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Sources of nitrogen and phosphorus emissions to Irish rivers and coastal waters: Estimates from a nutrient load apportionment framework

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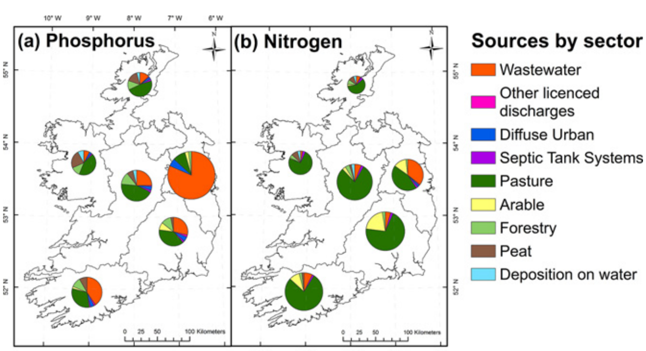
HIGHLIGHTS

- Performance of the SLAM framework in 16 catchments was good for N and satisfactory for P.
- Sources of P contributions vary by land use and hydrogeological characteristics.
- P from pasture is mainly driven by hydrogeological conditions, not pressure.
- Agriculture is the dominant source of N across all regions.
- Mitigation options should reflect local source-pathway-receptor relationships.

GRAPHICAL ABSTRACT



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ABSTRACT

More than half of surface water bodies in Europe are at less than good ecological status according to Water Framework Directive assessments, and diffuse pollution from agriculture remains a major, but not the only, cause of this poor performance. Agri-environmental policy and land management practices have, in many areas, reduced nutrient emissions to water. However, additional measures may be required in Ireland to further decouple the relationship between agricultural productivity and emissions to water, which is of vital importance given on-going agricultural intensification.

The Source Load Apportionment Model (SLAM) framework characterises sources of phosphorus (P) and nitrogen (N) emissions to water at a range of scales from sub-catchment to national. The SLAM synthesises land use and physical characteristics to predict emissions from point (wastewater, industry discharges and septic tank systems) and diffuse sources (agriculture, forestry, etc.). The predicted annual nutrient emissions were assessed against monitoring data for 16 major river catchments covering 50% of the area of Ireland. At national scale, results indicate that total average annual emissions to surface water in Ireland are over 2700 t yr^{-1} of P and 82,000 t yr^{-1} of N. The proportional contributions from individual sources show that the main sources of P are from municipal wastewater treatment plants and agriculture, with wide variations across the country related to local anthropogenic pressures and the hydrogeological setting. Agriculture is the main source of N emissions to water across all regions of Ireland. These policy-relevant results synthesised large amounts of information in order to identify the dominant sources of nutrients at regional and local scales, contributing to the national nutrient risk assessment of Irish water bodies.

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1. Introduction

Identifying appropriate measures to address eutrophication remains a major challenge across Europe, where many river and lake water bodies are still failing to meet Water Framework Directive (WFD; 2000/60/EC) objectives. Indeed the management of diffuse sources of nutrients in rural catchments is of particular interest globally, as environmental objectives compete with the increasing demand for food production. A challenge for current research is to improve understanding of nutrient processes in catchments by scaling up findings from plot scale and extrapolating understanding to entire regions or countries from a relatively small number of well-studied catchments (McGonigle et al., 2012). This scientific evidence can then inform environmental policy through the provision of tools and guidance to support water quality management and decision making (Defra, 2011).

In Ireland, integrated catchment management has been used to implement a systems-based approach for assessing and managing freshwater ecosystems (Daly et al., 2016). Characterisation of water bodies for the second cycle of the WFD has been undertaken including evaluation of physical, hydrochemical and ecological characteristics, and a risk assessment of pressures, pathways and impacts. These assessments used results from water quality models (e.g. Gill and Mockler, 2016; Mockler et al., 2016) to ensure that investigations and measures are efficiently targeted, and decision makers are better informed about the effectiveness of measures to date and the possible response of water bodies to future actions (Ní Longphuirt et al., 2016).

Water mobilises and transports nutrients through the landscape and the attenuation potential varies considerably with hydrological settings and type of nutrient (Archbold et al., 2016). For instance, nitrate is typically delivered to streams via subsurface pathways (Kröger et al., 2007; Tesoriero et al., 2009). The majority of phosphorus from diffuse sources is driven by storm events and delivered via overland flow (Jordan et al., 2005), although significant quantities may also be delivered via tile drainage (Monaghan et al., 2016; Zimmer et al., 2016) and groundwater pathways (Mellander et al., 2016) with individual hot-spots of nutrient loss, or critical source areas, contributing a relatively high proportion of the nutrients exported from the landscape (Pionke et al., 2000). As hydrology is a key driver of nutrient delivery at catchment scale, all relevant hydrological processes should be adequately represented in a water quality model (Medici et al., 2012).

In cases where water quality is impacted by excess nutrients, load apportionment modelling can support the proportional and pragmatic management of water resources. There are two broad approaches to load apportionment modelling, (i) load-orientated approaches which apportion origin based on measured in-stream loads (Grizzetti et al., 2005; Greene et al., 2011; Grizzetti et al., 2012), and (ii) source-orientated approaches where amounts of diffuse emissions are calculated using models typically based on export coefficients from catchments with similar characteristics (MCOS, 2002; Jordan and Smith, 2005; Smith et al., 2005; Campbell and Foy, 2008; Ní Longphuirt et al., 2016). The Source Load Apportionment Model (SLAM) framework (Mockler et al., 2016) takes the latter approach which enables estimates of the relative contribution of sources of nitrogen (N) and phosphorus (P) to surface waters in catchments without in-stream monitoring data. This allows the approach to be applied throughout Ireland, independently of the availability of measured in-stream data. It integrates information on point discharges (urban wastewater, industry and septic tank systems) and diffuse sources (pasture, arable, forestry, etc.) and catchment data, including hydrogeological characteristics where applicable.

Significant changes in the magnitude and sources of phosphorus in Irish rivers have occurred over the last two decades, due to both improved wastewater treatment works and changes in land management practices (O'Boyle et al., 2016), altering the relative contributions from point and diffuse sources (Ní Longphuirt et al., 2016). As regulation of point discharges continues to reduce emissions, other sources of nutrients may start to control water quality in these areas. This study aimed to quantify the sources of P and N emissions in Irish rivers in order to support the identification of potential pressures resulting in eutrophication. The objectives of this paper are to (i) evaluate the performance of the SLAM framework for predicting nutrient loads in Irish rivers by comparing the outputs of the model with measured in-stream loads, (ii) using the SLAM results, identify the main sources of nutrients in Ireland at national, regional and local scales, and (iii) compare and contrast the main sources and delivery pathways of agricultural and wastewater emissions.

2. Data & methods

2.1. Study area

The Republic of Ireland has an area of 70,000 km^2 and a population of approximately 4.6 million people, with the largest urban centres located in Dublin on the east coast, and Cork on the south coast. The land cover is predominantly pasture supporting over 6.4 million cattle. There is a mild maritime climate, with mean annual temperatures of approximately 10 °C. Annual rainfall varies from in excess of 3000 mm in the western mountains to <800 mm along the east coast. Load apportionment results by sector were analysed nationally, for six regional (formerly River Basin) districts, and at a local scale for 583 sub-catchments ranging in area from 24 to 390 km^2.

2.2. Source Load Apportionment Model (SLAM) Framework

The Source Load Apportionment Model (SLAM) Framework (Mockler et al., 2016) incorporates multiple national spatial datasets relating to nutrient emissions to surface water, including land use and physical characteristics of the sub-catchments. Separate modules were developed for each type of nutrient source to facilitate upgrading and comparisons with new data or methods. For example, in the current version (v 2.05) of the framework, two modules have been upgraded with output from more advanced export-coefficient based models; the agriculture (pasture & arable) and septic tank systems modules now use spatial outputs from the Catchment Characterisation Tool (CCT) (Archbold et al., 2016) and SANICOSE models (Gill and Mockler, 2016), respectively.

