NUTRIENT LOAD APPORTIONMENT TO SUPPORT THE IDENTIFICATION OF APPROPRIATE WATER FRAMEWORK DIRECTIVE MEASURES

Eva M. Mockler, Jenny Deakin, Marie Archbold, Donal Daly and Michael Bruen

Eva M. Mockler (corresponding author; email: eva.mockler@ucd.ie), UCD Dooge Centre for Water Resources Research and UCD Earth Institute, University College Dublin, Dublin 4, Ireland; Jenny Deakin, Environmental Protection Agency, Clonskeagh, Dublin 14, Ireland; Marie Archbold, Environmental Protection Agency, Clonskeagh, Dublin 14, Ireland; Donal Daly, Environmental Protection Agency, Clonskeagh, Dublin 14, Ireland; and Michael Bruen, UCD Dooge Centre for Water Resources Research and UCD Earth Institute, University College Dublin, Dublin 4, Ireland.

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ABSTRACT

A model for predicting the sources of nutrient loads (phosphorus and nitrogen) to water has been developed to support Water Framework Directive (WFD) implementation. The Source Load Apportionment Model (SLAM) framework described in this paper integrates catchment data and pressure information from point discharges and diffuse sources to enable characterisation of source–pathway–receptor relationships. Hydrogeological controls have a strong impact on nutrient fluxes, particularly in agricultural catchments, and have been incorporated into the diffuse agricultural model (the CCT). Results for the Suir catchment matched the measured loads of nitrogen and phosphorus well, and showed that pasture is the dominant source of nitrogen. The main sources of phosphorus in sub-catchments varied between diffuse agriculture, wastewater and industrial discharges. A relatively small proportion (13%) of the Suir catchment area requires a reduction in phosphorus emissions to achieve Good status. In these areas, model results can be used in conjunction with local knowledge gathered through the WFD characterisation process to identify significant pressures that contribute excessive nutrient loads. An example of assessing load reduction scenarios is presented to illustrate how modelling can support catchment scientists in identifying appropriate measures.

INTRODUCTION

Eutrophication is the most significant issue for surface waters in Ireland. Although nutrient levels are mostly stable or decreasing (Bradley *et al.*, 2015), half of Irish river water bodies still require improvements to bring them to 'Good' status, as required by the Water Framework Directive (WFD) (2000/60/EC). The agricultural sector is the most common suspected source of nutrient pollution in Irish rivers (Bradley *et al.*, 2015). However, quantifying nutrient losses from agricultural sources is challenging due to the complex and heterogeneous nature of their sources and transportation pathways in the landscape (Deakin *et al.*, this issue). There can be individual nutrient loss hot-spots, or critical source areas (Pionke *et al.*, 2000), which contribute a relatively high proportion of the nutrients exported from the landscape. To estimate accurately the nutrient losses from agriculture, information on the spatial distribution of the fertiliser and slurry applied to land is required together with information on the characteristics of the geological environment that influence the transmission and attenuation of nutrients.

Source apportionment is used to estimate the nutrient load from various sectors entering water bodies, following attenuation or treatment. Agriculture is typically the principal source of nitrogen in water bodies in Europe, whereas in many countries households and industries tend to be the dominant contributors of phosphorus (Bøgestrand *et al.*, 2005). Nutrient load apportionment can be a useful tool to support catchment characterisation, including an assessment of monitoring data, within an appropriate management framework; see e.g. Deakin (2013). A ranking of annual nutrient export by sector is needed to focus the time and money spent on catchment management in the areas where they will achieve the most impact.

The general method of source-oriented load apportionment models is an export coefficient approach, i.e.:

1. calculate available annual average nutrient loads from each sector;
2. reduce these loads by a factor to account for treatment (e.g. urban wastewater) or attenuation in the environment (e.g. diffuse agricultural

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sources), where relevant—this estimates the annual in-stream loads from each sector;
3. compare the estimated annual in-stream loads with annual loads calculated from measurements, where available (Bøgestrand *et al.*, 2005).

Previous studies have undertaken load apportionment modelling using this export coefficient method (MCOS, 2002; Jordan and Smith, 2005; Smith *et al.*, 2005; Campbell and Foy, 2008; Ní Longphuirt *et al.*, 2015) and with statistical methods based on monitoring data (Daly *et al.*, 2002; Grizzetti *et al.*, 2005; 2012; Greene *et al.*, 2011).

The Source Load Apportionment Model (SLAM) is a new nutrient modelling framework developed for Irish catchments. The SLAM framework has integrated the most up-to-date conceptual understanding of hydrogeological pathways for nutrient transport with national datasets relating to nutrient management. The purpose of the model is to inform a risk-based approach to improving and maintaining nutrient-sensitive ecosystems, which will also contribute to protecting drinking and bathing water resources now and into the future. Using meaningful data analyses and modelling results, SLAM supports the development of river basin management plans and the identification of potential appropriate measures. Results from the SLAM framework could also be useful in multi-criteria decision analysis and scenario analysis to forecast impacts of policies including, for example, Food Wise 2025 (DAFM, 2015) and Irish Water's capital investment programme.

The SLAM estimates the average annual amount of nitrogen (N) and phosphorus (P) exported from all major sources in Irish subcatchments, using the best available national datasets. Many Irish subcatchments are 'semi-ungauged', and have fewer than twelve measured nutrient concentrations a year. Therefore, SLAM has been developed as an export coefficient model, which does not require monitoring data for calibration. However, the model was developed and tested against monitoring data, where the data are available, to establish confidence in the export coefficients.

The objectives of this paper are to (1) describe the SLAM framework, including equations, assumptions and input data, and (2) demonstrate the use of the SLAM to identify potential sector-level measures in the Suir catchment, including:

- assessment of the nutrient load reductions required to return all subcatchments to 'Good' WFD status;
- evaluation of the SLAM results for 29 subcatchments of the Suir for N and P;
- an example of using the SLAM results to identify potential appropriate measures for a sector by quantifying possible load reduction scenarios.

DATA AND METHODS

STUDY CATCHMENT

The Suir is a 3500km^2 catchment in the south-east of Ireland, with 29 subcatchments (Fig. 1), some of which have unsatisfactory water quality. The catchment land use is predominantly agriculture, mainly pasture, with an average stocking rate of 1.4 livestock units ha^{-1}. The total population is 200,000, and the average density of septic tank systems in the catchment is 8km^{-2}. The largest town is Waterford at the mouth of the Suir estuary, with a population of approximately 56,000 in 2016.

NUTRIENT LOADINGS DATA

For each source of nutrients, the relevant spatial data were combined with loadings information from monitoring data, environmental reporting or previous research to produce annual load estimates. The main input datasets for each of the sector submodels are outlined in Table 1.

The 2012 CORINE (Co-ordinated Information on the Environment) land cover data-series was devised as a means of compiling geospatial environmental information in a standardised format across Europe (Lydon and Smith, 2014). In SLAM version 2.04, the CORINE 2012 data were used in the non-agricultural land cover calculations i.e. the forestry, peatlands and urban submodels. As most of Ireland (68%) is classified as agricultural land, the quality of data for agricultural land management has a major impact on the overall confidence in a load apportionment model. Hence, the Land-Parcel Identification System (LPIS) was used as input data to the pasture and arable submodels, as it provides a superior source of agricultural land management information, particularly when combined with additional data from the DAFM. See Zimmerman *et al.* (2016) for details of the LPIS and its use in environmental research.

