 1990-2014 reported to the United Nations framework convention on climate change. Environmental Protection Agency, Johnstown Castle, Co. Wexford, Ireland.

Fay, D., McGrath, D., Zhang, C., Carrigg, C., O'Flaherty, V., Carton, O.T. and Grennan, E. (2007). Towards a national soil database. Final report for EPA project 2001-CD/S2-M2.

Fealy, R.M., Green, S., Loftus, M., Meehan, R., Radford, T., Cronin, C. and Bulfin, M. (2009) Teagasc EPA Soil and Subsoils Mapping Project – Final Report, Volume I. Teagasc, Dublin, UK.

Flessa, H., Ruser, R., Dorsch, P., Kamp, T., Jiminez, M., Munch, J. and Beese, F. (2002) Integrated evaluation of greenhouse gas emissions (carbon dioxide, methane, nitrous oxide) from two farming systems in southern Germany. Agriculture, Ecosystems and Environment, 91, 175-189.

Forbes, T.J., Dibb, C., Green, J.O., Hopkins, A. and Peel, S. (1980) Factors affecting the productivity of grassland. The Grassland Research Institute and Agricultural Development and Advisory Service, Joint Permanent Pasture Group, Maidenhead, Berkshire, 141pp.

Foy, R.H., Tunney, H., Carroll, M.J., Byrne, E., Gately, T., Bailey, J.S. and Lennox, S.D. (1997). A comparison of Olsen and Morgan soil phosphorus test results from the cross-border region of Ireland. Irish Journal of Agricultural and Food Research, 36, 185-193.

Galvin, L.F. (1986) Aspects of land drainage development in Ireland over the last twenty-five years. Country Papers 1, Proceedings, Symposium 25th International Course on Land Drainage. 131-147.

Gifford, G.F. and Hawkins, R.H. (1978) Hydrologic impact of grazing on infiltration: a critical review. Water Resources Research, 14, 305-313.

Gooday, R., Anthony, S., and Fawcett, L. (2008) A field scale model of soil drainage and nitrate leaching for application in Nitrate Vulnerable Zones. Environment Modelling and Software, 23, 8, 1045-1055.

Gooday, R., Anthony, S., Calrow, L., Harris D. and Skirvin, D. (2016) Predicitng and Understanding the Effectiveness of Measures to Mitigate Rural Diffuse Pollution. SNIFFER Project DP1, ADAS UK Ltd, Wolverhampton, UK.

Government of Ireland (2014) European Union (Good Agricultural Practice for Protection of Waters) Regulations 2014. Government Buildings, upper Merrion Street, Dublin 2, Ireland.

Green, W.H. and Ampt, G.A. (1911) Studies on soil physics, 1: The flow of air and water through soils. Journal of Agricultural Science, 4, 1-24.

Hansen, S. (2009) Effect of soil compaction on nitrous oxide emission from a soil fertilised with mineral fertiliser or cattle slurry. IOP Conference Series, Earth and Environmental Science, 6, 3pp.

Hansen, S. Maehlum, J. and Bakken, L. (1993) Nitrous oxide and methane fluxes in soil influenced by fertilisation and tractor traffic. Soil Biology and Biochemistry, 25, 621-630.

Haygarth, P.M., Hepworth, L. and Jarvis, S.C. (1998) Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. European Journal of Soil Science 49, 65-72.

Heathwaite, L. (1995) Sources of eutrophication: hydrological pathways of catchment nutrient export. International Association of Hydrological Sciences, Publication No. 230, 161-175.

Hennessy, T., Buckley, C., Cushion, M., Kinsella, A. and Moran, B. (2011) National Farm Survey of Manure Applications and Storage Practices on Irish Farms. Teagasc, Oak Park, Carlow, Ireland.

IPCC (2000) IPCC Grood Practice Guidance and Uncertainity Management in National Greenhouse Gas Inventories, Chapter 4 Agriculture. Institute of Global Environmental Strategies (IGES), on behalf of the Intergovernmental Panel on Climate Change (IPCC), Hayama, Japan.

IPCC (2006) 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 4 Agriculture, Forestry and Other Land Use, Chapter 10 Emissions from livestock and Manure Management. Institute of Global Environmental Strategies (IGES), on behalf of the Intergovernmental Panel on Climate Change (IPCC), Hayama, Japan.

Irish National Soils Map, 1:250,000k, V1b(2014). Teagasc, Cranfield University. Jointly funded by the EPA STRIVE Research Programme 2007-2013 and Teagasc.