The key input dataset for the agriculture module (i.e. the CCT) was the Land-Parcel Identification System (LPIS) which was combined with land management data from the Department of Agriculture, Food and the Marine (DAFM). This annual average export coefficient model calculated leaching rates based on methods from existing models for N (Shaffer et al., 1994; del Prado et al., 2006) and P (Heathwaite et al., 2003). In addition, the model applied pathway-dependent attenuation coefficients related to the hydrogeological conditions, which were inferred from GIS maps of relevant properties including soil drainage, subsoil permeability and depth to bedrock (details in Appendix A.4). These coefficients were determined following a literature review and expert elicitation for the two pathway categories grouped into (i) 'near surface' pathways including overland and drain flow, and (ii) groundwater (Archbold et al., 2016). Further details on the CCT are available in Appendix A.4. The 2012 CORINE (Lydon and Smith, 2014) level 3 land cover data were used in the forestry, peatlands and urban sub-models. Various export coefficients (Table A2) were then applied in each of the modules to estimate their annual nutrient emissions to water (see Appendix A). Loads from direct discharges were calculated from data collected by the EPA, including Annual Environmental Reports, the EPA Licensing Enforcement and Monitoring Application (LEMA), and the Pollutant Release and Transfer Register (PRTR) database. The total annual nutrient load at the outlet of each sub-catchment (L_i) was calculated as:

$$L_i = (Point_i + Diffuse_i)^*(1-Lake_i) \quad (1)$$

where, $Point_i$ = sum of nutrient loads discharged from wastewater treatment plants, industrial discharges and emissions from septic tank systems; $Diffuse_i$ = sum of diffuse nutrient losses from agriculture, forestry, peatlands, urban areas and atmospheric deposition; and $Lake_i$ = estimated nutrient lake retention factor.

The SANICOSE model quantifies the annual nutrient contribution for each septic tank or package treatment plant systems for on-site domestic wastewater treatment (commonly referred to as Domestic Wastewater Treatment Systems and referred to in this study as septic tank systems) at their specific locations, as estimated from GIS data on property locations and sewered areas. The model was developed specifically for the SLAM Framework following a synthesis of 15 years of field studies on on-site systems in Ireland across many different soils types (Gill et al., 2007; Gill et al., 2009a; Gill et al., 2009b; Gill et al., 2009c; Sulleabhain et al., 2009; O'Luanaigh et al., 2012; Donohue et al., 2015). The calculation of average annual load combines factors relating to the efficiency of the septic tank systems with attenuation factors for the hydrogeological flow pathways based on the CCT coefficients (Gill and Mockler, 2016).

Details of the development and application of the SLAM framework are available in Mockler et al. (2016), and the framework structure and user interface are described in Mockler (2016). A summary of the models and data sources for each component of Eq. (1) as implemented in SLAM version 2.05 are provided in Appendix A.

2.3. Nutrient monitoring data

The SLAM results were compared with monitoring data to assess the model performance prior to its extension to cover the entire country using the R 'stats' and 'graphics' packages (R Core Team, 2016). Sixteen major river catchments covering 50% of the area of Ireland (Fig. 1 a) were selected based on the availability of both monitoring data and loadings information. These catchments are in the national riverine inputs monitoring programme that is managed by the Irish Environmental Protection Agency (EPA). The design of the programme followed the Comprehensive Study on Riverine Inputs and Direct Discharges (RID) principles (OSPAR, 1998). O'Boyle et al. (2016) provide details of the sampling and analysis methodologies for the total phosphorus (TP) and total nitrogen (TN) concentration data used in this study. The locations of the flow and nutrient concentration monitoring stations (Table 1) are largely at the tidal limits, and they are generally upstream of the large wastewater treatment plant discharges associated with the major coastal urban centres. It was preferable to test the model in areas dominated by diffuse sources of nutrients as load emission estimates from large direct discharges on Irish coasts are based on detailed annual monitoring and reporting and are more reliable than estimates of diffuse emissions.

The loadings information for diffuse and point sources used in this study relate to three recent years, 2012–2014, hence, monitoring data for these years only were used to evaluate the model performance. In most catchments, the wettest year of this period was 2014, with 2013

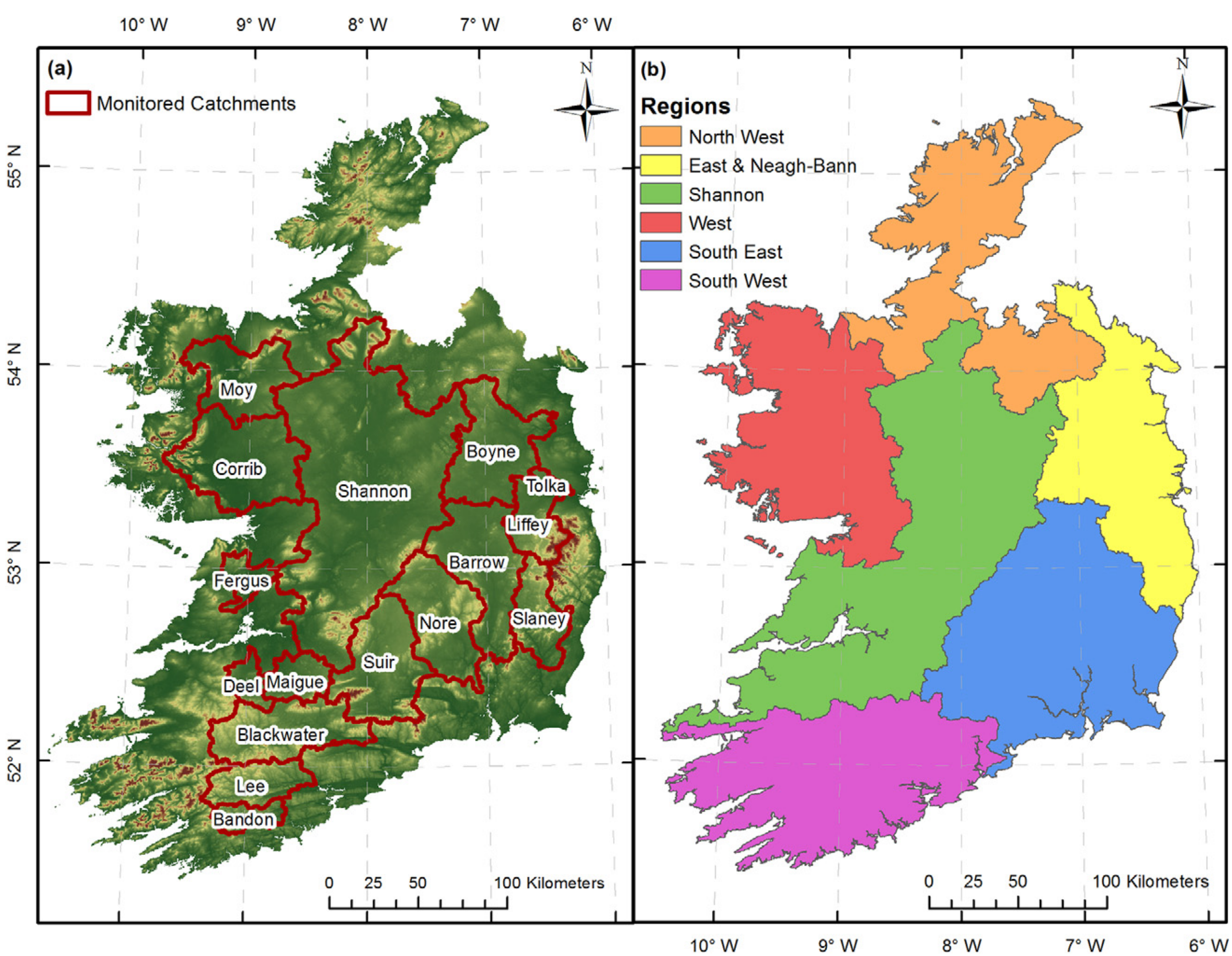


Fig. 1. Locations of (a) 16 catchments with monitored TN and TP emissions to water and (b) catchment regions based on previously defined River Basin Districts.

Table 1
Chemistry and hydrometric monitoring station codes and catchment areas.

River	Nutrient monitoring stations	Area (km^2)	Hydro stations	Area (km^2)
Bandon	20B02-0900	515	20002	424
Barrow	14B01-3500	2804	14029	2778
Blackwater	18B02-2600	2384	18002	2334
Boyne	07B04-2200	2624	07012	2408
Corrib	30C02-0460	3103	30061	3136
Deel	24D02-1450	489	24029	486
Fergus	27F01-0700	554	27002	511
Lee	19L03-0700	1177	19013	796
Liffey	09L01-2350	1119	09022	848
Maigue	24M01-0900	807	24008	806
Moy	34M02-1100	1949	34001	1975
Nore	15N01-2400	2427	15006	2418
Shannon[a]	25S01-2600/25S01-2900	11619	25055/ 25001/ 25075	10817/646/ 10782
Slaney	12S02-2350	1321	12,001	1031
Suir	16S02-2700	2629	16,011/16009[b]	2144/1583[b]
Tolka	09T01-1150	149	09037	138

[a] Flow monitoring at Ardnacrusha, Annacotty, and Shannon tailrace at Parteen Weir.
[b] Upstream flow monitoring station 16,009 (Suir2) on the Suir River used where data was unavailable at station no. 16011.

being the driest (Fig. 2). The long-term annual flows were within the ranges of annual flows for these three years in all of the study catchments except for the Bandon, which has had a notable increase in flows in recent years possibly due to issues with changes in the rating at this station (Fig. 2). As flow data were not available for the Corrib catchment for the study period, the 1990–2011 long term average (104 m^3 s^{-1}) was assumed for each year.