IN-STREAM LOAD ESTIMATION

As streamflow and concentrations are not generally measured continuously, various approximation methods are used for estimating annual in-steam loading from instantaneous water-quality sampling (e.g. Quilbé *et al.*, 2006; Cassidy and Jordan, 2011). The flow-weighted mean concentration ($\mathbf{L_{f\text{-}w}}$) method (Eq. 1) has been widely used in many studies (Jiang *et al.*, 2014 and references therein). It uses n individual spot measurements of nutrient concentration $\mathbf{C_i}$ (mg l^{-1}), together with the mean streamflow for the day of the measurement, as follows:

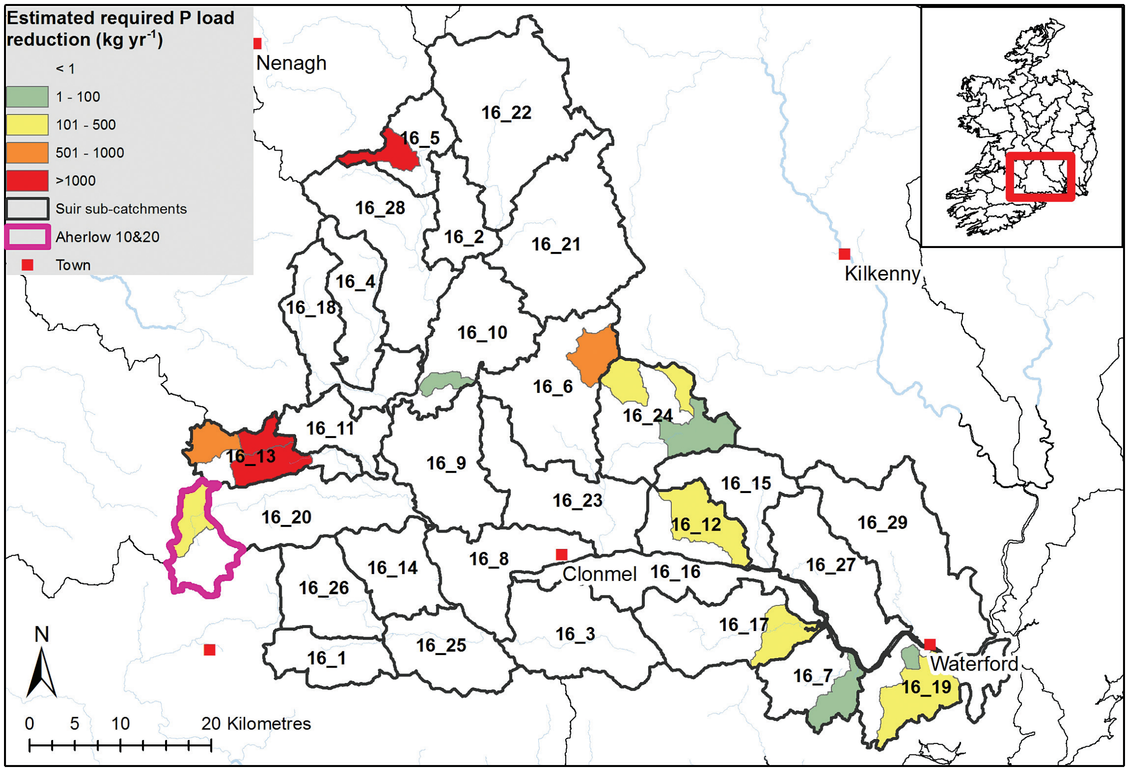

Fig. 1—Load reduction required for phosphorus loads to attain 'Good' status based on annual average concentration of 0.035mgl^{-1}.

$$L_{f\text{-}w} = \boldsymbol{K}\frac{\Sigma_{i=1}^{n}(C_iQ_i)}{\Sigma_{i=1}^{n}(Q_i)}.\bar{Q} \qquad (1)$$

where $\mathbf{Q_i}$ is the corresponding daily mean streamflow (m^3 s^{-1}), $\bar{Q}$ is the mean streamflow and is the unit conversion factor.

Flow and nutrient concentration data for the Suir catchment were taken from the OSPAR riverine inputs and direct discharges programme (RID), similarly to (O'Boyle *et al.*, 2015). Additional nutrient monitoring data were supplied by the Environmental Protection Agency (EPA) for several locations within the Suir catchment, with a range of sampling frequencies from monthly to quarterly (EPA 2006).

Where hydrometric data for daily mean flows are not available within a reasonable proximity to the EPA nutrient sampling location, average annual flows were estimated using the 30 percentile flow calculated by the EPA HydroTool (EPA 2016). Load estimates can then be calculated from simple annual averages as:

$$L_{avg} = \bar{C}\cdot\bar{Q}\cdot K \qquad (2)$$

Nutrient load estimates using this simple method were used for load reduction assessments at waterbody or subcatchment scale (outlined below), where representative hydrometric stations were not identified.

LOAD REDUCTION ASSESSMENT AND SCENARIO ANALYSES

Where a water body or subcatchment was identified as being 'At Risk' of not achieving 'Good' WFD status due to high nutrient concentrations, a load reduction assessment was carried out (Fig. 2), as follows.

- *Step 1:* Assessment of in-stream load reduction required using an in-stream load estimation method and the nutrient environmental quality standard (EQS), or a surrogate/equivalent proxy, to achieve the required WFD objective, typically 'Good' status. This was carried out (a) at water body or subcatchment level, and (b) at catchment scale, taking the transitional and coastal (TRaC) water bodies into consideration.
- *Step 2:* Comparison of in-stream loads and the load apportionment model results with consideration of the likely sources of additional loads that have not been modelled (e.g. incidents and accidents) where there are notable differences.
- *Step 3:* Identification of scenarios to achieve load reduction and issue/areas for investigative assessments

As environmental quality standards have not been set in legislation for total phosphorus (TP), EQS values for molybdate reactive phosphorus (MRP) were used as a conservative proxy in Step 1.

Table 1—Data sources for the SLAM framework (v. 2.04).

Submodel	*Input data and source*
Wastewater 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)
	2011–13 PRTR database (EPA, 2015)
Septic tank systems	Non-sewered house dataset and surface water bodies (EPA, Ordnance Survey Ireland)
	Karst feature vulnerability and subsoil permeability (GSI)
Pasture (diffuse agriculture)	2012 LPIS (DAFM dataset)
	2010 agricultural census data (CSO, 2010)
Arable (diffuse agriculture)	2012 LPIS (DAFM dataset)
	2010 agricultural census data (CSO, 2010)
	Good Agricultural Practices (GAP) Regulations (Government of Ireland, 2014)
	Fertiliser 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)

Based on mean values, the EQS values are 25µg P l^{-1} and 35µg P l^{-1} for High and Good status boundaries in rivers, respectively (Government of Ireland, 2009). Although the contribution of MRP to TP may vary widely in discharges (e.g. EPA, 2015), assuming that all of the TP is biologically

