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Catchment Models and Management Tools for Diffuse Contaminants (Sediment, Phosphorus and Pesticides): DiffuseTools Project

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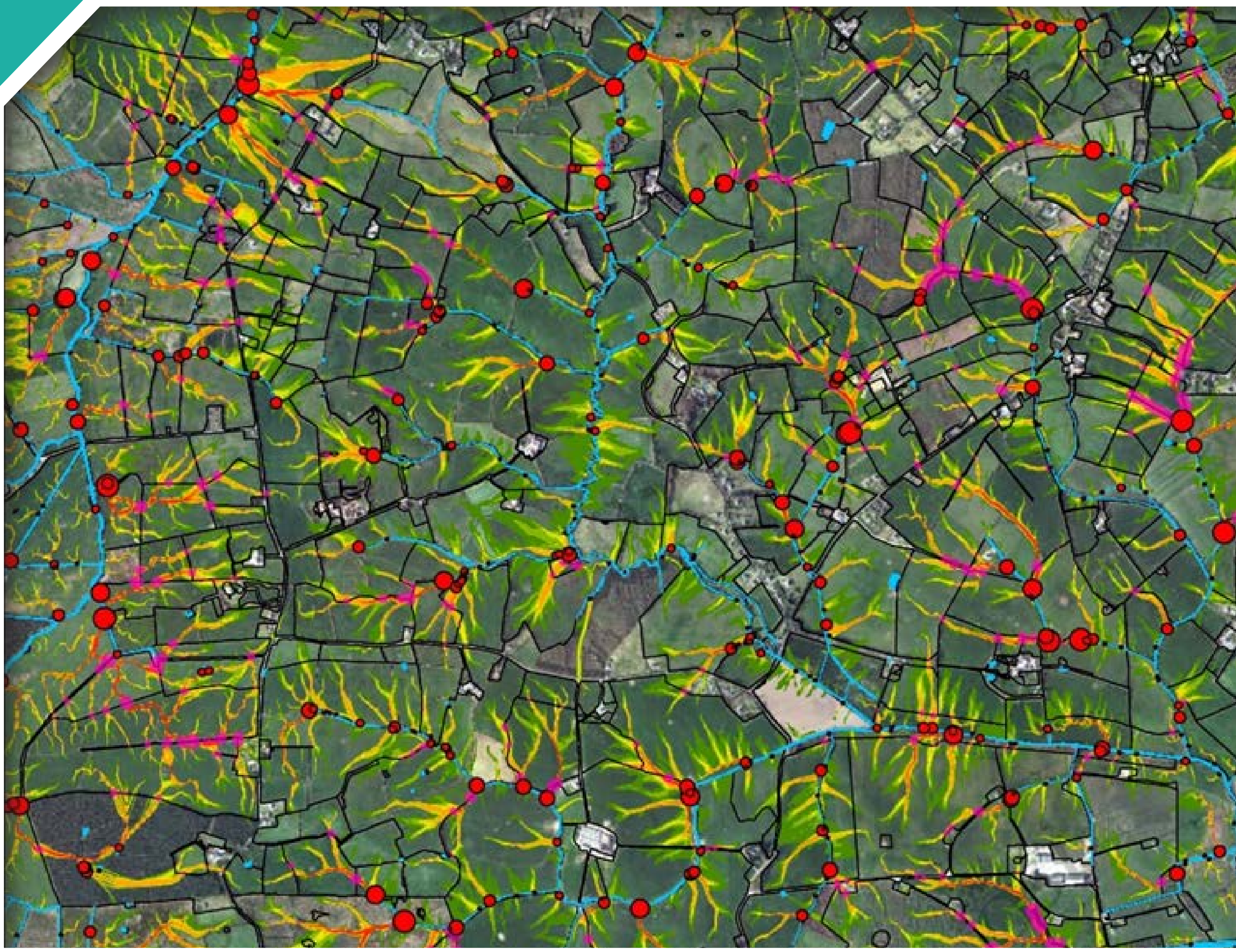

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Government of Ireland

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EPA RESEARCH PROGRAMME 2021–2030

Catchment Models and Management Tools for Diffuse Contaminants (Sediment, Phosphorus and Pesticides): DiffuseTools Project

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EPA Research Report

Supplementary Technical Reports on GIS modelling and pesticides referenced in this report are available on request from EPA Research (research@epa.ie) or to download (https://eparesearch.epa.ie/safer/)

Prepared for the Environmental Protection Agency

by

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The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

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Executive Summary

Agricultural pollution continues to be a major cause of eutrophication of waterbodies and water quality degradation in Ireland and internationally, with the success of mitigation measures hampered by the diffusivity of pollution sources and pathways. Ireland must meet international water quality obligations set by the EU Water Framework Directive (WFD), with the aim of achieving “good ecological and chemical status” in all and high status in some High Status Objective (HSO) waterbodies by 2027. For surface waters, S.I. 272 sets out the required standards. Good ecological status is assessed using environmental quality standards, including an annual mean unfiltered reactive phosphorus (P) concentration not exceeding 0.035 $mg\,l^{-1}$ in Irish rivers. The WFD includes provisions from the Nitrates Directive that aimed to protect waterbodies from agricultural nitrogen and P pollution by implementing a Nitrates Action Programme and S.I. 605, Good Agricultural Practice for Protection of Waters Regulations. The most recent Environmental Protection Agency (EPA) water quality assessment over the period 2013–2018 found that 52.8% of surface waterbodies assessed are of good or high ecological status (a decline of 2.6% compared with 2010–2015), with the remaining (47.2%) being of moderate, poor or bad ecological status. This has given rise to concerns that the current mitigation measures, which in the past have been heavily source focused, do not go far enough, and has led to renewed interest in the development of decision support tools (DSTs) for P loss management. These can spatially map P, sediment and pesticide losses from agricultural land to waterbodies at high resolution and are needed by farmers, catchment managers, policymakers and water agencies to improve cost-effective targeting of pollution mitigation measures. The University College Dublin (UCD)-led DiffuseTools project has developed such modelling tools suitable for implementation nationally to estimate P and sediment losses from diffuse sources and their delivery points to surface waters using the latest data, science and geographical information systems (GISs).

Here we develop high-resolution quantitative modelling to (1) support better targeting and prioritising of diffuse pollution mitigation measures and thereby increase their cost-effectiveness; (2) inform engineering designs (flow volumes) and interventions required to meet WFD targets; (3) improve functional land management and inform farm practices (e.g. where to spread excess fertiliser); and (4) facilitate modelling the effects of climate change. Note that this work focuses on surface runoff transport, and subsurface pathways are not considered here.