Annual loads were calculated as the product of the flow-weighted annual mean concentration of TP or TN and the annual flow (see O'Boyle et al., 2016). Monitoring of flow and nutrients occurred at different locations along the rivers, hence, in order to calculate the loads the flow values were scaled by the ratio of the areas draining to the nutrient and flow monitoring stations. There was <20% difference in monitored area for all catchments except for the Slaney and Suir (Table 1).

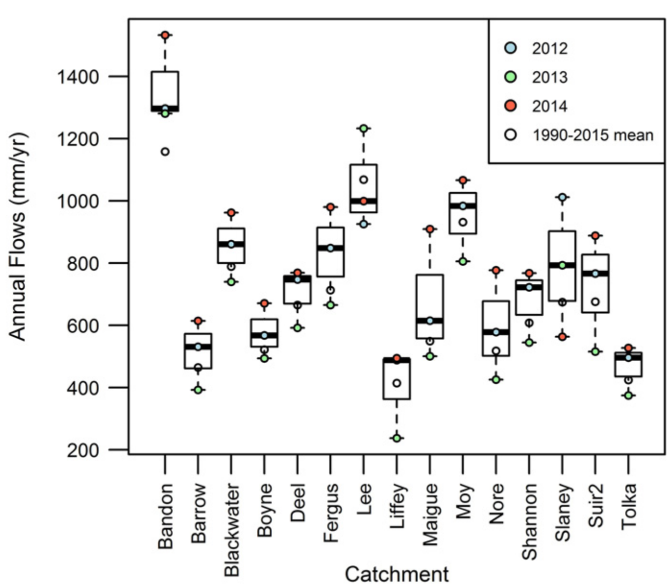


Fig. 2. Boxplots of area normalised annual flow (mm/year) for the study period, with annual values indicated for 2012 (blue), 2013 (green), 2014 (red), and the 1990–2015 long-term mean flow (black). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

3. Results and discussion

3.1. Comparison with monitoring data

Prior to analysing the load apportionment by sector, the performance of the SLAM in predicting the total annual loads was evaluated by comparison with monitoring data. The SLAM results provide a reasonable representation of the variance of the 3-year annual average loads, with satisfactory coefficient of determination (r^2) for TP ($r^2 = 0.78$, $p \leq 0.01$, $n = 16$) and TN ($r^2 = 0.82$, $p \leq 0.01$, $n = 16$). The error bars highlight the interannual variability in measured emissions which are not captured by the model. Fig. 3 c and d indicate the individual results for each test catchment. The model tends to underestimate P loads, particularly in the Bandon, Lee and Blackwater catchments, i.e. primarily in areas in the south and south-west of Ireland where iron-rich soils facilitate P mobilisation (Mellander et al., 2016) which is not yet captured by the agriculture module (CCT). The model also underestimates N loads, particularly in the Slaney, Barrow and Blackwater. Some underestimation was expected as each sector model only includes major sources of nutrients and does not incorporate incidental losses that may occur throughout the year.

3.2. National overview

3.2.1. Main sources of phosphorus in Irish rivers

The average annual P emissions to surface water in Ireland were estimated at over 2700 t yr^{-1}. Source contributions varied by region (Fig. 3 a, Table 2), with municipal wastewater discharges and pasture dominant overall. The clear dominance of wastewater in the East region (78%) reflects the distribution of the population, which is clustered around the capital city, Dublin. Pasture is the dominant diffuse source of P, contributing up to 47% at regional level. Forestry and peatlands were estimated to contribute up to 13% and 23%, respectively, of P emissions in some regions, whereas contributions from septic tank systems and other licenced discharges (i.e. industrial discharges) were low across all regions, representing <3% of P emissions.

3.2.2. Main sources of nitrogen in Irish rivers

Agriculture is the main source of N, as seen in regions of the United Kingdom (Bowes et al., 2014) and across Europe (Bøgestrand et al., 2005). In Ireland, average annual N emissions to water were estimated at over 82,000 t yr^{-1}, with pasture being the dominant source overall (Fig. 4 b, Table 3). The sources were more spatially consistent compared to P emissions, as nitrate from diffuse sources is typically delivered to streams via subsurface pathways, with links between increasing nitrate concentrations and groundwater contributions (Tesoriero et al., 2009; Tedd et al., 2014). Emissions from arable land reflected the locations of the most crop-intensive areas in the more freely draining soils of the country in the East (14%) and South-East (20%) where there is less rainfall. The proportion of emissions of N from septic tank systems was low on average (2%), with higher contributions in the North-West (5%) and West regions (4%), reflecting the relatively high density of non-sewered properties in these areas. Contributions from wastewater were relatively low across all regions (≤7%) except for the East (33%), the latter due to the high proportion of the population living in this region.

3.2.3. Point vs. diffuse sources at local scale

Fig. 5 a & b illustrate the range of nutrient export rates of P and N emissions to water for each of the 583 sub-catchments in Ireland and the percentage contributions from point sources (Fig. 5 c & d). Point sources of nutrients were classified as wastewater, other licenced discharges, and septic tank systems. Farmyards point sources are also likely to be contributing but are not included in this version of the framework (although they are somewhat implicit in the CCT). The locations of urban areas and the associated wastewater and industrial discharges

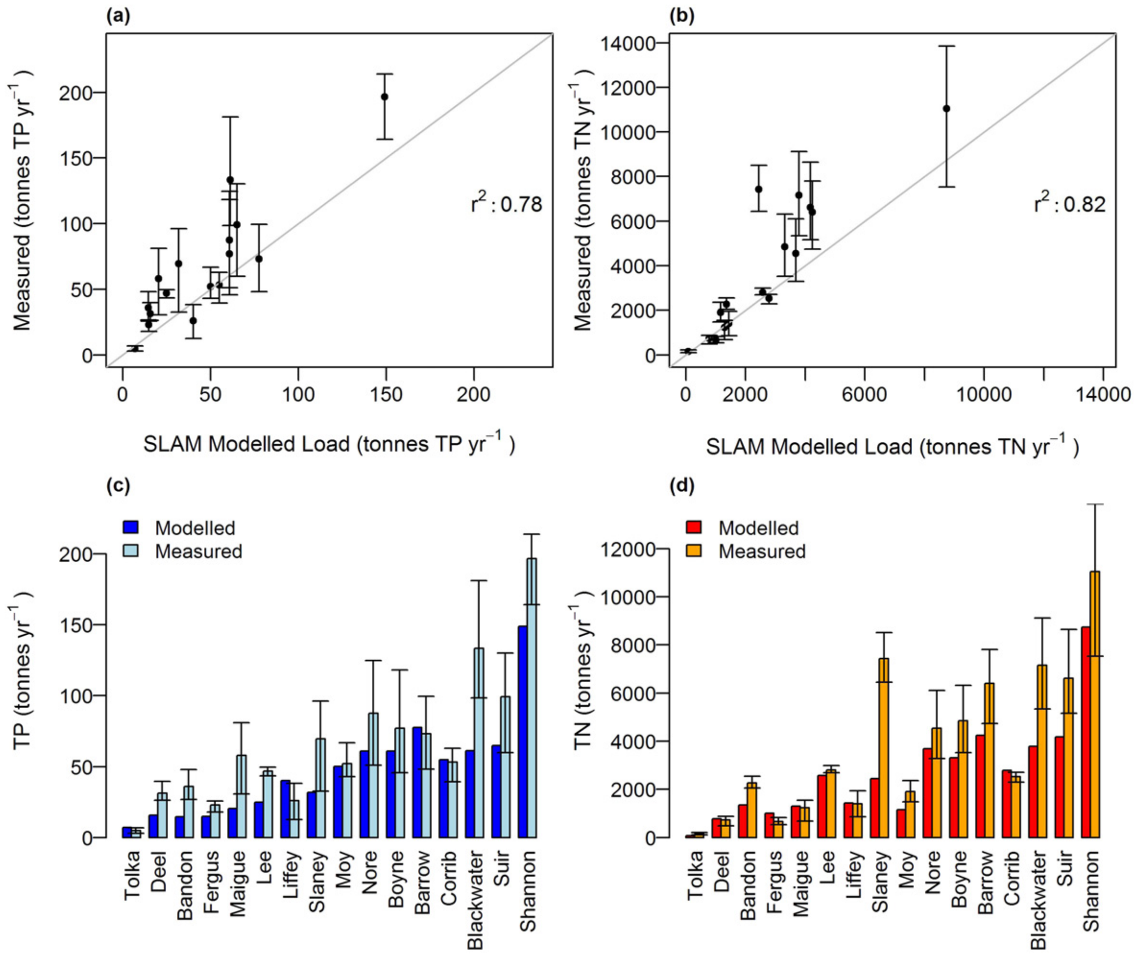


Fig. 3. Modelled SLAM nutrient emissions of (a) TP and (b) TN compared to average measured annual nutrient fluxes from 2012 to 2014 (error bars show the range of measured loads). Bar charts of modelled and measured (c) TP and (d) TN for each catchment (ordered by increasing catchment area).

drive the highest emission rates of P (Fig. 5 a & c). Mid-ranging phosphorus export rates (0.25–0.5 kg ha^{-1} yr^{-1}) typically coincide with agricultural lands with poorly draining soils. However, for nitrogen, agricultural intensity has a dominant impact on the magnitude of the total emission rates, with the majority of emissions coming from the East and South of the country, reflecting the coincidence of higher intensity agricultural land on more freely draining soils (Fig. 5 b & f).