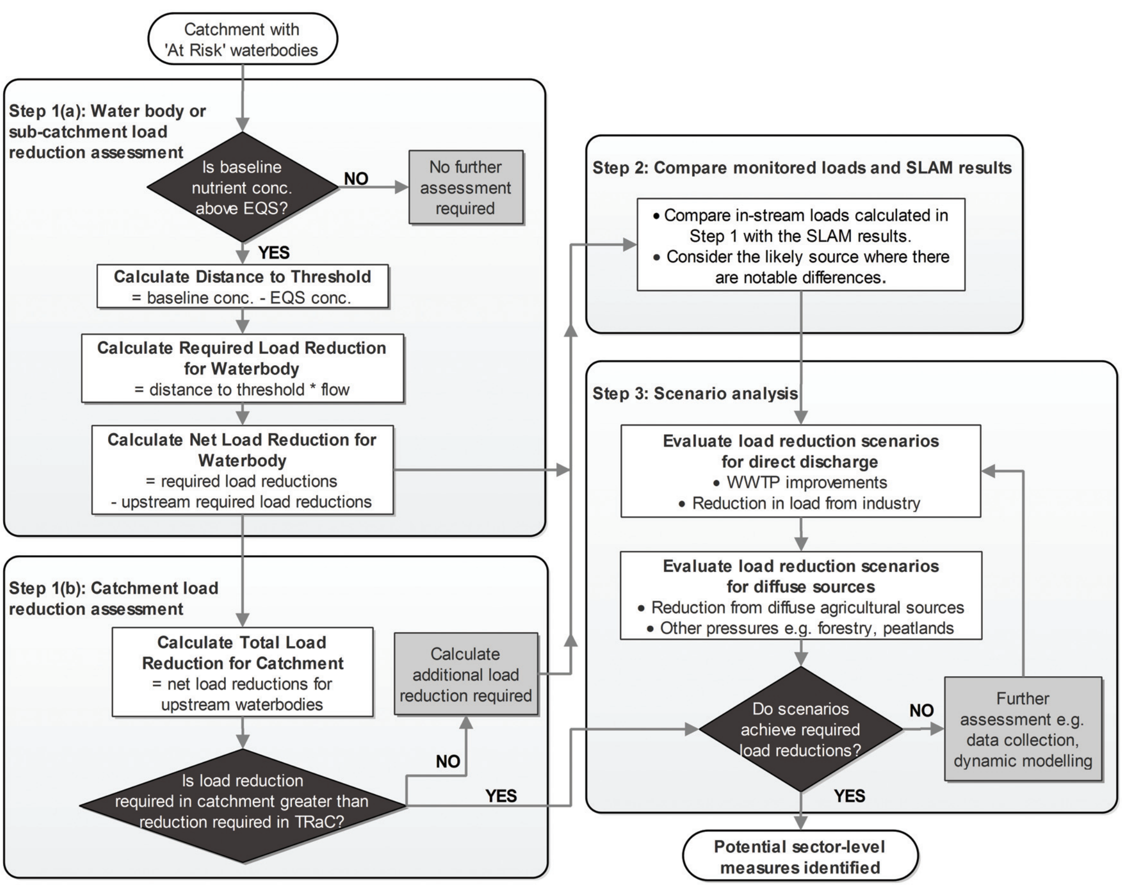

Fig. 2—Process flow chart to identify nutrient load reductions required and potential sector level measures.

available in the form of MRP is a conservative approach. In other words, if the total phosphorus load meets the MRP EQS, and MRP is only a fraction of the TP load, then the MRP requirements will be met.

Following the above method, an assessment of alternative mitigation measures was undertaken in the 'at risk' area of the subcatchment Suir_SC_090. Measures relating to upgrading wastewater treatment plants to either secondary or tertiary treatment were evaluated using nutrient reduction factors dependent on treatment level (OSPAR, 2000). There are many potential measures in relation to diffuse sources; for example, Schoumans *et al.* (2014) distinguished 83 measures for reducing nutrient losses to surface waters under the following four groupings:

1. nutrient and livestock management, e.g. apply P near the roots instead of broadcast;
2. crop and soil management, e.g. reduce soil compaction;
3. water, land-use and landscape management, e.g. vegetated buffer strips;
4. surface water management, e.g. create wetlands.

For the Aherlow example, enhancement of buffer strips was selected as the potential measure to mitigate diffuse P losses in overland flow. Several experiments have reported extremely effective, although widely varying, P removal rates (Hoffmann *et al.*, 2009; Kay *et al.*, 2009; Sharpley *et al.*, 2009; Stevens and Quinton, 2009). The effectiveness of buffer strips in controlling P in surface runoff is highly dependent on how they are placed and managed (Dorioz *et al.*, 2006), and is typically a function of width, slope, soils and hydrogeological conditions (Collins *et al.*, 2016). Some increases in P loads have been reported due to inadequate width or design which may become sources of soluble P release as they become more saturated (Hoffmann *et al.*, 2009; Kay *et al.*, 2009). Overall, the enhancement of buffer strips was identified as a viable potential measure for the agricultural sector in this example.

SOURCE LOAD APPORTIONMENT MODEL STRUCTURE

The SLAM framework was implemented in an ArcGIS-based model with individual modules for each sector to facilitate further development and collaboration. Each sector module was linked to relevant national datasets; for example, the pasture module used the Land-Parcels Identification System (LPIS) and the urban wastewater module incorporated data from annual environmental reports (AERs). The total annual nutrient load at the outlet of each subcatchment (L_j) was calculated as:

$$L_j = (Point_j + Diffuse_j)*(1 - Lake_j) \tag{3}$$

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

The methods and data sources for each component of Eq. 3 as implemented in SLAM version 2.04 are detailed below.

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 AER data, where available. If an AER return 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:

$$Agglom_{N,P} = Agglom_{WWTP} + Agglom_{untreated} \tag{4}$$

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:

$$Agglom_{\mathrm{WWTP}} = \begin{cases} AER_{N,P}, & where\ AER_{N,P}\ available \\ \mathrm{PE} \times RATE_{N,P} \times TREAT_{N,P}, & where\ AER_{N,P}\ unavailable \end{cases} \tag{5}$$

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

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

During storm events, the sewer network fills rapidly and cannot discharge all wastewater to the treatment plant, resulting in discharges to surface waters. 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). The possible 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. Hence, there are significant uncertainties surrounding estimates. 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:

$$Agglom_{untreated} = \begin{cases} \frac{ConclN_{WWTP}}{DILUTE} \times Q_{SWO,}\ where\ Q_{swo}\ available \\ \frac{LoadIN_{WWTP}}{(1+\%Loss)} \times \%Loss,\ where\ Q_{swo}\ unavailable \end{cases} \quad (6)$$

where Q = discharge volume from SWOs ($m^3\ yr^{-1}$) as reported in the AER; $ConcIN_{WWTP}$ = average influent concentration of N or P ($kg\ m^{-3}\ yr^{-1}$) to WWTP; $LoadIN_{WWTP}$ = 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%).

In the United States, the USEPA attributes 4% of total volume and 9% of the total biological oxygen demand (BOD_5) load from municipal discharges to SWOs (US EPA 2004: Chapter 4). In Ireland, over 1000 SWO and EO locations were reported in 2013 AERs, of which approximately 400 reported values for spill frequency and volume. On average, these overflows spilled 23 times per year, discharging an average of over 21,000m^3 each of mixed storm water and effluent. Assuming a storm water dilution factor of 7, each of these points contributed an annual average of over 90 kg N and 15 kg P to surface waters. There is a large degree of uncertainty surrounding estimates of nutrient emissions from SWOs; however, in some areas these are suspected significant pressures.