For dissolved P, the DST used an ArcGIS ModelBuilder framework to semi-automate workflows and facilitate future updates and focused on legacy soil P sources, which represent the main long-term threat to water quality, and surface hydrological pathways, which are the dominant control of diffuse P losses in Irish catchments. To model soil P sources nationally, a scenario analysis was used, whereby all agricultural soils were assumed to have soil Morgan P concentrations ($mg\,l^{-1}$) in one of the four soil P index classes. For each scenario, surface runoff P concentrations were modelled using a relationship with soil P concentrations derived from an existing Irish literature review. To model surface runoff volumes, the hybrid soil moisture deficit model, version 2 (HSMDv2), was developed further to calculate the excess water above field capacity and disaggregate lumped surface runoff and percolation depths. This new model, HSMDv3, was then applied nationally on the Irish Centre for High-End Computing (ICHEC) supercomputer using Met Éireann’s weather and Reanalysis (MÉRA) datasets to model 30-year annual average surface runoff and percolation depths (1981–2010), and the results were spatially joined to a new Soils Hydrology Ireland soil drainage class map. Runoff volume accumulation downslope was then modelled using a hydrologically corrected 5-m-resolution NEXTMap digital elevation model (DEM). Annual surface runoff from agricultural soils was then calculated for each 5-m grid cell, and at field, upslope drainage area and WFD river sub-basin scales to suit different stakeholders, as were estimates of P export and accumulation downslope to waterbodies. The locations of breakthrough lines and delivery lines, where surface runoff pathways cross field boundaries and then enter waterbodies, were predicted nationally, and attributes calculated including global positioning system (GPS) coordinates, area/length, surface runoff volumes and indices of P loss, to assist on-the-ground targeting and prioritisation of measures to reduce P inputs to surface waters. Only small edges of field boundaries (nearer the source) and riparian zones (at the receptor) need special targeting. These maps were generated at different scales to suit the different needs of a variety of users/stakeholders as well as the EPA. These include catchment managers, Local Authority Waters Programme (LAWPRO) scientists, Agricultural Sustainability Support and Advisory Programme (ASSAP) staff and farmers, as well as all those concerned with river basin management plans, agri-environmental schemes and targeted protection of special ecosystems or species, such as the freshwater pearl mussel.

To model sediment losses and particulate P loss risk, the SCIMAP (Sensitive Catchment Integrated Modelling Analysis Platform) was applied nationally using the 5-m-resolution DEM. In addition, a catchment-scale approach was tested using a simplification of the INCA model (SimplyP).

The project reviewed the published literature on pesticide export from agricultural land to rivers, including numerical models. Numerical modelling of exceedances of pesticide concentration limits in rivers is challenging, regardless of the modelling approach. The project analysed pesticide data from multiple Irish sources and concluded that occurrences of high herbicide concentrations in rivers are intermittent and not exclusively associated with high water fluxes but may also be due to issues in the preparation and application of the pesticides.

Recommendations:

1. Riparian measures to mitigate P and sediment delivery to waterbodies should prioritise the major delivery and breakthrough locations identified in this research, with local solutions appropriate to the magnitude of the delivery.
2. Cost–benefit analysis of available options is needed to inform decisions.
3. Research is required to quantify the mobilisation of soil P by quickflow.
4. Edge-of-field measurements of surface runoff and P concentrations are needed, targeted at modelled surface runoff pathways, delivery points and non-hydrologically sensitive areas.
5. Further research into the hydrological connectivity of flow pathways is needed, specifically the slow-down, impediment and reinfiltration of overland flow on shallow slopes in heavily vegetated areas or at flow diversions.
6. For pesticides and sediment, there is a need for a frequent monitoring regime that is designed to capture the full temporal and spatial variability to better estimate loads, and to support understanding and modelling of both the episodic and baseline concentrations and loads. This must include subsurface monitoring for chemicals transported via below-ground pathways.

1 Introduction

1.1 Background

Globally, agriculture faces significant challenges in achieving environmentally sustainable intensification that balances the competing demands of intensifying food production to meet global food security while reducing environmental emissions of nutrients [nitrogen and phosphorus (P)], sediments, pesticides and other pollutants (Godfray and Garnett, 2014). Intensive agriculture relies on the use of nutrient inputs from chemical fertilisers and pesticides to drive production, but these are costly, non-renewable resources with often poor use efficiencies (Murphy *et al*., 2019; Thomas *et al*., 2020a). Most countries, including Ireland, import all P fertilisers from a finite mineral resource concentrated in just a few countries worldwide, with global peak phosphate rock production disputably predicted to occur in as little as 30 years (Li *et al*., 2019). Excessive losses of agricultural pollutants from agricultural land to surface waters from both point and diffuse sources through hydrological pathways (surface, subsurface, groundwater and preferential) or airborne emissions can cause eutrophication of waterbodies, algal blooms, hypoxic or anoxic events, deterioration of water quality, shifts in ecosystem community structure, reduced biodiversity, loss of aquatic habitats, sedimentation and turbidity, and physiological/behavioural changes in aquatic life (Bol *et al*., 2018). These effects have serious implications for ecosystem services such as drinking water, industry, fisheries, aesthetics and recreation (Bilotta *et al*., 2007). Furthermore, agricultural land can be affected by soil erosion, lower soil fertility, pest resistance and reduced biodiversity. Policymakers are, therefore, attaching greater priority to the efficient management of natural resources and improving the environmental performance of farm systems (Czyżewski *et al*., 2021).

Ambitious national growth targets for agricultural output in Ireland have been set in the decade-long Food Wise 2025 government initiative (DAFM, 2015a), with a vision of smart, green growth that aims to deliver sustainable intensification (economically, environmentally and socially) supported by initiatives such as Bord Bia's Origin Green and aligned with the EU Green Deal, Farm to Fork and possible future Common Agricultural Policy (CAP) reform. Increasing productivity and efficiencies in ecosystem services while maintaining and protecting natural resources for long-term needs could be facilitated, in part, by using the latest science, technologies and big data. The dairy industry is expanding following the abolition of milk quotas in 2015, with intensive stocking rates >170 kg organic nitrogen $ha^{-1}y^{-1}$ facilitated by a Nitrates Directive derogation for Ireland (2014/112/EU), subject to additional requirements. At the same time, Ireland must meet international water quality obligations set by the EU Water Framework Directive (WFD) (2000/60/EC), with the aim of achieving "good ecological and chemical status" in all waterbodies by 2027. For surface waters, S.I. 272 European Communities Environmental Objectives (Surface Waters) Regulations 2009 (Government of Ireland, 2009) sets out the required standards. Good ecological status is assessed using environmental quality standards, including an annual mean unfiltered reactive P concentration not exceeding $0.035\,mg\,l^{-1}$ in Irish rivers. The WFD includes provisions from the Nitrates Directive (91/676/EEC) that aims to protect waterbodies from agricultural nitrogen and P pollution by implementing a Nitrates Action Programme (NAP) and S.I. 605, Good Agricultural Practice for Protection of Waters Regulations 2017 (Government of Ireland, 2017). In Ireland, the NAP is regulated by the Nitrates Regulations on a whole-of-territory basis, which provides statutory support, enforcement provisions, and a range of good agricultural practices designed to limit nitrogen and P losses from agricultural systems. These include minimum requirements for manure, slurry and silage storage, management of dirty water from yards and farm buildings, restrictions on stocking rates, fertiliser application rates/timings/locations and soil P status, implementation of nutrient management records and balances, and establishing green cover over the winter months. If Ireland breaches WFD targets as the result of agricultural intensification, it could lose its derogation status to stock livestock above the 170 kg organic nitrogen $ha^{-1}y^{-1}$ NAP threshold, which would severely hinder national policy aims.