3.2.4. Emerging pressures

Phosphate dosing of raw water supplies began for the first time on a trial basis in the south west of Ireland in November 2016 to control plumbosolvency in mains water. This has the potential for significant contributions to P concentrations in surfaces waters due to mains water leakage, as has been seen in other countries where the practice has been widespread for decades (Gooddy et al., 2017), and will be included in future iterations of this P load apportionment research.

3.3. Comparisons of sources and pathways

3.3.1. Impact of pressure vs. pathway on emissions from pasture

In order to explore the main drivers of nutrient emissions from pasture, relationships between the modelled P and N losses (kg ha^{-1} yr^{-1}) and indicators of pressure (livestock units per hectare) and delivery pathway (percentage area of poorly drained soils) in the 583 sub-catchments were examined in an R 'pairs plot' (R Core Team, 2016). The results (Fig. 6) show that:

- There is a strong inverse relationship between the modelled P and N load emissions from pasture in the 583 sub-catchments ($r^2 = 0.71$, $p \leq 0.001$). These contrasting delivery mechanisms in the conceptual models of P and N for diffuse agriculture reflect the scientific understanding of key nutrient delivery mechanisms present in Irish conditions (as outlined in Section 1).

Table 2
Phosphorus (P) emissions to water by region and percentage contribution from sources.

Region	Area (km^2)	P total (t yr^{-1})	Wastewater	Other licenced discharges	Diffuse urban	Septic tank systems	Pasture	Arable	Forestry	Peatlands	Deposition on water
North West	9842	264	13%	1%	4%	3%	47%	1%	12%	15%	4%
East (& Neagh-Bann)	8458	1015	78%	0%	7%	1%	8%	2%	2%	1%	0%
South East	12,850	378	28%	2%	7%	2%	38%	8%	12%	3%	0%
South West	11,181	416	39%	2%	5%	1%	31%	2%	11%	8%	1%
Shannon	18,014	422	24%	1%	5%	2%	43%	1%	13%	6%	4%
West	10,458	271	8%	0%	4%	2%	43%	0%	13%	23%	8%
Total	70,803	2766	42%	1%	6%	2%	29%	2%	9%	7%	2%

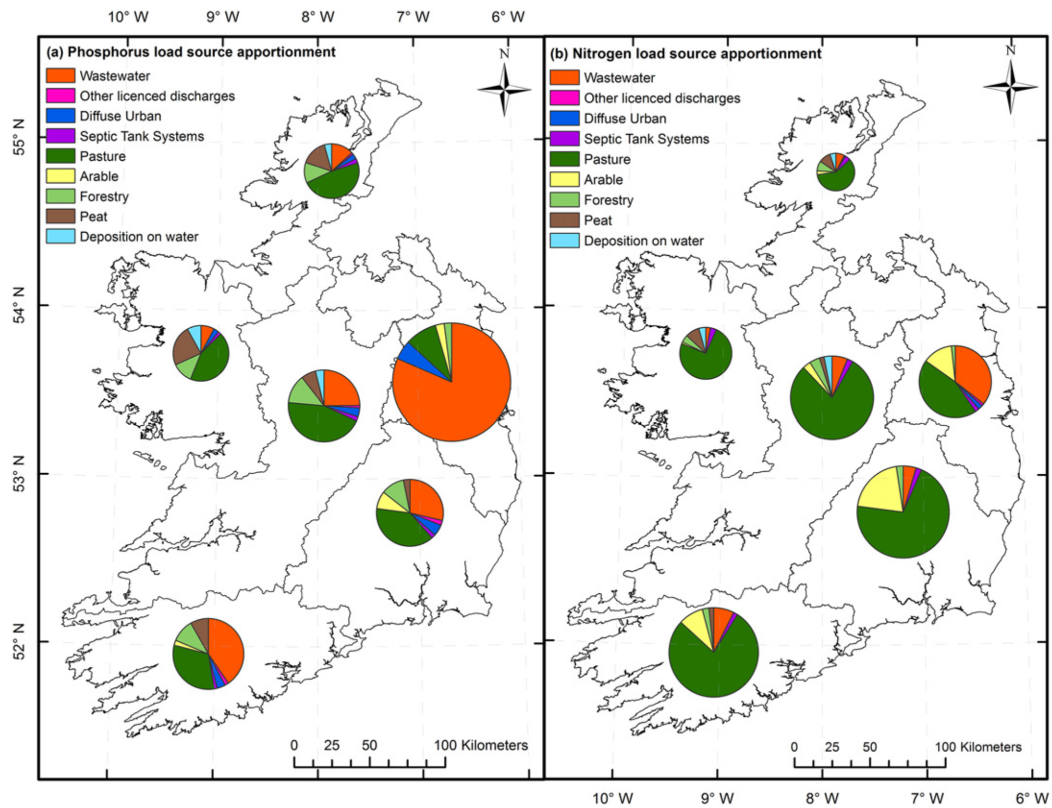


Fig. 4. Load apportionment of (a) phosphorus and (b) nitrogen emissions to water by region. The size of the pie indicates the relative total nutrient emissions.

- Phosphorus emissions from pasture increase with the percentage area of poorly drained soils ($r^2 = 0.87$, p ≤0.001), whereas N emissions decrease ($r^2 = 0.65$, p ≤0.001).
- There is further contrast between P and N source pressures and the consequent emissions, as there is a strong increasing relationship between N emissions from pasture and livestock units ($r^2 = 0.78$, p ≤0.001) whereas the relationship with P is weakly negatively correlated ($r^2 = 0.43$, p ≤0.001).

These relationships emphasises that P emissions from pasture are not driven by the magnitude of the pressure (indicated by the livestock units), as hydrogeological conditions have an overriding impact on transport and attenuation (Mellander et al., 2015). In fact, the majority of diffuse P emissions most likely come from a relatively small proportion (critical source areas) of these poorly drained soils (Doody et al., 2012). This highlights that in order to be effective from a water quality perspective, tailored N and P mitigation options are required that take into account the local source-pathway-receptor relationships. For example, measures for reducing N emissions typically need to focus on managing source loads, as interrupting the pathway is challenging (Deakin et al., 2016).

3.3.2. Municipal wastewater treatment vs. septic tank systems

Municipal wastewater treatment in Ireland has improved significantly in recent years driven in part by European legislation, including the Urban Waste Water Directive (91/271/EEC). Phosphorus emissions from wastewater have decreased by 40% from the reported emissions from wastewater in 1999 (OSPAR, 2000), over which time there has been an increase in population of 0.8 million (21%). The majority of the population, over 3 million people, are connected to over 1000 wastewater treatment plants across the country. Industrial discharges are also connected in addition to these domestic discharges, resulting in a total population equivalent (p.e.) of 4.6 million being treated annually. The average treatment efficiency rate is approximately 62% and 56% for P and N, respectively.

The remainder of the population in Ireland, over 1.4 million people, use decentralised septic tank or package treatment plant systems for domestic wastewater treatment (referred to in this study as septic tank systems), and this number is rising with a 20% increase recorded in the nine year period from 2002 to 2011 (Central Statistics Office, 2013). These 500,000+ individual systems generate over 950 t yr^{-1} of P and 5800 t yr^{-1} of N, respectively, assuming 1.9 kg P yr^{-1} and 11.62 kg yr^{-1} N per household. Due to the wide distribution of these

Table 3
Nitrogen (N) emissions to water by region and percentage contribution from sources.