Jordan, P., Melland, A.R., Mellander, P.-E., Shortle, G. and Wall, D. (2012). The seasonaility of phosphorus transfers from land to water: implications for trophic impacts and policy evaluation. Science of the Total Environment, 434, 101-109.

Kroulik, M., Kumhala, F. Hula, J. and Honzik, I. (2009) The evaluation of agricultural machines field trafficking intensity for different soil tillage technologies. Soil and Tillage Research, 105, 171-175.

Laws, J.A. and Chadwick, D.R. (2005) The impact of dairy herd intensification on manure management. Final report for the Milk Development Council. A review of current and impending legislation and current management practices regarding livestock manures arising on dairy farms in England, Scotland and Wales, with reference to herd intensification. 64pp.

Le Mer, J. and Roger, P. (2001) Production, oxidation, emission and consumption of methane by soils: a review. European Journal of Soil Biology, 37, 25-50.

Lennox, S.D., Foy, R.H., Smith, R.V. and Jordan, C. (1997) Estimating the contribution from agriculture to the phosphorus load in surface waters. In: Phosphorus Loss from Soil to Water, H. Tunney, O.T. Carton, P.C. Brookes and A.E. Johnston (eds), CAB International, Wallingford, 55-75pp.

Lewis, C., Rafique, R., Foley, N., Leahy, P., Morgan, G., Albertson, J., Kumar, S. and Kiely, G. (2013) Seasonal exports of phosphorus from intensively fertilised nested grassland catchment. Journal of Environmental Sciences, 25, 1-11.

Li, Y., Tullberg, J. and Freebairn, D. (2007) Wheel traffic and tillage effects on runoff and crop yield. Soil and Tillage Research, 97, 282-292.

Lord, E.A. (1992) Modelling nitrate leaching: Nitrate Sensitive Areas. Aspects of Applied Biology, 30, 19-28.

Lord, E.A. and Anthony, S. (2000) MAGPIE: A modelling framework for evaluating nitrate losses at national and catchment scales. Soil Use and Management, 16, pp. 167-174.

Matthews, R., Chadwick, D., Retter, A., Blackwell, M. and Yamulki, S. (2010) Nitrous oxide emissions from small scale farmland features of United Kingdom livestock farming systems. Agriculture, Ecosystems and Environment, 136, 192-198.

McGinnity P., De Eyto E., Gilbey J., Gargam P., Roche W., Stafford T., McGarrigle M., Ó'Maoilédigh N. and Mills P. (2012). A predictive model for estimating river habitat area using GIS-derived catchment and river variables. Fisheries Management and Ecology, 19, 69-77.

McGuckin, S.O., Jordan, C. and Smith, R.V. (1999) Deriving phosphorus export coefficients for corine land cover types. Water Science and Technology, 39, 12, 47-53.

McHugh, M., Wood, G., Walling, D., Morgan, R., Zhang, Y., Anthony, S. and Hutchins, M. (2002) Prediction of sediment delivery to water courses from land: Phase II. Environment Agency Report No. P2-209.

Mellander, P.-E., Melland, A.R., Murphy, P.N.C., Wall, D.P., Shortle, G. and Jordan, P. (2014). Coupling of surface water and groundwater nitrate-N dynamics in two permeable agricultural catchments. Journal of Agricultural Science, 152, S107-S124.

Mockler, E.M., Deakin, J., Archbold, M., Gill, L., Daly, D. and Bruen, M. (2017) Sources of nitrogen and phosphorus emissions to Irish rivers and coastal waters: Estimates from a nutrient load apportionment framework. Science of Total Environment, 601-602, 326-339.

Monteny, G., Bannink, A., and Chadwick, D. (2006) Greenhouse gas abatement strategies for animal husbandry. Agriculture, Ecosystems and Environment, 112, 163-170.

NASA (2009). Draft National Annex to I.S. EN 1991-1-4:2005, Eurocode 1 – Actions on structures – Part 1-4: General actions – Wind actions. National Aeronautics and Space Administration, Washington, USA.

NASA LP DAAC and METI (2011). ASTER Global Digital Elevation Model (GDEM), Version 2. Available from: http://www.noaa.gov/ [Accessed 4 May 2016]

Nemes, A., Wösten, J.H.M., Lilly, A. and Oude Voshaar, J.H. (1999) Evaluation of different procedures to interpolate particle-size distributions to achieve compatibility within soil databases. Geoderma, 90, 187-202.