All EU Member States need to have in place water quality monitoring programmes as part of WFD requirements. In Ireland, the most recent EPA water quality assessment, over the period 2013–2018, found that 52.8% of surface waterbodies assessed were of good or high ecological status (a decline of 2.6% compared with 2010–2015), with the remainder (47.2%) being of moderate, poor or bad ecological status (O'Boyle *et al*., 2019). Since the 2010–2015 assessment period, an overall net decline in 117 surface waterbodies (4.4%) has been observed, driven almost entirely by the water quality decline in river waterbodies since 2015, indicating increased pressures from human activities. Diffuse-source P is a major contributor to this decline. Over a quarter of monitored river sites are now seeing increasing P concentrations. Although lake water quality status is relatively stable, with some improvements, total P concentrations were found to be on the rise in over a quarter of the lakes analysed. Only 38% of transitional waters (estuaries and lagoons) are of good or better ecological status, with total P loads increasing by 31% since a low in 2012–2014, indicating increasing pressures from catchment-wide sources. A third of rivers and lakes and a quarter of estuaries are failing to meet their nutrient-based environmental quality standards. This has given rise to concerns that the current mitigation measures, which in the past have been heavily source focused, do not go far enough, and has resulted in renewed interest in the development of decision support tools (DSTs) for P loss management (Drohan *et al*., 2019).

Significant pressures identified as the causes included diffuse P losses from poorly drained agricultural soils, as well as wastewater discharges, excess sediment runoff, hydromorphological alterations to habitats (e.g. land drainage and channel maintenance) and forestry (O'Boyle *et al*., 2019). Pesticides and herbicides are also having an impact in certain areas, with some drinking water supplies in Ireland affected by pesticide exceedances of regulatory limits. As a result, the National Pesticide and Drinking Water Action Group is working with stakeholders to raise awareness of best practices and the requirements of the Irish Nitrates Action Programme, the Sustainable Use of Pesticides Directive (2009/128/EC) and the Good Agricultural Practice regulations. Key to water quality improvements is the establishment of the Local Authority Waters Programme (LAWPRO), which carries out local on-the-ground catchment assessments and promotes the implementation of targeted mitigation measures to improve water quality at a local level, as well as the Agricultural Sustainability Support and Advisory Programme (ASSAP) run by Teagasc and dairy cooperatives, which advise farmers on measures they can take to protect watercourses. These programmes focus on Priority Areas for Action (PAAs) identified in the River Basin Management Plan (RBMP; currently 2018–2021), and efforts in the previous WFD cycle's PAAs have shown encouraging water quality trends as a result, including net improvements in 81 waterbodies (16.7%). However, despite actions to improve water quality across all pressure types and sectors undertaken by these and other public bodies, significant challenges remain.

One of the key recommendations of the EPA report *Water Quality In Ireland 2013–2018* (O'Boyle *et al*., 2019) to improve Ireland's water quality is implementing the right measures in the right places. Diffuse agricultural pollution is difficult to manage because of the complex diffusivity of pollutant sources and pathways to receptors, which are spatiotemporally heterogeneous (do not occur uniformly in the landscape) and controlled by nutrient management, land management, livestock behaviour, soils, hydrogeology, topography and weather patterns (Mellander *et al*., 2018). As a result, mitigation has been much less successful than in the case of point-source controls (Jarvie *et al*., 2013). Agri-environment policies and schemes such as the Green Low-carbon Agri-environment Scheme (GLAS) (DAFM, 2015b) have historically adopted the approach of blanket implementation of mitigation measures, such as riparian buffer zones, across a farm, regardless of actual localised pollution risk. This reduces the cost-effectiveness of measures compared with targeted approaches (Doody *et al*., 2012; Qiu and Dosskey, 2012; Ó hUallacháin, 2014; Thomas *et al*., 2016a) and increases the amount of agricultural land that would be taken out of production, which can dissuade farmers from adopting such measures (Buckley *et al*., 2012).

Efforts must therefore focus on improving the delineation of the critical source areas (CSAs) of diffuse agricultural pollution, currently done in Ireland with the EPA's Pollutant Impact Potential (PIP) map (Packham *et al*., 2020). CSAs are areas where disproportionately large amounts of pollutants are mobilised and transported via hydrologically connected pathways to waterbodies (Pionke *et al*., 2000). These can be relatively small areas of the landscape but export disproportionate amounts of nutrients, suspended sediments, pesticides and/or other pollutants during storm events (Thomas *et al*., 2016b; Djodjic and Markensten, 2019). Targeting site-specific mitigation measures and best management practices at these vulnerable, high-risk areas is, therefore, more environmentally efficient and cost-effective than blanket regulatory measures that ignore the inherent spatial and temporal heterogeneities of diffuse pollution (Doody *et al*., 2012). There is, therefore, a need for robust DSTs that spatially model and map CSAs of diffuse pollution, provide cost–benefit analyses and facilitate targeting of measures by farmers, catchment managers and other stakeholders at key locations in the landscape (Cole *et al*., 2020).

1.2 Aims, Objectives and Project Outline

For the reasons outlined in section 1.1, this study aims to use the latest science, big data and geographical information system (GIS) techniques to develop tools for modelling diffuse source pollution that can improve mapping of diffuse agricultural pollution for P, sediments and pesticides, quantify P loads and precisely locate where diffuse pollutants enter waterbodies, to more cost-effectively target mitigation measures. Modelling the specific components of diffuse P losses is more appropriate than lumping together the losses of different forms (particulate or dissolved) and from different sources (desorption of soil P, soil erosion, dissolved P from fertiliser/manure) and transport pathways (surface runoff, subsurface and preferential, e.g. tile drains). Otherwise, models are unable to differentiate between the ways P is released to water travelling by different pathways or between the ways that different forms of P are transported, and cannot identify or mitigate the riskiest forms/sources/pathways (Reid *et al*., 2018). Specialised models that focus on specific forms, sources or pathways of P losses of most concern or priority would reduce data demands and complexity and be more policy applicable (Thomas *et al*., 2016b). The focus of this study was the remobilisation of legacy soil P because, as well as hindering effective mitigation, it represents the main long-term threat to water quality in Ireland and internationally (Jarvie *et al*., 2013). This is in contrast to incidental transfers of fertiliser P, which have been somewhat mitigated through legislative measures, such as improved slurry storage and closed periods for spreading in winter (Shore *et al*., 2016). The study also focused on surface hydrological pathways only, which are the dominant method of controlling diffuse P losses in Irish catchments (Mellander *et al*., 2015). As subsurface flow pathways will dominate in well-drained soils and karst areas, the study also mapped percolation depths and subsurface flow (the difference between percolation and Geological Survey Ireland (GSI) groundwater recharge; see section 2.2.4 and Figure 2.17).

The overall objectives of the project were to:

1. model, map and quantify diffuse P losses from soil P sources from surface runoff pathways at high resolution, accounting for spatiotemporal soil moisture deficits (SMDs), weather variations and accumulation of flow; this must be done using the latest Land Parcel Identification System (LPIS) data, hybrid soil moisture deficit (HSMD) model, soil drainage class map and a national 5-m digital elevation model (DEM). Modelling details are described in a separate Technical Modelling Report, available from the EPA;
2. identify cost-effective locations along surface runoff pathways, which could be used to target mitigation measures designed to reduce pollutant losses;
3. model sediment loss risk nationally to identify areas at highest risk;
4. analyse the latest international pesticide modelling literature and Irish monitoring data to inform modelling of pesticide loss. Details of this analysis are in a separate Technical Pesticides Report, available from the EPA.

2 Modelling Diffuse Source Phosphorus Pollution

2.1 Introduction

A range of water quality models exist internationally to model fate-and-transport of agricultural P, including the SWAT, HSPF and SHETRAN, which have been applied and tested in Irish catchments (Nasr *et al*., 2007). In addition, a suite of Irish DSTs have been developed specifically for modelling diffuse P losses in Ireland, ranging in scale from subfield to catchment (Drohan *et al*., 2019). Also a new catchment model (SMART) was developed to better model flow pathways (Mockler *et al*., 2016a). The DSTs are summarised in the next three sections.