Region	Area (km2)	N total (t yr^{-1})	Wastewater	Other licenced discharges	Diffuse urban	Septic tank systems	Pasture	Arable	Forestry	Peatlands	Deposition on water
North West	9842	4286	7%	1%	1%	5%	59%	3%	9%	10%	5%
East (& Neagh-Bann)	8458	13,000	33%	1%	2%	2%	45%	14%	2%	1%	1%
South East	12,850	20,594	4%	1%	1%	2%	69%	20%	2%	1%	0%
South West	11,181	19,551	7%	0%	0%	2%	77%	9%	2%	2%	0%
Shannon	18,014	17,171	6%	1%	1%	2%	78%	3%	4%	2%	3%
West	10,458	7589	3%	0%	1%	4%	74%	1%	5%	9%	4%
Total	70,803	82,190	10%	1%	1%	2%	69%	10%	3%	2%	2%

systems they can be described as a diffuse source, however, they are treated as small point sources in this study as this relates to how potential mitigation measures are applied. The SANICOSE model (Gill and Mockler, 2016) was implemented nationally, producing estimated nutrient emissions from each septic tank system which were fed into the SLAM framework. The average treatment efficiency rates of these systems were 95% and 67% for P and N, respectively. Hence, on average, the N treatment efficiency rate is comparable to that from wastewater treatment plants, whereas the P treatment rate by septic tank systems is superior. The average sub-catchment contributions were 2% and 3% for N and P, respectively. However, the treatment efficiency from septic tank systems is widely varying, and underperforming systems have the potential to cause impact in small streams, particularly in areas of poor permeability during low flow periods, where there is little baseflow for dilution (Archbold et al., 2010). In areas with a high density of septic tank systems (over 19/km^2), annual nutrient emissions contributions can be up to 22% for P and 13% for N (Gill and Mockler, 2016). Advancements in techniques to identify the contributions of pollution from human sources (e.g. Richards et al., 2017) and their application in Irish catchments (Bedri et al., 2016; Flynn et al., 2016) will provide valuable data to validate and refine these source apportionment estimates.

3.3.3. Wastewater treatment plants vs. combined sewer overflows

There are over 1500 combined sewer overflows (CSOs) in Ireland that convey wastewater and stormwater generated from urban areas drained by combined sewer systems. In the SLAM, emissions from the wastewater module include discharges from treatment plants and CSOs. This uses annual values supplied in the 2014 Annual Environmental Reports, where available, to estimate emissions from CSOs, and otherwise assumes a proportion of the collected load from the agglomeration (see Appendix A). The SLAM results indicate CSOs account for 5% of the total P emissions in Ireland (12% of the wastewater emissions) and 1% of N emissions (10% of the wastewater emissions). However, there are significant uncertainties surrounding these estimates as the actual losses are dependent on complex network-specific characteristics including precipitation patterns, the connected impervious area, the rainfall-runoff rate, and the storage volume in the sewer network. Studies of emissions from individual storm events using detailed monitoring and hydrodynamic modelling (e.g. Quijano et al., 2017) are required to fully characterise the complex discharges from CSOs in agglomerations and resulting water quality impacts. Following from a recent review of low-cost monitoring technologies use in Irish sewer networks (Morgan et al., 2017), further research on monitoring and treating CSO overflows in Irish sewer networks is planned which could provide network-specific data to improve this model.

3.3.4. Groundwater vs. surface water pathways

On average, the groundwater pathway contributed approximately 5% of P and 25% of N to surface waters, increasing up to 70% of P and 80% of N in some hydrogeologically susceptible sub-catchments. This is based on the SLAM modules for agriculture (pasture and arable) and septic tank systems, which calculated the emissions to surface waters through two pathway categories; (i) 'near surface' pathways including overland and drain flow, and (ii) groundwater. By including these pathways in the conceptual models of nutrient transport and attenuation, the three-dimensional relationships of sources, pathways and receptors in catchments are emphasised.

To achieve successful outcomes, water quality management strategies must be tailored to the local hydrogeological conditions and main pollutant pathways. Deakin et al. (2016), outlined two contrasting examples, showing that in a freely draining karstified catchment with predominantly subsurface pathways, measures must target managing inputs to groundwater, whereas in a catchment underlain by poorly draining soils, the transport of P via overland flow and interflow, and from small point sources, were key issues and hence measures were required to intercept pathways and mitigate discharges.

3.4. Model performance and uncertainties

Inconsistencies between individual SLAM results and measured instream loads (Fig. 3) may be due to several sources including input data issues, unknown sources not accounted for in the model e.g. accidents or non-compliant farmyards, or inadequate monitoring data. Uncertainties in a model-based study, such as this, can arise in (i) model context, (ii) model structure, (iii) input (forcing) data and (iv) identification of parameter values (Walker et al., 2003). Uncertainties have not been quantified in this study, however, each source is discussed qualitatively in this section.

(i) Model context

It is important that the context in which the model results are used including assumptions and boundary conditions is justifiable, ensuring that the model is not applied outside of its intended scope. The SLAM predicts the annual emissions of P and N to water, and does not include inter- or intra-annual variability. Hence, changes due to hydrological variability and groundwater lag times are not explicitly captured. This is in contrast to statistical load apportionment modelling which develops functional relationships between nutrient concentrations and discharge monitoring data (e.g. Greene et al., 2011). Therefore the SLAM results are not directly suitable for assessing pressures in these cases e.g. identifying sources of high nutrient concentrations at low flows. However, the chosen modelling approach does facilitate predictions in ungauged or poorly gauged catchments, facilitating the range of spatial assessment produced in this study.

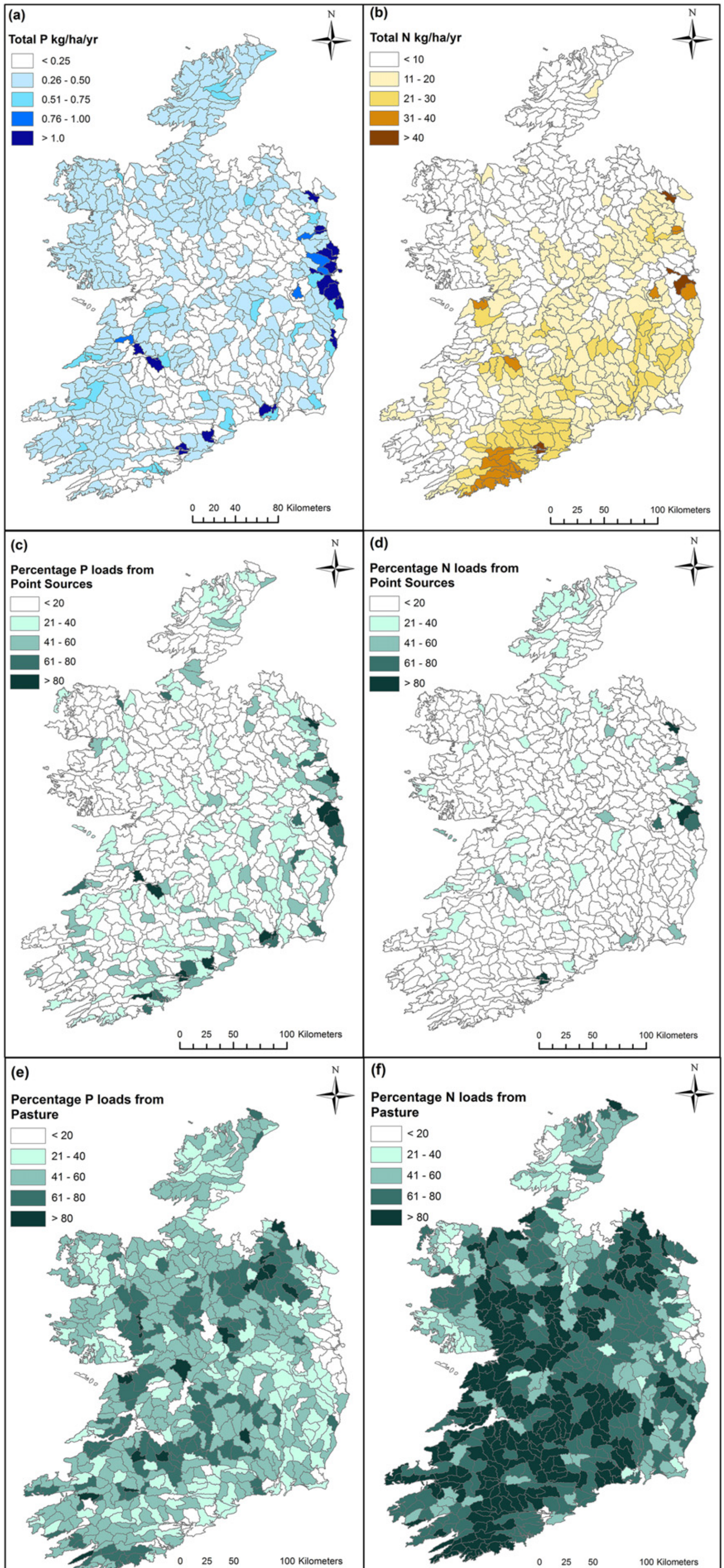
(ii) Model structure

Several nutrient sources have not been included in the current version of the framework, such as sewer misconnections which are potentially significant in urban areas (Ellis and Butler, 2015), and leaching of N from abandoned landfill sites. In addition, unknown sources and emerging pressures, including from phosphate dosing of raw water supplies, are also present in many catchments. These issues will inform the direction of model development and will drive new research and data collection in order to improve model performance.