Newell-Price, P., Smith, K. and Williams, J. (2016) Review of evidence on the principles of crop nutrient management and nutrition for grass and forage crops: Research Review No. 3110149017, AHDB, Stoneleigh, UK.

Ní Longphuirt, S., Mockler, E.M., O'Boyle, S., Wynne, C. and Stengel, D.B. (2016) Linking change in nutrient source load to estuarine responses: an Irish perspective. Biology and Environment: Proceeding of the Royal Irish Academy, 116, 3, 295-311.

Nicholson, F., Rollett, D and Chambers, B. (2011) Review of pollutant losses from solid manure stored in temporary field heaps. Defra project WT1001. Final Report, 56pp.

O'Boyle, S., Quinn, R., Dunne, N., Mockler, E.M. and Ní Longphuirt, S. (2016) What have we earned from over two decades of monitoring riverine nutrients inputs to Ireland's marine environment. Biology and Environment: Proceeding of the Royal Irish Academy, 116, 3, 313-327.

O'Mara, F. (2006) Climate Change – Development of Emission Factors for the Irish Cattle Herd. 2000-LS-5.1.1–M1 special report. Environmental Protection Agency, Johnstown Castle, Co. Wexford, Ireland.

Oenema, O., Velthof, G., Yamulki, S. and Jarvis, S. (1997) Nitrous oxide emissions from grazed grassland. Soil Use and Management, 13, 288-295.

Owens, P.N., Walling, D.E., He, Q. and Shanahan, J. (1997) The use of caesium-137 measurements to establish a sediment budget for the Start catchment, Devon, UK. Hydrological Science Journal, 42, 405-423.

Parkinson, R. (2000) Enhancing the effective utilisation of animal manures on-farm through compost technology. Final project report, MAFF project code WA0519. Ministry of Agriculture, Fisheries and Food, Newton Abbot, UK.

Prosser, I.P. and Rustomji, P. (2000) Sediment transport capacity relations for overland flow. Progress in Physical Geography, 24, 2, 179-193.

R Core Team (2016). R: A language and environment for statistical computing. R Foundation for Statistical Computing. Available from: https://www.R-project.org/

Reidy, B., Simo, I., Sills, P. and Creamer, R.E. (2016) Pedotransfer function for Irish Soils – estimation of bulk density (ρ_b) per horizon type. Soil, 2, 25-39.

Robinson, D. and Naghizadeh, R. (1992) The impact of cultivation practice and wheelings on runoff generation and soil erosion on the South Downs: some experimental results using simulated rainfall. Soil Use and Management, 8, 4, 151-156.

Russell, M.A., Walling, D.E. and Hodgkinson, R.H. (2001) Suspended sediment sources in two small lowland agricultural catchments in the UK. Journal of Hydrology 252, 1-24.

Schneider M.K. (2007) Logical Decision Tree to Classify Soil Type Units in the SGDBE into HOST Classes Calculation Rules. In Schneider, M. K., Brunner, F., Hollis, J. M., and Stamm, C. (2007) Towards a hydrological classification of European soils: preliminary test of its predictive power for the base flow index using river discharge data. Hydrology and Earth System Science, 11, 1501-1513.

Scholefield, D., Lockyer, D., Whitehead, D. and Tyson, K. (1991) A model to predict transformations and losses of nitrogen in UK pastures grazed by beef cattle. Plant and Soil, 132, 165-177.

Sheriff, S.C., Rowan, J.S., Melland, A.R., Jordan, P., Fenton, and Ó hUallacháin, D. (2015) Investigating suspended sediment dynamics in contrasting agricultural catchments using ex situ turbidity-based suspended sediment monitoring. Hydrology and Earth System Science, 19, 3349-3363.

Shreve R. (1974) Variation of mainstream length with basin area in river networks. Water Resources Research, 10, 1167-1177.

Shore, M., Murphy, S., Mellander, P.-E., Shortle, G., Melland, A.R., Crockford, L., O'Flaherty, V., Williams, L., Morgan, G. and Jordan, P. (2017). Influence of stormflow and baseflow phosphorus on stream ecology in agricultural catchments. Science of the Total Environment, 590-591, 469-483.

Shortle, G. and Jordan, P. (eds) (2017). Agricultrual Catchments Programme Phase 2 Rpeort. Teagasc, 145pp.

Silgram, M., Jackson, B., Quinton, J., Stevens, C. and Bailey, A. (2007) Can tramline management be an effective tool for mitigating phosphorus and sediment loss? Proceedings of the 5th International Phosphorus Transfer Workshop (IPW5), Heckrath G, Rubaek G, Kronvang B (eds), 3–7 September, Silkeborg, Denmark; pp 287–290.