2.1.1 EPA Catchment Characterisation Tool

In Ireland, the national DST for P is the EPA's Catchment Characterisation Tool (Mockler *et al*., 2017; Packham *et al*., 2020), which is a geospatial annual average export coefficient model for each P loss pathway. The model quantifies P loads ($\mathrm{kg\,ha^{-1}\,y^{-1}}$) and concentrations ($\mathrm{mg\,l^{-1}}$) from each pathway using national LPIS, soil and hydrogeological maps. Outputs include the EPA PIPv2 map, which shows categorical risk of P loss nationally at the subcatchment scale, with areas at the highest risk ranking being CSAs, shown in darker blue (Figure 2.1). The map is currently being used by LAWPRO, researchers and other stakeholders to characterise and screen catchments, and to identify PAAs in WFD RBMPs for monitoring, mitigation and on-the-ground assessments. This project contributes to updating this tool.

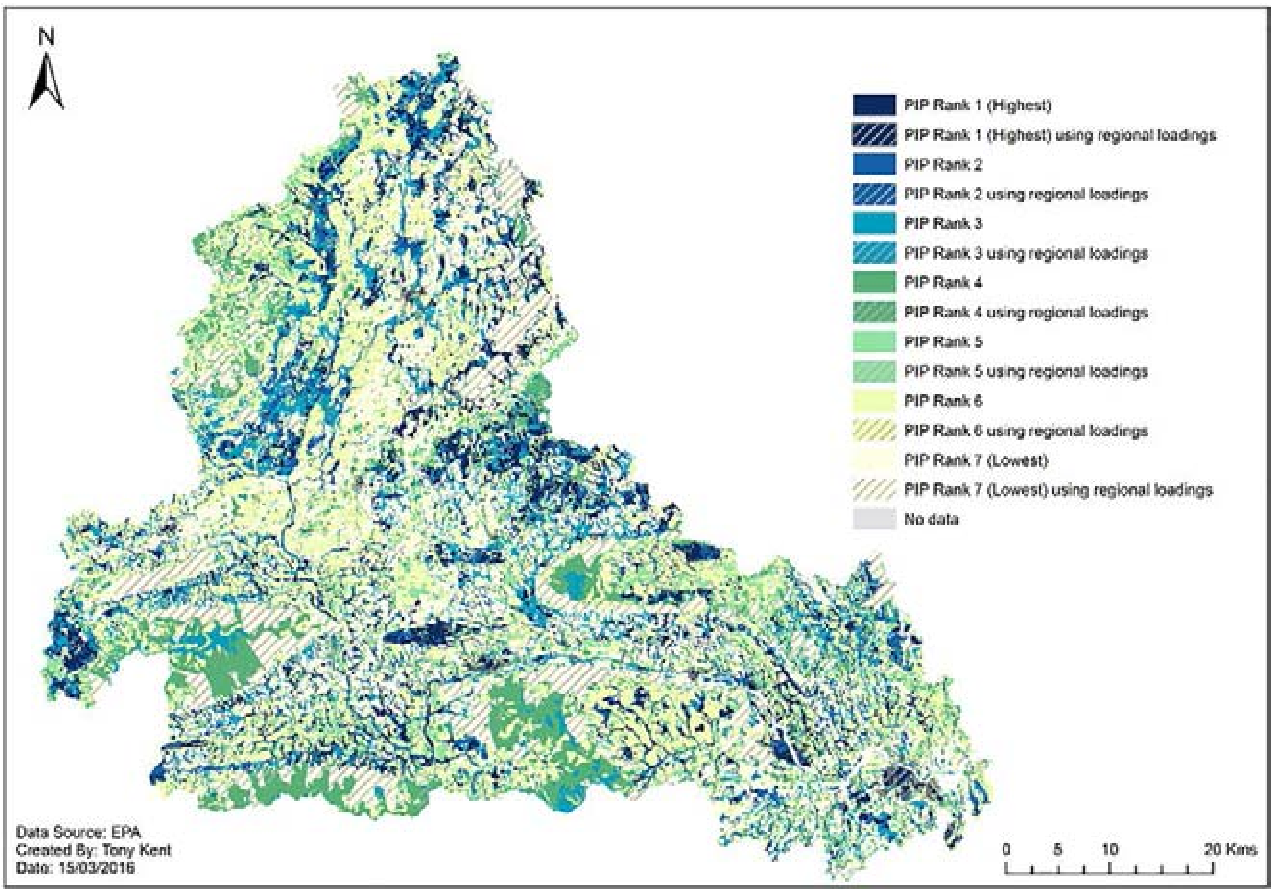

Figure 2.1. PIPv2 map for phosphate to surface water arising from diffuse agricultural sources for the Suir catchment. Areas at highest risk, CSAs, are in darker blue. Source: EPA.

2.1.2 Modified phosphorus ranking scheme

The Modified P ranking scheme (Magette *et al*., 2007) is a spreadsheet-based, field-scale tool inspired by the P index adopted in most US states (Sharpley *et al*., 2003), which ranks the risk of P loss from pasture, giving a risk score for each field. It integrates site-specific source and transport factors controlling P losses, with each factor assigned a weighting reflecting its relative importance. Applied at field and subcatchment scales, testing gave positive results. However, the application, development, testing and validation of these phosphorus ranking scheme tools were severely limited by a lack of good-quality data. The factor weightings are based on professional judgement rather than calibration, and risk from different pathways and sources, including fields and infrastructure (e.g. dirty water from farmyards), are lumped, which masks component risk variability. Furthermore, they do not model topographic influences of surface runoff pathways.

2.1.3 Soil topographic indices and critical source area (P) indices

The CSA index, which focuses on high-resolution (2 m) modelling of CSAs of diffuse soil P losses within a GIS, was developed and applied in four intensely monitored Teagasc Agricultural Catchments Programme (ACP) catchments using high-resolution datasets, including microtopography from light detection and ranging (LiDAR) DEMs, field-scale soil P concentrations, soil chemistry data (to derive water extractable P) and field-surveyed soil maps (Thomas *et al*., 2016b). Hydrologically sensitive areas (HSAs) are characterised using a soil topographic index (STI) (Walter *et al*., 2002), which uses the topographic wetness index (TWI) (Beven and Kirkby, 1979) derived from slope and flow accumulation, and maps of saturated hydraulic conductivity (K_{sat}) and soil depth. A modification to the STI, called the HSA index, reduced estimates of transport risk in areas upslope of flow sinks (pits and depressions, such as behind hedgerows) that were big enough to impede and hydrologically disconnect flow pathways (Thomas *et al*., 2016a). This allowed the identification of breakthrough and delivery points where P loss pathways crossed field boundaries (e.g. at gateways) and were delivered to waterbodies, respectively, where mitigation measures could be more cost-effectively targeted.

This approach was also applied to the Upper Bann catchment in Northern Ireland by Cassidy *et al*. (2019) to develop a carrying capacity framework for soil P and hydrological sensitivity and identify non-HSAs with low soil P for redistributing risk. Farm-specific P runoff risk maps were also sent to over 500 farms with the aim of changing farming practices and targeting measures at CSAs to improve water quality.