(iii) Input (forcing) data

Several key input data sets to the SLAM framework, including LPIS and the annual environmental reports for WWTPs and industrial discharges, are collected and maintained for other management-orientated purposes and were not originally quality controlled for use in a national nutrient model. Zimmermann et al. (2016) outlined the administrative, spatial and thematic limitations of using the LPIS geodatabase for spatial modelling. Similar difficulties were encountered with other national datasets with tailored data quality reviews required prior to use in the SLAM. Furthermore, the structures of data sources changed periodically to meet the requirements of the data's primary purpose and, as a result, the model framework needed to be revised. This lack of control or first-hand knowledge of input data can result in unknown uncertainties being present in the model results.

Fig. 5. Total P (a) and N (b) emissions to surface water prior to lake retention (kg ha^{-1} yr^{-1}), percentage contributions from point sources for P (c) and N (d) and percentage contributions from pasture for P (e) and N (f) for 583 sub-catchment.


(a)
Total P kg/ha/yr
< 0.25
0.26 - 0.50
0.51 - 0.75
0.76 - 1.00
> 1.0
0 20 40 80 Kilometers
(b)
Total N kg/ha/yr
< 10
11 - 20
21 - 30
31 - 40
> 40
0 25 50 100 Kilometers
(c)
Percentage P loads from Point Sources
< 20
21 - 40
41 - 60
61 - 80
> 80
0 25 50 100 Kilometers
(d)
Percentage N loads from Point Sources
< 20
21 - 40
41 - 60
61 - 80
> 80
0 25 50 100 Kilometers
(e)
Percentage P loads from Pasture
< 20
21 - 40
41 - 60
61 - 80
> 80
0 25 50 100 Kilometers
(f)
Percentage N loads from Pasture
< 20
21 - 40
41 - 60
61 - 80
> 80
0 25 50 100 Kilometers

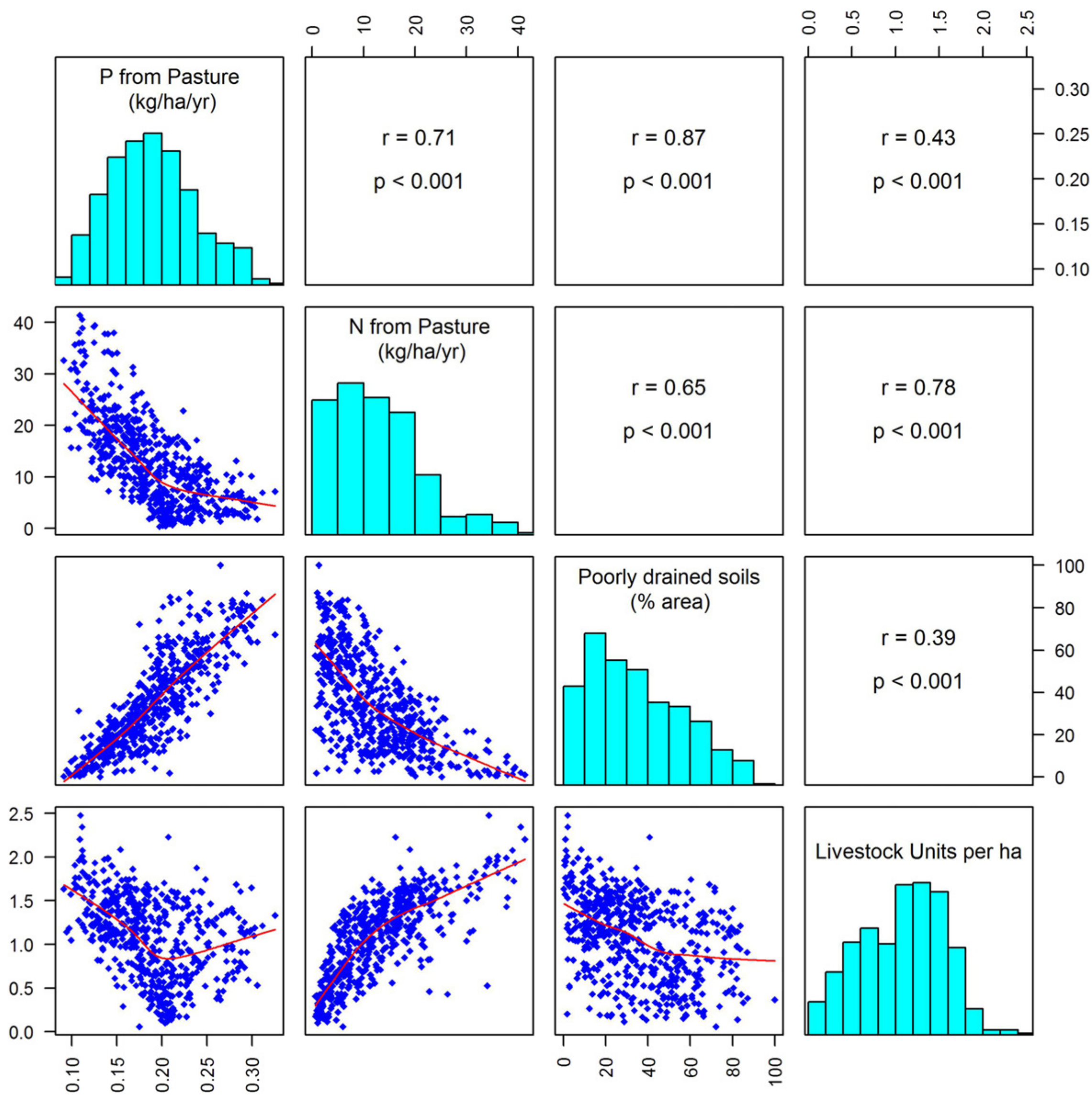


Fig. 6. Relationships between P loads from pasture (kg ha^{-1} yr^{-1}), N loads from pasture (kg ha^{-1} yr^{-1}), the area of pasture with poorly drained soils (%) and the livestock units per ha for 583 sub-catchments. This 'pairs plot' (R Core Team, 2016) shows histograms in the diagonal panels, with correlation plots for each pair of variables shown in the lower panels and the corresponding Spearman's rank-order correlation coefficient (r) and significance (p) for each shown in the mirrored upper panel.

(iv) Identification of parameter values

The model evaluation presented here and the development of the CCT-P and CCT-N models that calculated the agricultural emissions are based on national monitoring data. However, the uncertain nature of low temporal resolution monitoring programmes, including the monthly regime used in this study, is well researched (Johnes, 2007; Cassidy and Jordan, 2011; Jiang et al., 2014; Snelder et al., 2017). This is particularly relevant for P, as >80% of P emissions can occur in 10% of the year during high flow events (Ockenden et al., 2016), which are likely not well captured by low-temporal resolution monitoring data. Hence, it is predicted that, just like the monitoring data, the SLAM model under predicts the actual annual P emissions. However, the environmental quality standards are also based on this data and have been shown to reflect the impact of nutrients on river ecology (Donohue et al., 2006).

3.5. Informing integrated catchment management

In order to improve water management in Ireland, the Irish EPA has substantially strengthened the evidence-base on which decisions are made by implementing a rigorous risk-based approach to the water body characterisation and assessments required under the WFD (Daly et al., 2016). The SLAM framework was developed to support the proportional and pragmatic assessment of every sub-catchment within this national characterisation process. The purpose of these assessments was to determine the significant pressures impacting on water bodies that are *At Risk* of not meeting their WFD objectives. The significant pressures are those that are hydrogeologically connected to the water body and are known (or suspected) to be causing impact. Determining which of the multitude of pressures within a water body are significant is important so that measures can be more efficiently and specifically targeted to achieve water quality improvements.

The source apportionment results were produced for every subcatchment in Ireland and included in the EPA's WFD risk assessment alongside other national datasets, including ecological status and trends in ecological and chemical monitoring data; information on land use, pressures, pathways and sensitivity of receptors; enforcement, audit and inspection information from regulatory agencies; and local, on-the-ground knowledge from the Local Authorities and

Fisheries agency staff (Daly et al., 2016). This systems-approach is vital for integrated catchment management and effective WFD implementation (Voulvoulis et al., 2017).