INDUSTRIAL DISCHARGES

The industrial discharges were estimated from two sources depending on the facility licence. Industrial discharges licensed 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 licensed 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:

$$Industry_{N,P} = Reported_{N,P}\ OR(25\% \times L_{max}) \quad (7)$$

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

SEPTIC TANK SYSTEMS

A conceptual model based on export coefficients was developed to estimate the nutrient losses from septic tank systems. This model builds on previous methods (Kavanagh and Bree, 2009; EPA, 2013), incorporating results from research (Gill *et al.*, 2009a,b). The model 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 submodel structure and factors.

DIFFUSE AGRICULTURAL LOSSES

The total nutrient loads to surface water were modelled by the Catchment Characterisation Tool (CCT) (Archbold *et al.,* 2016 and references therein). The calculation of nutrient losses of N or P from diffuse agriculture ($DiffAgri_{N,P}$) can be summarised as:

$$DiffAgri_{N,P} = Leached_{N,P} \times [(NS \times \alpha_{N,P}) + (GW \times \beta_{N,P} \times \varphi_{N,P})] \quad (8)$$

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 DWTS leached load to surface water via near surface pathway; and GW = fraction of DWTS leached load to groundwater (= 1 − NS), where:

$$NS = \frac{P_e - R}{P_e} \quad (9)$$

$$GW = \frac{R}{P_e} \quad (10)$$

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).

The near surface pathway ($\alpha_{N,P}$), groundwater pathway ($\beta_{N,P}$), and groundwater bedrock transport factors (f_N) for N and P are outlined in the Appendix (Tables A1 to A5).

DIFFUSE N LOSSES

For pasture, the CCT used the Ncycle_IRL Model (del Prado *et al.*, 2006) to calculate the leached nitrate values associated with the fertiliser application rate and soil drainage and pasture type. Inputs to the pasture model were from the LPIS (see Zimmerman *et al.*, 2016), which provide applied rates of N (kg ha^{-1} yr^{-1}) at land parcel level. Nutrient reduction factors were then applied to the leached amount of nutrients (Eq. 8) to predict the final load to rivers.

Diffuse nutrient losses from arable land cover were calculated similarly to the NIRAMS model (Dunn *et al.*, 2004). The CCT estimated the available (net) nutrients using the maximum allowable fertilisation rates (Government of Ireland, 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. The amount of leached nutrients is calculated using the NLEAP model (Shaffer *et al.*, 1994):

$$Leached_N = Available_N \times \left(1 - \exp\left[-K^*\left(\frac{WAL}{SATC}\right)\right]\right) \quad (11)$$

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; and 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).

DIFFUSE P LOSSES

The P model of the CCT was loosely based on the Phosphorus Indicators 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 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 of the PIT model were modified to match available nutrient monitoring data in Irish agricultural catchments (see Appendix, Tables A2 and A4). Further information on model development is available in Archbold *et al.* (2016) and references therein). The CCT outputs are presented as pollution impact potential (PIP) maps which indicate the degree of risk from diffuse sources of P (Fig. 3).

OTHER DIFFUSE NUTRIENT SOURCES

Nutrient export coefficients linked to CORINE land cover classes have been used in many studies to estimate annual load apportionment (McGuckin *et al.*, 1999; MCOS, 2002; Smith *et al.*, 2005; White and Hammond, 2006; 2009; Campbell and Foy, 2008; Ní Longphuirt *et al.*, 2015). In the SLAM framework, the diffuse nutrient emissions from forestry, peatlands and urban areas were calculated as:

$$Diffuse_{N,P} = Area \times Export_{N,P} \quad (12)$$

where *Area* = area of the land cover category from Corine 2012 (ha) and $Export_N$, = export coefficient for N or P (kg ha^{-1} yr^{-1}).

Forestry covers over 10% of Ireland's land surface (Table 2) and over half of this is on blanket peat. The forestry submodel coefficients (MCOS 1999; Table 2) are being further developed as the effects of forestry operations, particularly clear-felling and wind-rowing activities, have been shown to significantly impact the environment (Campbell and Foy, 2008; Kelly-Quinn *et al.*, 2014; O'Driscoll *et al.*, 2014).

The peatlands submodel export coefficients (MCOS 1999; Table 2) are based on CORINE data for inland marshes and peat bogs. Recent studies have reported a broad range of values for emissions from peat, which are dependent on land management practices. For example, reported export rates for MRP from Bord na Móna actively worked bogs in the Lough Derg and Lough Ree catchments range from 0.16 to 0.64kg ha^{-1} (Kirk McClure Morton, 1999). In comparison, the export rate from Boora parklands (cutover bog converted to lakelands) was 0.03kg ha^{-1}. Hence, as with the forestry submodel, the structure and coefficients of the peatlands model are under review.

Diffuse urban P loads originate from a variety of sources including atmospheric deposition and runoff from roads, roofs, construction sites and gardens. In addition, subsurface nutrient emissions are often present from, for example, sewers and graveyards (Groundwater Task Team, 2010). The urban submodel structure (Eq. 12) is being developed to include these sources explicitly; however, the version presented here is based on CORINE land cover export coefficients (MCOS, 1999; Table 2).

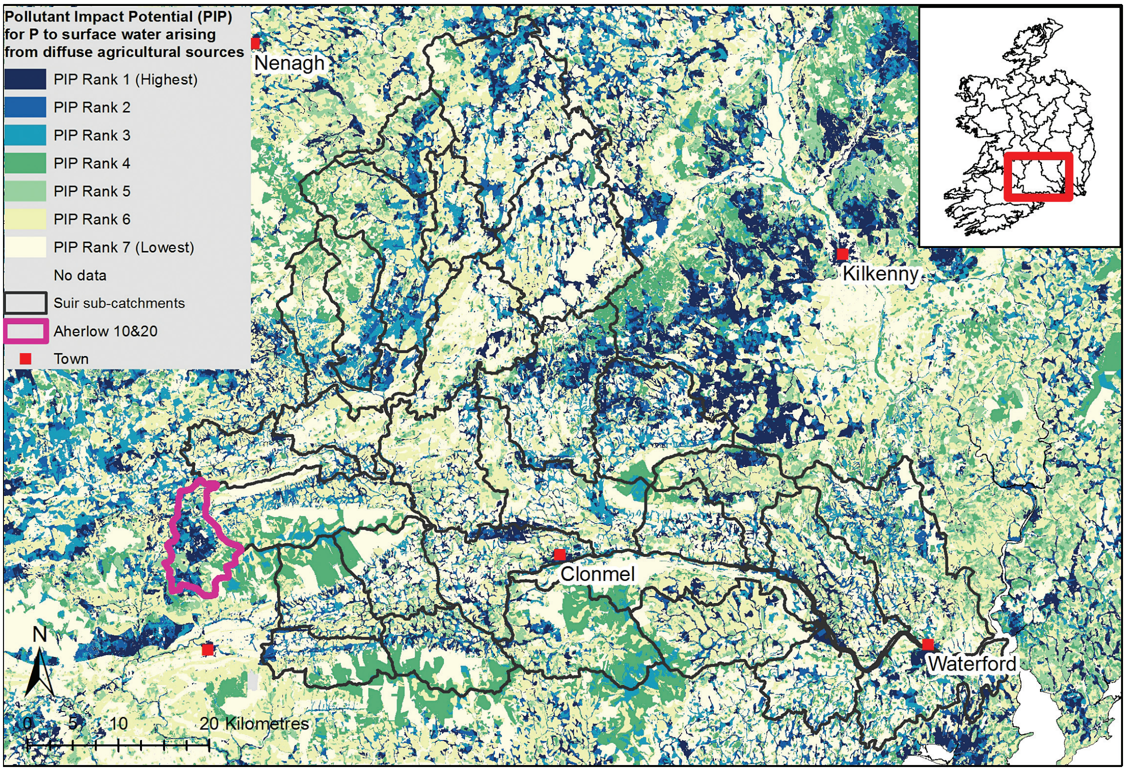

Fig. 3—PIP map for phosphate to surface water arising from diffuse agricultural sources, with dark blue areas indicating the highest risk.