2.1.4 INCA-P and SimplyP

There are five widely used modern catchment-scale water–nutrient models: SWAT, INCA, AnnAGNPS, HSPF (as used in BASINS) and HYPE (Wellen *et al*., 2015). All of these simulate the export of P, and other nutrients, to rivers. The INCA model was developed in the UK, originally to simulate the export of nitrogen, but was then expanded for other areas and nutrients, including INCA-P for simulating P export (Jackson-Blake *et al*., 2016) and INCA-Sed for fine sediment modelling (Jarritt and Lawrence, 2007). INCA-P requires a very large number of input parameter values (148, of which 45 need adjustment or calibration from measurable characteristics, although this depends on the number of reaches and subcatchments used) and it requires input of hydrological fluxes generated by an external hydrological model, so considerable effort is required to run it. This motivated the development of the SimplyP model, which incorporates its own simple integrated hydrological model and requires minimal calibration and thus can be implemented in ungauged catchments. Its sediment component has performed satisfactorily in small catchments (Rankinen *et al*., 2010).

2.1.5 SWAT, HSPF and SHETRAN

Other conceptual models have been used to model sediment and P concentrations in rivers. The SWAT model is the most widely used (Gassman *et al*., 2014). HSPF is the hydrological catchment simulation component of the US EPA BASINS package (Duda *et al*., 2012). SHETRAN is a physically based, fully distributed, hydrological model based on a distributed grid structure (Ewan *et al*., 2000). In a previous EPA-funded project (LS2.2) HSPF produced the best simulation of discharge, but SWAT produced the best simulation of total P export (Nasr *et al*., 2007). A hybrid model, consisting of the hydrological components of HSPF combined with the P components of SWAT, gave improved simulations (Igbal and Bruen, 2014). However, all these models have issues with estimating peak concentrations, and calibration requires considerable amounts of data, so their implementation at a national scale is challenging. Furthermore, these models do not provide high-resolution spatially distributed predictions of P loss risk, and hence are unsuitable for the precise targeting of diffuse pollution measures at the subfield scale along runoff (transport) pathways.

2.1.6 National gridded hydroclimatic indices

There is a constant demand for high-quality, long-term gridded datasets of hydroclimatic variables in Ireland with high spatial and temporal resolution (Nolan and Flanagan, 2020). Such datasets are realistically possible only through the application of numerical weather prediction (NWP) simulations, and can be used in fields such as agriculture, water resource estimation and management, hydrology and hydrogeology, public health, energy and planning, and studies on observed climate change trends and vulnerability. In 2017, Met Éireann completed a 36-year reanalysis of the Irish climate using the HARMONIE model and the ALADIN-HIRLAM NWP system, known as MÉRA (Whelan *et al*., 2016; Gleeson *et al*., 2017), which has a resolution of 2.5 km^2. A definitive comparison between the MÉRA dataset and two other high-resolution historical weather simulations from COSMO-CLM5 (Rockel *et al*., 2008) and WRF v3.7.1 (Skamarock *et al*., 2008) NWP models was undertaken by Irish Centre for High-End Computing (ICHEC) researchers (Nolan and Flanagan, 2020). They showed that the MÉRA data had the fewest errors for most climate variables, and had the advantage of data assimilation, and, therefore, should be considered the primary source and utilised as the first national hydroclimate dataset for Ireland.

Furthermore, using MÉRA, COSMO-CLM5 and WRF NWP, Werner *et al*. (2019) developed high-resolution, gridded and multi-decadal (1981–2016) reference and actual evapotranspiration and SMD datasets for Ireland nationally, at daily, monthly, annual and 30-year time steps. SMDs were calculated using the HSMDv2 model (Schulte *et al*., 2005, 2015; Hallett *et al*., 2014). The whole country was assumed to be in one of five soil drainage classes (excessively, well, moderately, imperfectly and poorly drained) and the results were spatially joined to the Irish Soil Information System indicative soil drainage class map (Werner *et al*., 2019).

2.1.7 Model, knowledge and data gaps in Ireland

All of the tools described previously are applied at catchment or farm scale, and none makes use of the high-resolution datasets now available to identify or rank, at national scale, points at which diffuse-source pollution is likely to be delivered to rivers. Although farmers have farm-specific knowledge of their land that models cannot replicate without vast data resources, models do have the ability to model large areas at sufficiently high resolution to inform on-the-ground targeting of measures and "where to look" for issues (Djodjic *et al*., 2018). This has been successfully implemented by Ireland's national DST for P: the EPA's Catchment Characterisation Tool and PIPv2 maps (Packham *et al*., 2020). However, these are risk-based maps, which can be used at a maximum scale of 1:25,000 to target local catchment assessments, and are not designed, or suitable, to be used on their own as a basis for decisions at local or field scale. Recent advances in big data, such as high-resolution LiDAR DEMs and maps of soil type and soil P concentrations, in combination with GIS and STIs, have rapidly improved modelling and mapping of HSAs, CSAs and surface runoff pathways in agricultural landscapes (Thomas *et al*., 2016a,b; Drohan *et al*., 2019), but these approaches have been undertaken in only a few specific local catchments and datasets are typically not available nationwide. Furthermore, neither approach quantifies pollutant losses, which is necessary to determine the amount of mitigation required to achieve specified water quality goals. There is, therefore, a need to develop a diffuse P loss model that integrates the best components from each approach (Doody *et al*., 2016), utilising and developing nationally available datasets where possible, such as a national 5-m NEXTMap DEM now available to model flow and pollutant accumulation based on topography.

2.2 Methodology

2.2.1 Overview

The DiffuseTools' Technical Modelling Report, available from the EPA, describes in detail the methodology of the diffuse P loss modelling component of the DiffuseTools project. However, a brief overview is given here. Surface runoff and P exports were estimated for 5-m grid cells from estimates of surface runoff P concentrations and surface runoff volumes. Surface runoff P concentrations (mg l^{-1}) were predicted using a relationship with soil Morgan P concentrations (mg l^{-1}) (Jordan *et al.*, 2019). An important point is that all fields were assumed to be midpoint P index 1–4 soils using a scenario analysis and LPIS 2018 land use data (e.g. 6.5 mg l^{-1} for grassland and 8 mg l^{-1} for arable in the P index 3 scenario). This was necessary because a national high-resolution map of soil P concentrations is not currently available. When such a map becomes available, then its soil P information can be integrated into the model. Surface runoff volumes were a 30-year annual average (1981–2010) predicted using a specially calibrated version of the HSMD model (Schulte *et al.*, 2015), together with MÉRA gridded weather data (precipitation and reference evapotranspiration) and a new Soils Hydrology Ireland soil drainage class map. Calibration was undertaken using Teagasc ACP quickflow estimates. Accumulation of surface runoff volumes and P loads downslope based on topography were calculated using a hydrologically corrected 5-m NEXTMap DEM, which had waterbodies burned into it and pits and depressions filled. A national waterbodies map was used for the hydrological correction and to define the delivery points of diffuse-source P pollution to surface waterbodies, and was generated by merging and rasterising the Prime 2 datasets of streams, ditches, rivers, lakes, coastlines and estuaries generated by the EPA and Ordnance Survey Ireland (OSI). Breakthrough points were then identified at LPIS 2018 field boundaries.

The modelling was based predominantly in an ArcGIS (v10.7) ModelBuilder framework, to semi-automate workflows and allow model re-runs with new input datasets as they become available, although at some stages data processing was undertaken outside ArcGIS, in SAGA GIS and QGIS GRASS, because of their specialist algorithms.