Incorporating the SLAM results into the Irish EPA's catchment science assessments has facilitated assessment of nutrient load information in a logical, structured, consistent and comparative way across the country and has therefore provided robust assessment of the information. The results, however, are only one approach for determining significant pressures. The SLAM results should be used in combination with other tools, as nutrient load does not necessarily imply impact. The design of measures requires integrating hydrological and social science assessments to ensure decision makers have the best information when evaluating cost efficiency and effectiveness (Psaltopoulos et al., 2017), and models such as the SLAM provide some of the necessary information to feed into these assessments. For example, the annual percentage contribution of loads from septic tank systems may be small overall at the sub-catchment scale, but their impact in small stream headwaters can be significant during low flow periods (Withers et al., 2012). Similarly, it has been shown that transport factors are more important than source pressure in driving P losses from farms (Mellander et al., 2015). Hence, local investigative assessments are required prior to implementation of specific mitigation strategies.

The SLAM results have been analysed at a range of scales and coupled with other models in order to improve understanding of catchment dynamics. For example, the dynamic nature of anthropogenic pressures at catchment scale were examined using loading information spanning over a decade to explore the resulting impacts on Irish estuaries (Ní Longphuirt et al., 2016). At local scale, Mockler et al. (2016) evaluated the potential load reductions that could be achieved from alternative mitigation options targeting wastewater treatment plants and agricultural runoff in a nutrient-enriched water body. The upgrading of the SLAM Framework with new models and data will continue in order to support integrated catchment management in Ireland.

4. Conclusions

Catchment management can be supported by modelling at a range of scales and levels, all of which can reduce the resources required to synthesise and analyse large amounts of information. In this paper, national nutrient source apportionment results from the SLAM Framework indicated that:

- Agriculture was the dominant source of N across all catchments, whereas the dominant sources of P emissions varied by land use and hydrogeological and urban setting.
- P emissions from pasture were mainly driven by hydrogeological conditions, not pressure, highlighting that mitigation options should aim to interrupt the local source-pathway-receptor relationships.
- P emissions from wastewater are dominant in highly populated areas. However, municipal wastewater treatment has significantly improved in recent decades, more than off-setting increases in population.
- Annual nutrient loads from septic tank systems and industrial discharges are comparatively low at regional and sub-catchment scales. However, detailed investigative assessments are recommended where localised or low flow impacts are suspected.

These model results contributed to the characterisation and risk assessments of all Irish water bodies undertaken by the Irish EPA, which included a wealth of monitoring data and local knowledge, in order to identify the dominant sources of nutrients at regional and local scales.

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Appendix A. Source Load Apportionment Model structure

The main data sources for each component of the SLAM framework (Eq. (1)) as implemented in the SLAM version 2.05 are detailed in Table A1. A summary of the methods is provided in this appendix and further details are available in Mockler et al. (2016).

A.1. Wastewater discharges

A flexible model was developed for estimation of nutrient loads from wastewater treatment plants (WWTPs) to reflect the quality and quantity of available data at each plant. A data hierarchy was established which identified the best available data for each plant following international best practice (OSPAR, 2004). The actual nutrient load emissions reported for WWTPs were the preferred source, and were based on the 2014 Annual Environmental Reports (AERs), where available. If an AER was not available for a WWTP, data from the EPA Licensing Enforcement and Monitoring Application (LEMA) were used to estimate annual emission values. The annual agglomeration load, $Agglom_{N,P}$ (kg yr^{-1}), was calculated as:

$$Agglom_{N,P} = Agglom_{WWTP} + Agglom_{untreated} \tag{2}$$

where $Agglom_{WWTP}$ = load discharged from WWTP for N or P (kg yr^{-1}) and $Agglom_{untreated}$ = untreated sewerage from agglomeration for N or P (kg yr^{-1}).

Typically $Agglom_{WWTP}$ is the major component of the total agglomeration load, and was calculated as:

$$Agglom_{WWTP} = \begin{cases} AER_{N,P}\ , \text{ where } AER_{N,P} \text{ available} \\ PE \times RATE_{N,P} \times TREAT_{N,P}\ , \text{where } AER_{N,P} \text{ unavailable} \end{cases} \tag{3}$$

where $AER_{N,P}$ = reported annual emission for N or P (kg yr^{-1}) in AER; PE = population equivalent, as reported in LEMA; $RATE_{N,P}$ = nutrient production rate for N or P (kg yr^{-1}); and $TREAT_{N,P}$ = treatment efficiency factor based on WWTP treatment level.

Estimate of nutrient production rates ($RATE_{N,P}$) from 2014 AER data indicated about 12 g person^{-1} day^{-1}, or 4.38 kg yr^{-1} for N and 2 g person^{-1} day^{-1}, or 0.73 kg yr^{-1} for P. Treatment efficiency factors ($TREAT_{N,P}$) were related to treatment level (OSPAR, 2000), and are under review for Irish WWTPs.

Estimates of annual nutrient loads from untreated sewerage ($Agglom_{untreated}$) can include any loss from the sewer network from combined sewer overflows, also referred to as surface water overflows (SWOs), and emergency overflows (EOs). A tiered approach was identified for calculating $Agglom_{untreated}$ for each agglomeration to ensure best use of the available data, prioritising the AER data, and using a default estimate based on population equivalents where required, as follows:

$$Agglom_{untreated} = \begin{cases} \dfrac{ConcIN_{WWTP}}{DILUTE} \times Q_{SWO}\ , \text{where } Q_{SWO} \text{ available} \\ \dfrac{LoadIN_{WWTP}}{(1+\%LOSS)} \times \%LOSS, \text{where } Q_{SWO} \text{ unavailable} \end{cases} \tag{4}$$

where $QSWO$ = discharge volume from SWOs (m^3 yr^{-1}) as reported in the AER; $ConcINWWTP$ = average influent concentration of N or P (kg m^{-3} yr^{-1}) to WWTP; $LoadINWWTP$ = annual load at inlet of WWTP for N or P (kg yr^{-1}); $DILUTE$ = storm dilution factor (assumed value of 7 based on design guidance); and$\%LOSS$ = percentage of the total agglomeration load discharge by SWOs as reported in AER (default value of 3%).

A.2. Industrial discharges

The industrial discharges were estimated from two sources depending on the facility licence. Industrial discharges licenced by the EPA report annual emission values to the Pollutant Release and Transfer Register (PRTR). An average of three years (2011 − 2013) of TN and TP (kg yr^{-1}) values reported were used for each facility. Facilities licenced by local authorities under section 4 licences generally do not report annual emissions, and hence the licence limits were used to estimate an annual value (OSPAR, 2004), as:

$$Industry_{N,P} = Reported_{N,P}\ OR\ (25\% \times L_{max}) \tag{5}$$

where $Reported_{N,P}$ = annual nutrient emission load for N or P reported (kg yr^{-1}) and L_{max} = maximum allowable licenced nitrogen load for N or P (kg yr^{-1}).

A.3. Septic tank systems

A conceptual model based on export coefficients was developed to estimate nutrient losses from Domestic Wastewater Treatment Systems (referred to as septic tank systems in this study). The SANICOSE model (Gill and Mockler, 2016) synthesises over a decade of field studies on on-site systems in Ireland across many different soils types (Gill et al., 2007; Gill et al., 2009a; Gill et al., 2009b; Gill et al., 2009c; Sulleabhain et al., 2009; O'Luanaigh et al., 2012; Donohue et al., 2015) and combines factors relating to the efficiency of the septic tank systems with attenuation factors for the hydrogeological flow pathways, based on the diffuse pollution factors developed in the Pathways Project (Archbold et al., 2016). Three different conceptual pathways were considered through which loads from septic tank systems can reach surface water bodies:

- *Pathway 1:* inadequate percolation, i.e. surface pathway direct to surface water body;
- *Pathway 2:* near surface (soils and subsoils) pathway;
- *Pathway 3:* groundwater pathway.

See Gill and Mockler (2016) for further details on the septic tank system module structure and parameters.

A.4. Diffuse agricultural losses

The Catchment Characterisation Tool (CCT) calculated N and P losses from diffuse agriculture ($DiffAgri_{N,P}$) as follows:

$$DiffAgri_{N,P} = Leached_{N,P} \times \left[(NS \times \alpha_{N,P}) + \left(GW \times \beta_{N,P} \times \varphi_{N,P}\right)\right] \tag{6}$$

where $Leached_{N,P}$ = loads leached from the soils for N or P (kg yr^{-1}); $\alpha_{N,P}$ = near-surface pathway factors for N or P; $\beta_{N,P}$ = groundwater pathway factors for N or P; $\varphi_{N,P}$ = groundwater bedrock transport factor for N or P; NS = fraction of load to surface water via near surface pathway; and *GW* = fraction of load to groundwater (=1–NS), where:

$$NS = \frac{P_e - R}{P_e} \tag{7}$$

$$GW = \frac{R}{P_e} \tag{8}$$

P_e = annual effective precipitation (Hunter Williams et al., 2013) and *R* = average annual recharge from the GSI Groundwater Recharge Map (Hunter Williams et al., 2013).