Simo, I., Creamer, R.E., O'Sullivan, L., Reidy, B., Schulte, R.P.O. and Fealy, R.M. (2014) Irish Soil Information System: Soil Property maps. Final Technical Report 18, EPA STRIVE Programme 2007-2013. Teagasc, Johnstown Castle Environment Research Centre, Wexford, Ireland.

Sitaula, B., Hansen, S., Sitaula, J. and Bakken, L. (2000) Methane oxidation potentials and fluxes in agricultural soil, effects of fertilisation and soil compaction. Biogeochemistry, 48, 323-339.

Smith, K. and Frost, J.P. (2000) Nitrogen excretion by farm livestock with respect to land spreading requirements and controlling nitrogen losses to ground and surface waters: I. Cattle and sheep. Bioresource Technology, 71, 173-181.

Smith, K., Brewer, A., Crabb, J. and Dauven, A. (2001) A survey of the production and use of animal manures in England and Wales. II. Poultry manure. Soil Use and Management, 17 (1), 48-56.

Smith, K., Brewer, A., Crabb, J. and Dauven, A. (2001) A survey of the production and use of animal manures in England and Wales. III. Cattle manures. Soil Use and Management, 17 (2), 77-87.

Smith, P. and Smith, J. (2004) Review of the contributions to climate change (though greenhouse gas emissions) of fertiliser use on different soil types and through different application methods. Scottish Executive project ABRG: UEH/007/03, Final Report, 99pp.

Stromqvist, J., Collins, A., Davison, P. and Lord, E. (2008) PSYCHIC – a process based model of phosphorus and sediment transfers within agricultural catchments. Part 2 – A preliminary evaluation. Journal of Hydrology, 350, 303-316.

Teagasc (2017) Draft version- A Survey of Fertiliser Use in Ireland from 2005-2015 for Grassland and Arable Crops. Teagasc, Johnstown Castle Environment Research Centre, Wexford, Ireland.

Tóth, G., Jones, A. and Montanarella, l. (2013) LUCAS Topsoil Survey: Methodology, data and results. European Commission: Joint Research Centre: Institute for Environment and Sustainability, Ispra, Italy.

Trimble, S.W. and Mendel, A.C. (1995) The cow as a geomorphological agent. Geomorphology, 13, 233-253.

Ulén, B., Bechmann, M., Fölster, J., Jarvie, H.P. and Tunney, H. (2007) Agriculture as a phosphorus source for eutrophication in the north-west European countries, Norway, Sweden, United Kingdom and Ireland: a review. British Society of Soil Science, 2, 5-15.

Van Groenigen, J., Kuikman, P., de Groot, W., and Velthof, G. (2005) Nitrous oxide emission from urine treated soil as influence by urine composition and physical conditions. Soil Biology and Biochemistry, 37, 463-473.

Walling, D.E. and Zhang, Y. (2004) Predicting slope channel connectivity: a national scale approach. In Sediment Transfer through the Fluvial System. Proc. Moscow Sympos.. 107-114, IAHS Publication No. 288, IAHS Press, Wallingford.

Walsh S. (2012). A Summary of Climate Averages 1981-2010 for Ireland, Climatological Note No.14. Met Éireann, Dublin.

Webb, J. and Misselbrook, T.H. (2004) A mass-flow model of ammonia emissions from UK livestock production. Atmospheric Environment, 38, 2163-2176.

Webb, J., Misselbrook, T.H., Pain, B., Crabb, J. and Ellis, S. (2001) An estimate of the contribution of outdoor concrete yards used by livestock to the United Kingdom inventories of ammonia, nitrous oxide and methane. Atmospheric Environment, 35, 6447-6451.

Withers, P., Royle, S., Tucker, M., Watson, B., Scott, T., Silcock, P., Smith, G. and Dwyer, J. (2003) Field development of grant aid proposals for the control of diffuse agricultural pollution. Environment Agency, Technical Report, No. P2-261/09/TR, 97pp.

Yamulki, S. and Jarvis, S. (2002) Short term effects of tillage and compaction on nitrous oxide, nitric oxide, nitrogen dioxide, methane and carbon dioxide fluxes from grassland. Biology and Fertility of Soils, 36, 224-231.

Zhang, X., Liu, X., Zhang, M., Dalhgren, R. and Eitzel, M. (2010) A review of vegetated buffers and a meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. Journal of Environmental Quality, 39, 1, 76-84.