2.2.2 *Mapping receptors*

For receptor waterbodies, a new National Waterbodies map (shapefile and raster) was created, which merged EPA WFD and OSI Prime 2 datasets of streams, ditches, rivers, lakes, coastlines and estuaries. This was used in hydrologically correcting a 5-m NEXTMap DEM to define transport pathways (see section 2.2.5) and delivery points (see section 2.2.6). Note that all OSI references are related to © Ordnance Survey Ireland, licence 2019/OSi_NMA_074.

2.2.3 *Nutrient sources and mobilisation risk*

This project focused on soil P sources of diffuse pollution, as the movement of legacy soil P hinders mitigation and it is the P source representing the main long-term threat to water quality (Jarvie *et al.*, 2013). By accurately modelling and mitigating soil P losses via surface runoff pathways, the other P sources can be effectively mitigated at the same time (Withers *et al.*, 2003), along with losses of numerous other diffuse agricultural pollutants, including nitrogen, sediments, pesticides, other chemicals and pathogens. It must be noted that pollutant swapping can occur at mitigation locations, and losses can occur through other pathways (e.g. subsurface in the case of nitrogen) (Stutter *et al.*, 2019; Cole *et al.*, 2020).

To model soil P sources, a scenario analysis was used in which all agricultural soils were assumed to have soil P concentrations (mg l^{-1}) at the midpoint of each of the four soil Morgan P Indices (1–4), which depend on land use (Table 2.1). The model was run separately for each of the four P indices and ranges in surface runoff P loads from each were estimated. This scenario approach bypasses the need for national high-resolution soil P maps, as these are not yet available. The LPIS data for 2018 were used to spatially map agricultural field parcels and assign land use. LPIS data were pre-processed, including dealing with commonage issues, removing duplicates, fixing topology errors, and spatially erasing exclusion areas (non-agricultural field areas such as roads, tracks, farmyards, buildings, etc., the first two of which may

Table 2.1. Soil P concentration (mg l^{-1}) for land use and P index scenarios

Soil P index scenario	Grassland		Other crops	
	Range	Midpoint	Range	Midpoint
1	0.0–3.0	1.5	0.0–3.0	1.5
2	3.1–5.0	4.0	3.1–6.0	4.5
3	5.1–8.0	6.5	6.1–10.0	8.0
4	> 8.0	9.0	> 10.0	12.0

not be resolved in the DEM). It should be noted that surface runoff pathway/volume modelling included all areas nationally (see section 2.2.5).

Surface runoff P concentrations (mg l^{-1}) for grassland and arable soils were then estimated for each 5-m grid cell nationally for the midpoint concentration of each P index scenario using the relationship $y = 0.0049x^{1.657}$ derived from Irish literature review data by Jordan *et al*. (2019).

Unlike mineral soils, which slowly desorb P from the soil P storage pool, peats and soils with high organic matter have no or little P sorption or storage capacities, and hence applied fertiliser P is highly mobile and at high risk of incidental losses (González Jiménez *et al*., 2018). To map this, the LPIS 2018 data were intersected with peats in the new Soils Hydrology Ireland map, and a "high P mobility" attribute was created, flagging fields with high P mobility risk due to underlying peats.

2.2.4 Mapping soil moisture deficits and surface runoff for each soil drainage class

Surface runoff volumes were estimated nationally to help estimate surface runoff P loads. To account for antecedent soil moisture conditions, soil drainage, spatiotemporal weather variations and climatic averages, the use of Ireland's national HSMDv2 model (Schulte *et al*., 2015), in combination with MÉRA data, was investigated. For example, national hydroclimatic indices for Ireland were developed by Werner *et al*. (2019) using this model in combination with observed precipitation and calculated reference evapotranspiration MÉRA data as inputs, to estimate SMDs for five soil drainage classes (excessively, well, moderately, imperfectly, poorly) on a daily time step from 1981 to 2016.

To model daily surface runoff, the DiffuseTools project updated the existing HSMDv2 model. To do this, HSMDv3 was developed by this project using precipitation, reference evapotranspiration and soil drainage class maps from three intensively monitored Teagasc ACP catchments, which are representative of Irish agri-environmental conditions (Fealy *et al*., 2010). These include a grassland catchment with predominantly poorly drained soils (Ballycanew), a grassland catchment with predominantly well-drained soils (Timoleague) and an arable catchment with mixed soil drainage classes (Dunleer). In the three ACP catchments, precipitation in excess of field capacity [i.e. negative or minus SMD (mSMD) in mm day^{-1}] was related to daily independently estimated quickflow depths (mm day^{-1}) from October 2009 to September 2014. Next, HSMDv3 parameter values were calibrated to characterise the relationship between mSMD and quickflow, which was then used to predict daily surface runoff depths (mm day^{-1}) for each soil drainage class. Percolation depths were then calculated as the difference between effective rainfall and surface runoff (see the DiffuseTools' Technical Modelling Report, available from the EPA, for the equation details).

The HSMDv3 model was then run for the whole of Ireland using the ICHEC supercomputer and daily 1-km^2 Met Éireann observed precipitation data (Walsh, 2012) and 2.5-km^2 reference evapotranspiration data derived from the MÉRA dataset (Werner *et al*., 2019). The model was run four times, each time assuming that the whole country was in one of four soil drainage classes (well, moderately, imperfectly or poorly drained). From daily results for each run, monthly and annual sums (for actual evapotranspiration, effective rainfall, mSMD, surface runoff and percolation) or means (for SMD and mSMD) were calculated, together with 30-year (1981–2010) annual averages (climate averages). These annual average results for each hydrological variable were then spatially joined to the new Soils Hydrology Ireland map of soil drainage classes (well, imperfectly, poorly and very poorly drained classes, plus alluvium, peat, made and water). Peats and alluvium were assumed to behave as poorly drained soils for the purposes of national coverage of surface runoff predictions. Mapped surface runoff depths were then rasterised (5 m grid resolution) and converted to surface runoff volumes.

2.2.5 Surface runoff pathways, flow accumulation, upslope drainage areas and hydrological connectivity

Surface runoff pathways and the accumulation of the predicted flow volumes downslope to waterbodies along these pathways were then modelled using the 5-m bare earth NEXTMap DEM. A small fraction (set here at 5%) of the additional accumulated surface runoff from upslope that traverses a cell is assumed to interact with the soil P of that cell, and the accumulated surface runoff volume interacting with the soil ($l\,y^{-1}$) is calculated as the amount of overland flow that is generated within the cell, plus 5% of the accumulated flow from upslope that traverses the cell.

Upslope drainage area (UDA) boundaries were delineated directly for each watercourse perimeter segment in the r.watershed tool in QGIS GRASS, to produce a national UDA map that divided the country into hydrologically discrete areas of accumulation of flow (and thus diffuse pollution pathways) to waterbody segments. This was intersected with the areas generating surface runoff to create a HSA map that calculated the total surface runoff volume for each UDA. Flow sinks were identified using the methodology developed by Thomas *et al.* (2016b), by subtracting the hydrologically corrected DEM with flow sinks filled from the DEM prior to filling flow sinks. This gives a flow sink depth map for each 5-m grid cell.