ATMOSPHERIC DEPOSITION

In the SLAM framework, atmospheric deposition on land was accounted for in each diffuse sub-model, 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 1333km^2. $Export_N$ (Eq. 12) 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 9.4kg N ha^{-1} yr^{-1}. Uniform rates of 0.5kg ha^{-1} yr^{-1} total phosphorus deposition ($Export_p$ were assumed (Jordan, 1997).

LAKE RETENTION

Several approaches for estimation of lake retention are available, ranging in complexity from export

Table 2—Nutrient export rates in the SLAM model (v2.04) for urban, forestry and peat land cover.

	Land cover	*CORINE code*	*Area (ha)*	*% area of Ireland*	*N rate (kg ha^{-1} yr^{-1})*	*P rate (kg ha^{-1} yr^{-1})*
Forestry	Broad-leaved, coniferous and mixed forest	311,312,313	376,450	5.3	5.42	0.33
	Natural grassland, moors and heathland	321,322	141,118	2.0	5.42	0.33
	Transitional woodland–shrub	324	289,767	4.1	3.71	0.565
Peat	Inland marshes	411	19,392	0.3	2	0.325
	Peat bogs	412	1,033,832	14.7	2	0.2
Urban	Continuous urban fabric	111	3100	0.0	5	1.4
	Discontinuous urban fabric	112	111,914	1.6	5	0.86
	Industrial or commercial units	121	13,003	0.2	5	1.88
	Construction sites	133	304	0.0	5	2.15
	Green urban areas	141	3573	0.1	5	1.4
	Sport and leisure facilities	142	22,263	0.3	5	1.4

coefficient models based on lake retention time (e.g. Foy, 1992) to complex process-oriented models (e.g. Hipsey *et al.*, 2013). For the SLAM framework, a simple lake retention model is in development which reduces loads from catchments draining through all lakes above a threshold size of 50ha. As the Suir catchment does not contain any lakes greater than 50ha, the lake retention model was not included in this catchment.

RESULTS

COMPARISON WITH MONITORING DATA: SUIR CATCHMENT

Time-series of N and P in-stream loads calculated from monitoring data highlight the inter-annual variations and show a significant decreasing trend in TP since 1994 (Fig. 4a). The most recent three-year (2011–2013) averages of these monitoring data are 5715 and 71t yr^{-1} for TN and TP, respectively (Fig. 4a, 4b). The SLAM results calculated for the catchment draining to this monitoring point compared closely (within 1% error margin) with these three-year averages.

LOAD APPORTIONMENT BY SECTOR: SUIR CATCHMENT

The SLAM results for the Suir catchment (Table 3 and Table 4) showed that pasture was the dominant source of nitrogen (78%), whereas pasture and waste-water discharges were equally dominant sources of phosphorus (35% each). The total catchment TP loads were biased by the large contribution from the Waterford agglomeration (33t yr^{-1} TP) at the mouth of the catchment, which is equivalent to 26% of the total estimated TP losses. Within the Suir catchment, there were large variations in the percentage contributions from direct discharges for phosphorus between subcatchments. These ranged from 1% to 90% and reflect the population distribution in the catchment.

LOAD REDUCTION ASSESSMENT: SUIR CATCHMENT

Using the simple annual average load calculation (Fig. 2), the total load reduction required to reduce the average annual concentration of P below the threshold for 'Good' of 0.035mg l^{-1} was calculated as 8.4t yr^{-1} of TP. This is equivalent to a reduction of 7% of the total P load emissions from the catchment. However, these load reduction targets

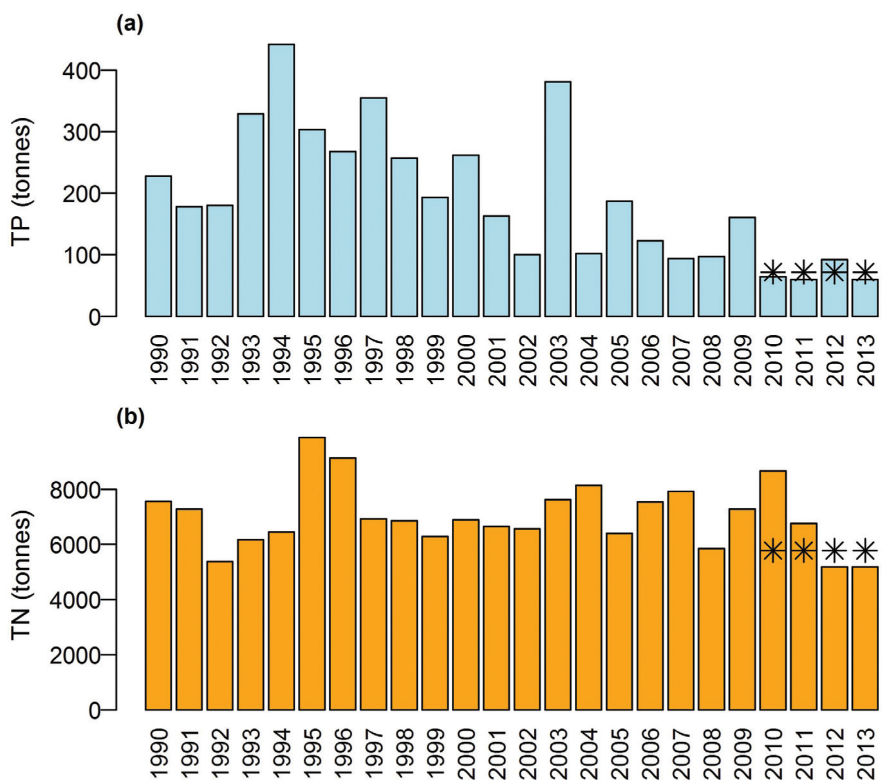

Fig. 4—Annual loads of (a) total phosphorus (TP) and (b) total nitrogen (TN) from 1990 to 2013 in the Suir catchment calculated from monitoring data. Asterisks indicate the SLAM modelled TN and TP loads, which represent recent average annual conditions and do not include inter-annual variability.

are confined to 13% of the catchment area (22 water bodies), with 5% of this area (eight water bodies) estimated to require reductions of $>0.5 t\ yr^{-1}$ of TP (Fig. 1). In fact, the majority of the required load reductions (55%) are limited to an area of only $39 km^2$ (two water bodies) in two of the 29 subcatchments (Ara_SC_010 and Fishmoyne_SC_010).