A.4.1. Phosphorus leaching calculation and export coefficients

The CCT-P model was loosely based on the Phosphorus Indicator Tool (PIT) (Heathwaite et al., 2003), which was developed in the UK to estimate phosphorus losses from agricultural soils to surface waters. For both pasture and arable land cover, the CCT-P model estimated the applied nutrients using the maximum allowable fertilisation rates (S.I. 31, 2014). In addition to the applied nutrients, a value of Teagasc soil P index 3 was assumed which is equivalent to a Morgan's p value of 8 mg l^{-1} and 6.5 mg l^{-1} for arable and pasture, respectively. $Leached_P$ was then calculated as 1% of the applied and soil P loads. The pathway coefficients (α_P *and* β_P) were modified from the PIT model to match available nutrient monitoring data in Irish agricultural catchments (Tables A2). The P groundwater factors (β_P) were related to the depth to bedrock, except in areas of peat soils (Table A3). The CCT-P model assumed that there is no attenuation of P in Irish aquifers. Further information on model development is available in Archbold et al. (2016 and reference therein).

A.4.2. Nitrogen leaching calculation and export coefficients

In the CCT-N model, the $Leached_N$ from pasture was calculated using Ncycle_IRL (del Prado et al., 2006) modelled values for groupings of fertilizer application rate, soil drainage type, and pasture type. The Land-Parcel Identification System (LPIS) provided applied rates of N (kg ha^{-1} yr^{-1}) at farm level (see Zimmermann et al., 2016). Diffuse nutrient losses from arable land were calculated similarly to the NIRAMS model (Dunn et al., 2004). The CCT-N model estimated the available (net) nutrients using the maximum allowable fertilisation rates (S.I. 31, 2014), atmospheric deposition and average off-take values. Denitrification varies by soil texture, with rates of 5%, 15% and 75% applied to sandy, loamy or clay/peaty soils, respectively. $Leached_N$ was calculated using the NLEAP model (Shaffer et al., 1994):

$$Leached_N = Available_N \times \left(1 - \exp\left[-K^* \left(\frac{WAL}{SATC}\right)\right]\right) \tag{9}$$

where,

$Available_N$ = available (net) N (kg ha^{-1} yr^{-1}).

K = leaching coefficient (0.7 for sandy soils and 1.2 for other soils).

WAL = water available for leaching (mm yr^{-1}), estimated from P_e.

SATC = Soil saturated capacity (mm yr^{-1}) estimated from Anthony et al. (1996) based on the soil drainage classification in the Irish National Soils Map (Teagasc et al., 2006).

For both pasture and arable areas, nutrient reduction factors (α_N, β_N, and φ_N) were then applied to the leached amount of nutrients ($Leached_N$) to predict the final losses to water ($DiffAgri_N$). The coefficients representing the delivery of N via near surface pathways (α_N) were linked to subsoil permeability, as N tends to move through the subsoils before arriving at the surface water receptor (Table A4). In addition, a map of the possible location of land drains (Mockler et al., 2014) indicated a preferential delivery pathway for nitrate in low permeability subsoils. Groundwater export coefficients (β_N) were determined following a literature and expert elicitation review (Packham et al., 2015). These coefficients vary by subsoil permeability and depth to bedrock (Table A5), both available as maps from Geological Survey of Ireland (GSI). The bedrock attenuation coefficient (φ_N) was linked to aquifer bedrock units with the potential for denitrification, mostly due to the presence of pyrite (Table A6).

A.5. Other diffuse nutrient sources

Nutrient export coefficients linked to CORINE land cover classes have been used in many studies to estimate annual load apportionment (e.g. Grizzetti et al., 2012 and references therein). In the SLAM framework, the diffuse nutrient emissions from forestry, peatlands and urban areas were calculated as:

$$Diffuse_{N,P} = Area \times Export_{N,P} \tag{10}$$

where *Area* = area of the land cover category from Corine 2012 (ha) and $Export_{N,P}$ = export coefficient for N or P (kg ha^{-1} yr^{-1}). The forestry, peatlands, and diffuse urban export coefficients (MCOS, 1999) are outlined in Table A7.

A.5.1. Atmospheric deposition

In the SLAM framework, atmospheric deposition on land was accounted for in each diffuse submodel, either explicitly (i.e. for diffuse agriculture) or implicitly (e.g. in the forestry export coefficients). In addition, direct deposition on open water was represented in the model. The EPA lake segment dataset was used to represent open water, which includes lakes with a total area of 1333 km^2. $Export_N$ (Eq. (10)) was calculated spatially for all lakes from a map of atmospheric deposition of N (Henry and Aherne, 2014), with an average rate of deposition on lakes of 9.4 kg N ha^{-1} yr^{-1}. Uniform rates of 0.5 kg ha^{-1} yr^{-1} total phosphorus deposition ($Export_P$) were assumed (Jordan, 1997).

A.6. Lake retention

For the SLAM framework, a simple lake retention model was implemented which reduces loads from catchments draining through all lakes above a threshold size of 50 ha. The retention factors developed during studies in the Lee catchment (Sullivan et al., 1995) were applied to catchments upstream of lakes, with $Lake_{iN}$, = 0.1 and $Lake_{iP}$ = 0.24 (see Eq. (1)).

Table A1
Data sources for the SLAM framework (v 2.05).

Sub-model	Input data & source
Waste water discharges	2014 Annual Environmental Report data (EPA, 2015)
	2014 EPA Licensing Enforcement and Monitoring Application (LEMA)
Diffuse urban sources	2012 CORINE land cover (Lydon and Smith, 2014)
Industrial discharges	Section 4 licence limits (EPA, 2015)
	2011–2013 PRTR database (EPA, 2015)
Septic tank systems	Non-sewered house dataset & surface water bodies (EPA, OSi)
	Karst feature vulnerability & subsoil permeability (GSI)
Pasture (diffuse agriculture)	2012 LPIS (DAFM dataset)
	2010 agricultural census data (Central Statistics Office, 2010)
Arable (diffuse agriculture)	2012 LPIS (DAFM dataset)
	2010 agricultural census data (Central Statistics Office, 2010)
	Good Agricultural Practices (GAP) Regulations (S.I. 31, 2014)
	Fertilizer application rates (Lalor et al., 2010)
Forestry	2012 CORINE land cover (Lydon and Smith, 2014)
Peatlands	2012 CORINE land cover (Lydon and Smith, 2014)
Atmospheric Deposition	Lake segment areas (EPA dataset)
	N deposition map (Henry and Aherne, 2014)

Table A2
Phosphorus near surface pathway factors.

Soil type	P near surface factor
Clayey soil classified as Wet	0.4
Sandy soil classified as Dry	0.2
Peat	0.4
Drain flow	0.7

Table A3
Phosphorus groundwater pathway factors.

Depth to bedrock	P subsoil factors
X Extreme (0–1 m and near Karst features)[a]	0.4
E Extreme (1–3 m)[a]	0.15
3–5 m[a]	0.05
5–10 m[a]	0.02
>10 m[a]	0.01

[a] Exception peat subsoils.

Table A4
Nitrate near surface pathway factors.

Subsoil permeability	N near surface factor
Low	0.2
Low & likely to have land drains	0.7
Moderate	0.55
High	1
N/A	0.95
Water/lake/rock	0.95

Table A5
Nitrate groundwater pathway factors.

Subsoil permeability Depth to bedrock	*Low*	*Moderate*	*High*	*N/A (DTB < 3 m)*
0–1 m	1.0	1.0	1.00	1.0
1–3 m	0.60	0.95	1.00	0.95
3–5 m	0.20	0.90	1.00	–
5–10 m	0.05	0.85	1.00	–
>10 m	0.01	0.75	1.00	–

Table A6
Nitrate groundwater bedrock pathway factors.

Bedrock unit[a]	Transport factor
Unit 1a, unit 1b	0.65
All other bedrock units	1

[a] Units as defined by the GSI Denitrification Map.

Table A7
Nutrient export rates in the SLAM model (v 2.05) for urban, forestry and peat land cover.

Land cover	CORINE Code	Area (km^2)	% Area of Ireland	Rates (kg ha^{-1} yr^{-1}) N	P
Urban					
Continuous urban fabric	111	31	0.0%	5	1.4
Discontinuous urban fabric	112	1119	1.6%	5	0.86
Industrial or commercial units	121–124	130	0.2%	5	1.88
Mine, dump, construction sites	131–133	3	0.0%	5	2.15
Green urban areas, sport and leisure	141, 142	36	0.4%	5	1.4
Forestry					
Broad-leaved, coniferous, mixed forest	311–313	3765	5.3%	5.42	0.33
Natural grassland, moors, heathland	321, 322	1411	2.0%	5.42	0.33
Transitional woodland-shrub	324	2898	4.1%	3.71	0.57
Peat					
Inland marshes	411	194	0.3%	2	0.33
Peat bogs	412	10,338	14.7%	2	0.2

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