2.2.6 Targeting mitigation locations

For each P index scenario, surface runoff P loads were calculated for each 5-m grid cell nationally, by multiplying the predicted surface runoff P concentration ($mg\,l^{-1}$) by the accumulated surface runoff volume interacting with the soil ($l\,y^{-1}$), and converting $mg\,y^{-1}$ values to $kg\,y^{-1}$ and $kg\,ha^{-1}\,y^{-1}$. As different users and stakeholders work at different scales of land/catchment management, surface runoff P loads, calculated at the 5-m scale, were then also aggregated (summed) at field scale corresponding to LPIS 2018 data, and at UDA and WFD river sub-basin scales. Accumulation of surface runoff P loads ($kg\,y^{-1}$) along multiple flow direction pathways was then predicted using the same approach as stated in section 2.2.5 in the r.watershed tool in QGIS GRASS, but using the surface runoff P loads (at 5-m scale) as the amount of overland flow per cell input. The output accumulation map was, therefore, the surface runoff P load that traverses each 5-m cell, including from UDAs. Thus, surface runoff P loads accumulate downslope along surface runoff pathways to waterbodies and along waterbodies to the WFD catchment outlet.

Breakthrough and delivery points, that is the locations where surface runoff pathways cross field boundaries and are delivered to waterbodies, respectively, were then identified using ArcGIS ModelBuilder workflows that pinpointed where flow pathways crossed LPIS 2018 parcel boundaries, and where they entered the waterbodies (see DiffuseTools' Technical Modelling Report, available from the EPA, for the complete methodology). The total and maximum accumulated surface runoff P loads ($kg\,y^{-1}$) at delivery points and breakthrough points, respectively, were then calculated, to further prioritise targeting of measures. The maximum rather than the total load was calculated at breakthrough points, because flow pathways often ran parallel to field boundaries, leading to the risk of double-counting. This was not as much of an issue at delivery points where adjacent waterbodies (burned into the DEM) were the steepest flow direction.

2.3 Results and Outputs

2.3.1 New 5-m national waterbodies dataset with gaps filled

The project produced a national waterbodies feature class, which includes rivers, streams, lakes, estuaries, coastal boundaries and some open drainage ditches. This is also available as a 5-m raster. The total rasterised waterbody perimeter was 494,327 km.

2.3.2 Maps of soil P sources and mobilisation risk

Soil P concentrations and surface runoff P concentrations were estimated for each P index scenario (Figure 2.2). Arable land use occurs predominantly in the south and east of the country, and the Morgan P index system has higher midpoint soil P concentrations for arable than grassland soils; this was reflected in the national maps, with soil P and surface runoff P concentrations typically higher in these areas under the P index 1–4 scenarios. It is important to note that these are ranges and do not reflect actual concentrations and distributions at any specific point, but are rather part of a scenario analysis to constrain expected ranges in P losses. At any particular point, these ranges are also influenced by stocking density, nutrient/fertiliser/feed management, grazing practices, soil management, soil properties and other aspects of plant and animal husbandry (Murphy *et al.*, 2019). Agricultural fields from LPIS 2018 with high P mobility risk due to underlying peats (mapped from the Soils Hydrology Ireland map) were also mapped; these fields have an inherently higher P mobilisation risk owing to their high soil organic

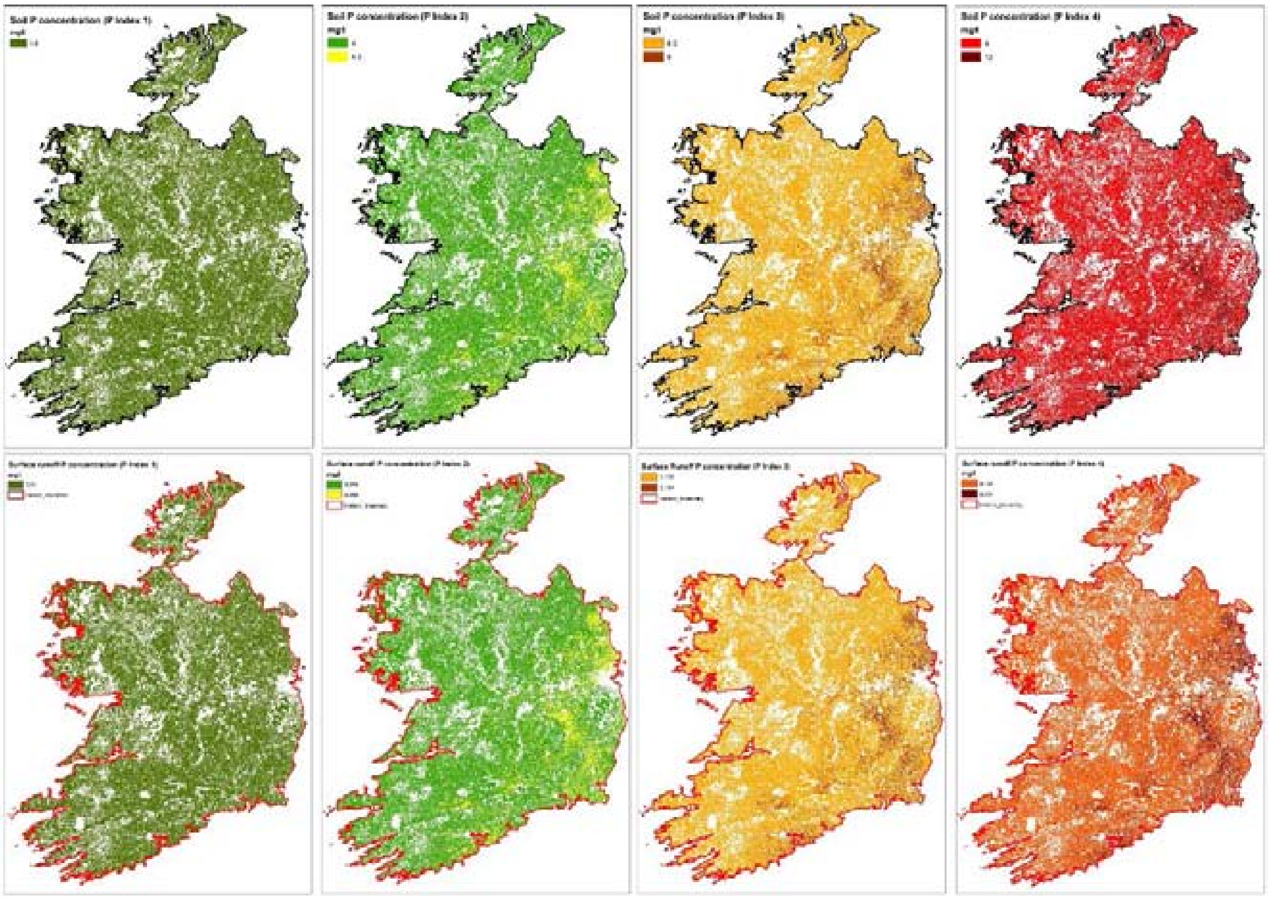

Figure 2.2. Morgan P concentrations (mg l^{-1}) in (top row) soil and (bottom row) surface runoff for each of the four soil Morgan P index scenarios.

matter content, which provides weak P binding sites and high P release.