SCENARIO ANALYSIS: AHERLOW EXAMPLE

The subcatchment area of two water bodies, Aherlow 10 and 20 (pink outlined area in subcatchment 16_20, Fig. 1) was selected for assessment of possible reduction measures. The area's land cover is predominantly pasture, with two wastewater treatment plants servicing population equivalents (p.e.) of 465 and 46, both with primary treatment. The results from the load reduction assessment and scenario analyses were as follows.

- *Step 1:* Using Eq. 2, monitoring data indicated that a load reduction of $150 kg\ yr^{-1}$ is required to reduce the P concentration to below the WFD 'Good' threshold of $0.035 mg\ l^{-1}$.
- *Step 2:* For this subcatchment area, the SLAM results predicted 18% lower loads compared to monitoring data. As discussed above, these load differences may be due to model or input data inaccuracies or an unknown nutrient source not accounted for in the model. Hence, local knowledge and investigative assessments would be required to adequately assess the local conditions.
- *Step 3:* Assuming that investigative assessment was undertaken and the SLAM results are representative of the loads in the subcatchment, the total load reduction required is equivalent to 58% reduction from wastewater emissions or a 13% reduction in losses from pasture. The following three scenario options were identified.

1. *Improvement of wastewater treatment:* The total load from the two wastewater treatment plants is calculated as $226 kg\ yr^{-1}$. Using nutrient reduction factors dependent on treatment level (OSPAR 2000), upgrading this plant to secondary or tertiary treatment would result in a load reduction of approximately 70 or $190 kg\ yr^{-1}$, respectively. Hence, scenario (1) would be able to achieve the required load reduction of $150 kg\ yr^{-1}$, if it was deemed economically feasible.
2. *Reduction in P losses from pasture:* The evaluation of measures to achieve the 13% from pasture, i.e. a reduction from ~ 0.2 to $0.17 kg\ ha^{-1}\ yr^{-1}$, is more difficult compared to scenario (1), as limited data are available in Irish catchments on the performance of targeted measures. In the Aherlow 10 and 20 subcatchment area, the transfer pathways for P were identified by the CCT as predominantly near surface (Fig. 3). These high-risk areas include pasture land cover with poorly drained soils. Based on reported removal rates in the literature (see Data and Methods section for references), enhancement of buffer strips was a viable potential measure for the agricultural sector to achieve the required load reduction of $150 kg\ yr^{-1}$ in this subcatchment. However, the success would depend on the applicability to local conditions and the specific design, implementation and maintenance details. Hence, any additional measures should be evaluated locally in liaison with local authorities and land owners.
3. *Combined reductions from wastewater and pasture:* Cost-effectiveness analysis is integral in the implementation of the WFD (Joyce and Convery 2009). Given the capital investment required for scenario (1) and the uncertainty of scenario (2), an evaluation by multicriteria decision analysis (not undertaken in this study) may identify an optimal acceptable solution that is a combination of measures targeted at wastewater emission and losses from pasture.

DISCUSSION

MODEL PERFORMANCE AND UNCERTAINTIES

Where a subcatchment is identified as being at risk of deterioration or not achieving Good status due to nutrients, source load apportionment can be used to rank the dominant sources of nutrient emissions. Discrepancies between the modelled and monitored loads may be from:

1. model structure, parameter or input data inaccuracies,
2. errors in estimating the annual nutrient load from monitoring data; or
3. unknown source(s) not accounted for in the model (e.g. accidents, non-compliant farmyards).

The SLAM results compared closely to the 2011–2013 TN and TP average loads. However, a major limitation of the SLAM framework is that results only represent average conditions and therefore the model does not reflect inter-annual hydrological and anthropogenic variability. Hence, the results underestimate loads compared to previous years, e.g. the 2009–2011 averages of 7570 and $95 t\ yr^{-1}$ for TN and TP, respectively, as the input data do not represent earlier management practices.

Table 3—Nitrogen load apportionment (SLAM v. 2.04) in the Suir catchment.

Water body	*Code*	*Area (km²)*	*Wastewater*	*Other licensed discharges*	*Diffuse urban*	*Septic tank systems*	*Pasture*	*Arable*	*Forestry*	*Peat*	*Deposition on water*	*N total (t yr⁻¹)*
Fishmoyne_SC_010	16_5	68	2%	0%	0%	2%	92%	2%	1%	0%	0%	134
Multeen[East]_SC_010	16_4	83	1%	0%	0%	2%	88%	0%	8%	0%	0%	105
Ara_SC_010	16_13	84	14%	0%	0%	3%	78%	3%	2%	0%	0%	216
Clashawley_SC_010	16_6	167	5%	0%	0%	2%	81%	12%	1%	0%	0%	389
Suir_SC_010	16_22	229	1%	14%	0%	1%	78%	3%	2%	0%	0%	518
Suir_SC_020	16_2	64	14%	1%	1%	3%	72%	10%	1%	0%	0%	173
Suir_SC_030	16_28	163	1%	0%	0%	2%	93%	2%	2%	0%	0%	289
Suir_SC_040	16_21	210	1%	0%	0%	2%	82%	11%	3%	2%	0%	400
Suir_SC_050	16_10	139	3%	0%	0%	2%	83%	11%	1%	0%	0%	399
Suir_SC_060	16_18	108	0%	0%	0%	2%	93%	2%	3%	0%	0%	228
Suir_SC_070	16_11	64	0%	0%	0%	2%	91%	7%	0%	0%	0%	193
Suir_SC_080	16_9	137	0%	0%	0%	1%	77%	21%	0%	0%	0%	439
Suir_SC_150	16_23	144	1%	1%	0%	2%	70%	26%	1%	0%	0%	435
Suir_SC_090	16_20	206	1%	0%	0%	2%	89%	2%	5%	2%	0%	351
Suir_SC_100	16_14	86	4%	16%	0%	1%	56%	18%	3%	1%	0%	251
Suir_SC_120	16_8	117	11%	0%	1%	2%	54%	32%	0%	0%	0%	385
Suir_SC_160	16_12	78	1%	2%	0%	2%	70%	24%	1%	0%	0%	214
Suir_SC_140	16_16	112	0%	0%	0%	3%	80%	9%	7%	0%	0%	207
Suir_SC_110	16_25	97	0%	0%	0%	1%	67%	21%	7%	3%	0%	149
Suir_SC_130	16_3	141	1%	0%	0%	2%	82%	4%	8%	3%	0%	164
Duag_SC_010	16_1	66	3%	0%	0%	1%	86%	0%	7%	2%	0%	110
Tar_SC_010	16_26	92	0%	0%	0%	2%	87%	4%	3%	3%	0%	138
Anner_SC_010	16_24	140	1%	0%	0%	2%	85%	8%	3%	1%	0%	253
Lingaun_SC_010	16_15	92	5%	0%	0%	1%	78%	13%	2%	0%	0%	176
Pil_SC_010	16_27	134	1%	4%	0%	2%	82%	10%	1%	0%	0%	318
Blackwater[Kilmacow]_SC_010	16_29	206	1%	0%	1%	2%	85%	9%	3%	0%	0%	420
Clodiagh[Portlaw]_SC_010	16_17	132	2%	0%	0%	1%	86%	7%	3%	0%	0%	291
Dawn[River]_SC_010	16_7	81	0%	0%	0%	2%	90%	6%	2%	0%	0%	251
Williamstown_SC_010	16_19	111	37%	0%	2%	2%	49%	9%	0%	0%	0%	429
Total		3553	4.4%	1.7%	0.4%	1.8%	78.0%	11.0%	2.2%	0.5%	0.1%	8024

Table 4—Phosphorus load apportionment (SLAM v. 2.04) in the Suir catchment.

Water body	*Code*	*Wastewater*	*Other licensed discharges*	*Diffuse urban*	*Septic tank systems*	*Pasture*	*Arable*	*Forestry*	*Peat*	*Deposition on water*	*P total* ($t\ yr^{-1}$)
Fishmoyne_SC_010	16_5	8%	0.0%	3%	2%	75%	2%	10%	0%	0%	1.4
Multeen[East]_SC_010	16_4	7%	3.1%	1%	2%	50%	0%	36%	1%	0%	2.1
Ara_SC_010	16_13	45%	0.9%	6%	1%	35%	1%	11%	0%	0%	2.8
Clashawley_SC_010	16_6	19%	0.3%	3%	2%	62%	8%	5%	1%	0%	4.1
Suir_SC_010	16_22	7%	16.3%	5%	1%	51%	2%	14%	4%	0%	6.0
Suir_SC_020	16_2	48%	3.9%	8%	1%	31%	3%	5%	0%	0%	2.4
Suir_SC_030	16_28	11%	0.0%	0%	2%	70%	1%	15%	1%	0%	3.4
Suir_SC_040	16_21	11%	0.0%	4%	1%	45%	4%	21%	13%	0%	4.9
Suir_SC_050	16_10	25%	0.2%	6%	2%	51%	6%	10%	0%	0%	2.8
Suir_SC_060	16_18	4%	0.2%	0%	2%	64%	1%	29%	0%	0%	2.1
Suir_SC_070	16_11	0%	0.8%	10%	4%	77%	7%	1%	0%	0%	0.8
Suir_SC_080	16_9	4%	0.2%	6%	2%	65%	21%	1%	0%	0%	2.0
Suir_SC_150	16_23	10%	4.6%	5%	2%	50%	16%	9%	4%	0%	3.0
Suir_SC_090	16_20	6%	0.2%	0%	3%	53%	1%	27%	10%	0%	5.4
Suir_SC_100	16_14	5%	48.8%	4%	1%	17%	6%	16%	3%	0%	4.9
Suir_SC_120	16_8	38%	0.1%	20%	2%	26%	15%	1%	0%	0%	4.2
Suir_SC_160	16_12	5%	39.5%	6%	1%	29%	9%	8%	2%	0%	2.8
Suir_SC_140	16_16	2%	0.0%	4%	3%	41%	3%	45%	1%	0%	2.6
Suir_SC_110	16_25	1%	0.0%	0%	3%	41%	8%	31%	16%	0%	3.1
Suir_SC_130	16_3	5%	0.0%	0%	2%	49%	2%	30%	12%	0%	4.0
Duag_SC_010	16_1	21%	0.0%	1%	0%	39%	0%	29%	9%	0%	2.4
Tar_SC_010	16_26	3%	0.0%	0%	2%	56%	2%	17%	20%	0%	2.3
Anner_SC_010	16_24	11%	0.1%	0%	2%	64%	5%	15%	4%	0%	3.6
Lingaun_SC_010	16_15	24%	0.5%	3%	1%	50%	7%	11%	3%	0%	2.3
Pil_SC_010	16_27	17%	5.9%	7%	5%	46%	8%	10%	0%	0%	3.6
Blackwater[Kilmacow]_SC_010	16_29	6%	3.0%	11%	3%	50%	4%	23%	0%	0%	5.0
Clodiagh[Portlaw]_SC_010	16_17	23%	0.0%	1%	1%	48%	3%	20%	3%	0%	3.4
Dawn[River]_SC_010	16_7	3%	1.4%	2%	4%	66%	3%	18%	0%	2%	1.7
Williamstown_SC_010	16_19	90%	0.2%	5%	0%	3%	0%	0%	0%	0%	36.5
Total		35.4%	4.2%	4.7%	1.5%	35.0%	3.8%	12.3%	3.1%	0.1%	125.7

There are greater uncertainties in the modelled P estimates compared to the N due to the episodic nature of P losses and associated difficulties in monitoring in-stream loads and hence in developing and testing models. Furthermore, underestimates and overestimates that occur within the subcatchments are averaged at the catchment scale, potentially masking prediction errors. Losses of nutrients vary depending on farm management practices (e.g. Doody *et al.*, 2014), and local hydrological flow paths (e.g. Ryan and Finnan 2015; Thomas *et al.*, 2016), which are not captured by the national assessment tool detailed here. Hence, in order to identify significant pressures in a water body, all available evidence should be considered in conjunction with the SLAM results, including knowledge from local authorities and investigative assessments gathered through the WFD characterisation process.

In the two subcatchments of the Suir requiring the largest load reduction, the SLAM results underestimate the measured annual nutrient loads. By interpreting these results with monitoring data, ecological assessments and additional local knowledge where feasible, areas can be identified for further investigative assessment of unknown sources of nutrients or non-compliance (Daly *et al.*, 2016).

PRESSURE/INPUT DATA

As the results of the SLAM framework are not calibrated to monitoring data, the performances of the submodels are dependent on high-quality input data. Close liaison with the data owners and local authorities is vital to ensure meaningful results. Quality control of the national databases is time-intensive and any remaining errors or omissions will impact on results. However, the synthesis of the large number of national databases by the framework results in a powerful tool for assessing the relative fate of nutrients in the Irish water environment.

The heterogeneity of the urban environment makes the identification of export coefficients more difficult compared to other land cover types (White and Hammond 2006; 2009). Several sources may not be captured by this approach, including localised practices and incidental spills. For example, a review of sewer misconnections by Ellis and Butler (2015) found typical rates of misconnection in the UK and Ireland ranging from 1% to 6%. Recent investigative assessments undertaken separately by Dublin City Council and South Dublin County Council showed that 8% of houses were misconnected. Hence, sewer misconnections are a potential source of nutrient losses from urban areas not currently included in the model. Phosphate dosing of mains water to prevent lead solvency from pipes is an additional source of P that may be introduced in the coming years and, if so, the model will be able to take this additional load into consideration.

MONITORING/ASSESSMENT DATA

It is clear from high-resolution monitoring that sub-hourly sampling is required in most catchments to capture temporal dynamics, particularly for P and suspended sediment (e.g. Campbell *et al.*, 2015; Mellander *et al.*, 2015). Many studies have estimated the uncertainty around load estimations calculated with less frequent data for both P and N (Johnes, 2007; Cassidy and Jordan, 2011; Jiang *et al.*, 2014). Cassidy and Jordan (2011) showed that most load estimates approaches produced a systematic underestimation of total annual nutrient P load, which is attributed to the flashy nature of the study catchment and high P export during short-duration high-flow events.

High-resolution data are typically only available for relatively small, headwater study catchments, for example the Agricultural Catchments Programme (Fealy *et al.*, 2010) study catchments which cover 0.001% of the country. As catchment scientists and managers need to make the most of available data in order to base decisions on the best available information, load calculations using less frequent data are still necessary. Comparison of estimation methods for N and P show that where strong correlations are present between flows and concentrations, the preferred method to estimate in-stream loads is to use regressions to interpolate concentration values on days without concentration data. This method is particularly suitable for sediments, particulate P and Total P (Quilbé *et al.*, 2006; Jiang *et al.*, 2014), although hysteresis may complicate relationships (Kelly-Quinn *et al.*, 2014). Methods using catchment models, e.g. SMART (Mockler *et al.*, 2016) can be used to investigate the contribution of individual high-flow events to annual loads in each catchment. Alternatively, when there are a large number of flow data but few measured concentrations and no correlations are present, the flow-weighted or ratio estimator method (Beale 1962) was preferred, as is used in this study.

The SLAM results are initially tested against loads calculated at the OSPAR monitoring station. In addition, in-stream load estimates are used for selected subcatchments to check the performance of the model. However, nutrient monitoring data are typically limited to twelve or fewer samples per station per year, particularly for first- and second-order streams which make up the majority (77%) of Irish river channels (McGarrigle, 2014). Given the uncertainties surrounding the load reduction assessment and quantification of achievable load reductions from measures, the MRP EQS of

$0.035 mg\ l^{-1}$ P was selected as the target TP value. This provides a margin of error and improves the probability of achieving 'Good' WFD status under the programme of measures selected.

SUPPORTING CATCHMENT MANAGEMENT

Iterations of learning and improved understanding are integral to both integrated catchment management and model development. As the second cycle of the WFD progresses, two contrasting sources of knowledge are providing a deeper understanding of catchment dynamics in Ireland:

1. the gathering of local knowledge and land management practices from local authorities and investigative assessments through the WFD characterisation process;
2. research into catchment processes, including detailed headwater and field-scale investigations.

Daly *et al.* (this issue) outline the challenging WFD objectives, which include preventing deterioration in water-body status, protecting satisfactory (Good and High status) water bodies and restoring substandard water bodies to good status. The SLAM framework aims to integrate all relevant information to support national management and policy by providing scientific justification needed to defend decisions in relation to measures. The framework and submodels are under continuous development, and aim to reflect new research findings both at home and internationally.

Diffuse agriculture was identified as the dominant source of N losses across all subcatchments of the Suir. There was a greater variation in main sources of P emissions, which included wastewater, industrial discharges, pasture, arable and forestry. The large wastewater discharge at the catchment outlet contributes 26% of the total P load emissions to water, However, headwater subcatchments (16_5 and 16_13) were identified as requiring the majority of the P load reductions required under the WFD.

The SLAM results provide an indication of the dominant sector(s) contributing nutrient pollution in subcatchments. These potential significant pressures can be considered alongside other available data and knowledge from catchment mangers and local authorities gathered during the WFD characterisation process. This type of desk-based assessment can inform any additional field investigation that needs to be undertaken (e.g. McGarrigle 2014) to identify the significant pressure(s) that require mitigation measures.

Potential management strategies can be assessed using the SLAM results to facilitate the targeting and prioritisation of measures at subcatchment and catchment scales. The scenario analysis presented for the Aherlow illustrates how load apportionment can provide input information for multiple-criteria decision analysis, and hence can be used to inform the selection of suitable measures in the WFD River Basin Management Plans. These analyses are being developed to predict what water quality changes may occur under projections of increased agricultural productivity and population.

CONCLUSIONS

The SLAM framework produces source load apportionment of N and P to support catchment characterisation and the identification of potential mitigation measures. The flexible framework is designed to support independent development of the individual sector submodels. Work is ongoing to improve aspects of the model to ultimately produce detailed national load apportionment results for Ireland.

In the Suir catchment, the dominant sources of N emissions to water were from pasture. Both pasture and wastewater discharges were dominant for P, with considerable variations in relative contributions between subcatchments. Identification of the locally dominant sources of emissions from the SLAM results can support catchment characterisation and management. Furthermore, simple load reduction scenario analyses, as demonstrated in this paper for the Aherlow subcatchment, can support the selection of potential measures.

The load assessment presented here is heavily reliant on both pressure and in-stream monitoring data and highlights the value of an integrated analysis of the national datasets available in Ireland. Tracking changes over time with long-term national monitoring data and modelling is vital to improve understanding of catchment dynamics (Ní Longphuirt *et al.* and O'Boyle *et al.*, this issue), and to support the iterative improvement of the integrated catchment management process.

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APPENDIX: SUMMARY OF PATHWAY CCT FACTORS

The nutrient loads to surface water from diffuse agricultural losses were modelled by the Catchment Characterisation Tool (CCT) (Archbold *et al.*, 2016 and references therein). Nutrient losses of nitrate (N) or phosphorus (P) from diffuse agriculture ($DiffAgri_{N,P}$) are calculated from Eq. 8, using estimates of loads leached from the soils for N or P ($Leached_{N,P}$ kg yr^{-1}) and the following near surface pathway ($\alpha_{N,P}$), groundwater pathway ($\beta_{N,P}$), and groundwater bedrock pathway factors (φ_N) for N or P (Tables A1–A5).

Table A1—Nitrate near-surface pathway factors.

Subsoil permeability	*N near-surface factor*
Low	0.2
Low and likely to have land drains	0.7
Moderate	0.55
High	1
N/A	0.95
Water/Lake/Rock	0.95

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—Nitrate groundwater pathway factors.

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

Table A4—Phosphorus groundwater pathway factors.

Depth to bedrock	*P subsoil factors*
X: Extreme (0–1m and near karst features)*	0.4
E: Extreme (1–3m)*	0.15
3–5m*	0.05
5–10m*	0.02
> 10m*	0.01
*Exception: peat subsoils	0.9

Table A5—Nitrate groundwater bedrock pathway factors.

*Bedrock unit**	*Transport factor*
Unit 1a, Unit 1b	0.65
All other bedrock units	1
*As defined by the GSI Denitrification Map.	

NEAR SURFACE PATHWAY FACTORS FOR N AND P

Delivery of nitrate to the near surface pathway is determined by factors depending on the subsoil permeability as the nutrient tends to move through the subsoils before arriving at the surface water receptor. It is assumed that the nutrient moves a certain distance before reaching the surface water receptor. Delivery through overland flow is considered to be negligible (incidental losses are not considered in this annual average model). A map of the possible location of land drains was produced (Mockler *et al.*, 2014), which indicates a preferential delivery pathway for nitrate in low permeability subsoils.The P delivery factors were originally based on the PIT model (Heathwaite *et al.*, 2003) coefficients (see Packham *et al.*, 2013). These factors were amended to reflect Irish conditions following assessment and checking with EPA monitoring data.

GROUNDWATER PATHWAY FACTORS FOR N AND P

Groundwater pathway factors for nitrate were determined following a literature and expert elicitation review, as described in Packham *et al.* (2013). The factors for N vary with subsoil permeability and depth to bedrock (Table A3), both available as maps from GSI. The P factors vary by depth to bedrock, except in areas of peat soils (Table A4).

BEDROCK TRANSPORT FACTORS FOR N AND P

Attenuation of N in bedrock is related to aquifer bedrock units with the potential for denitrification, as represented in the GSI denitrification map. Units 1a and 1b in A5 are those identified as denitrifying bedrock, mostly due to the presence of pyrite. No attenuation in bedrock is assumed for P, hence the bedrock transport factor is 1.