2.3.3 Maps of soil moisture deficits and flow volumes

A relationship was derived between daily mSMD (mm day^{-1}; from HSMDv3) and estimated quickflow (mm day^{-1}) in the three Teagasc ACP catchments from October 2009 to September 2014 (Figure 2.3). Daily mSMD values are the catchment average, and only days with quickflow > 0 were included (a total of 1636 daily data points). The time series of rainfall, reference evapotranspiration (ET_0), actual evapotranspiration (ET_a), effective rainfall, SMD, mSMD and quickflow (all mm day^{-1}) for the three Teagasc ACP catchments are shown in Figure 2.4. The temporal dynamics of mSMD are closely matched with estimated quickflow for each catchment, although mSMD values are consistently much higher, as shown also in Figure 2.3. Furthermore, the differences between catchments are pronounced, with Ballycanew (predominantly poorly drained) having much larger mSMD and quickflow than the well-drained Timoleague. The accuracy of the soil drainage class map being used to spatially join/map HSMDv3 results is important and should ideally be field verified or refined using soil samples (as was done in the Teagasc ACP catchments).

Figure 2.5 shows the comparison between daily surface runoff depth (predicted in HSMDv3 using Figure 2.3) and quickflow in the three catchments over the same period. Surface runoff is predicted during the vast majority of quickflow events, and typically in similar magnitudes, but peaks tend to be underestimated during extreme storm events. Overall, predicted annual surface runoff and annual quickflow amounts were very similar for the three catchments, with a maximum absolute difference of 11.2 mm y^{-1} (Table 2.2).

The national 30-year annual mean HSMDv3 results (1981–2010) are shown in Figure 2.6 as 5-m rasters for actual evapotranspiration, effective rainfall, SMD,

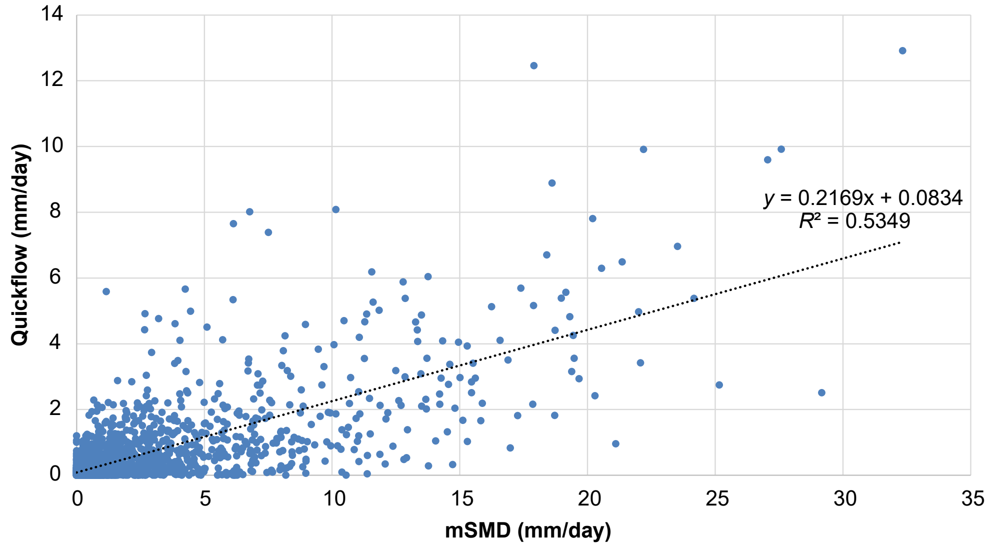

Figure 2.3. Calibrated relationship between daily mSMD and estimated quickflow for the three Teagasc ACP catchments from October 2009 to September 2014.

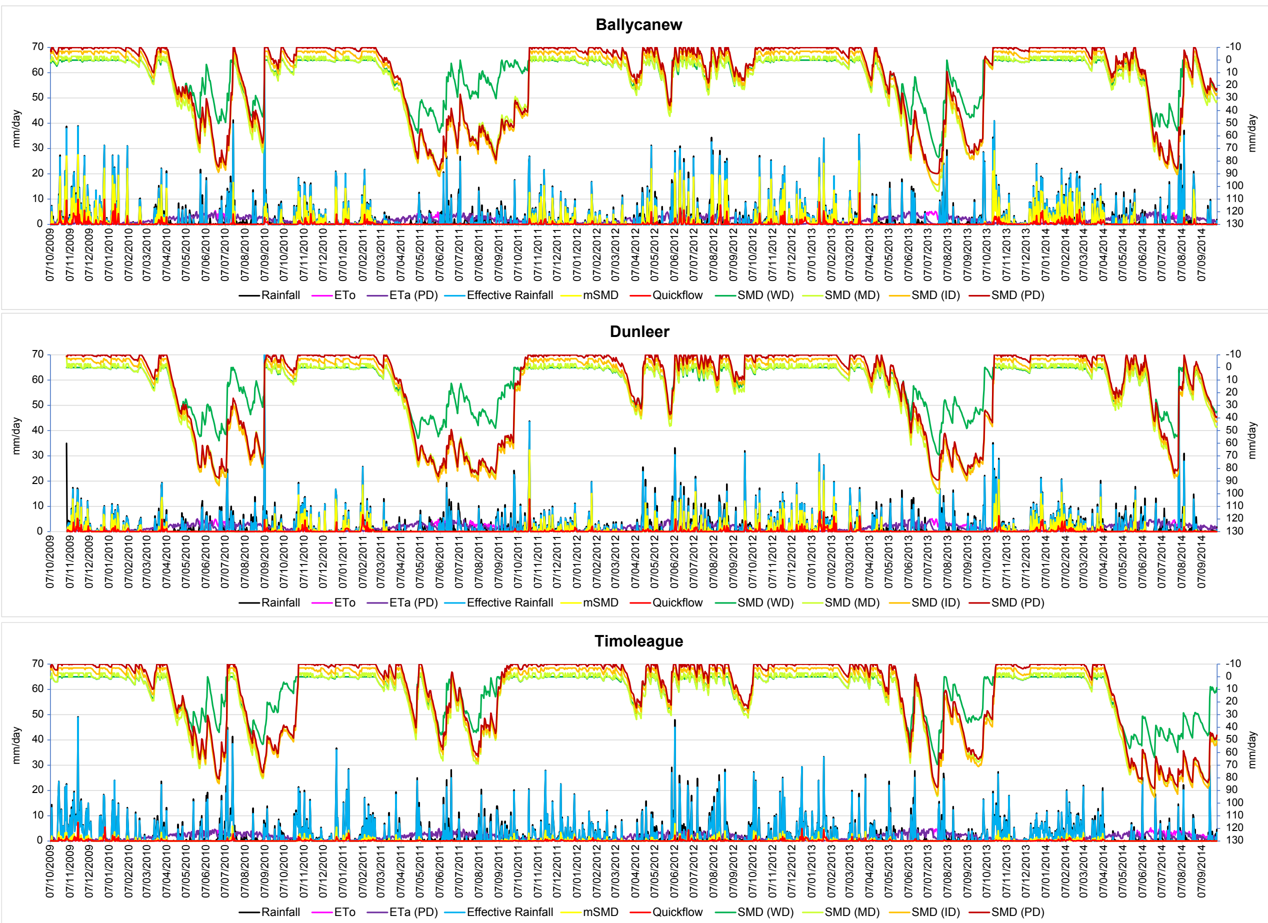

Figure 2.4. Daily hydrological time series (mm day^{-1}) for the three Teagasc ACP catchments from October 2009 to September 2014. The secondary axis shows SMD (mm day^{-1}). ID, imperfectly drained; MD, moderately drained; PD, poorly drained; WD, well drained.

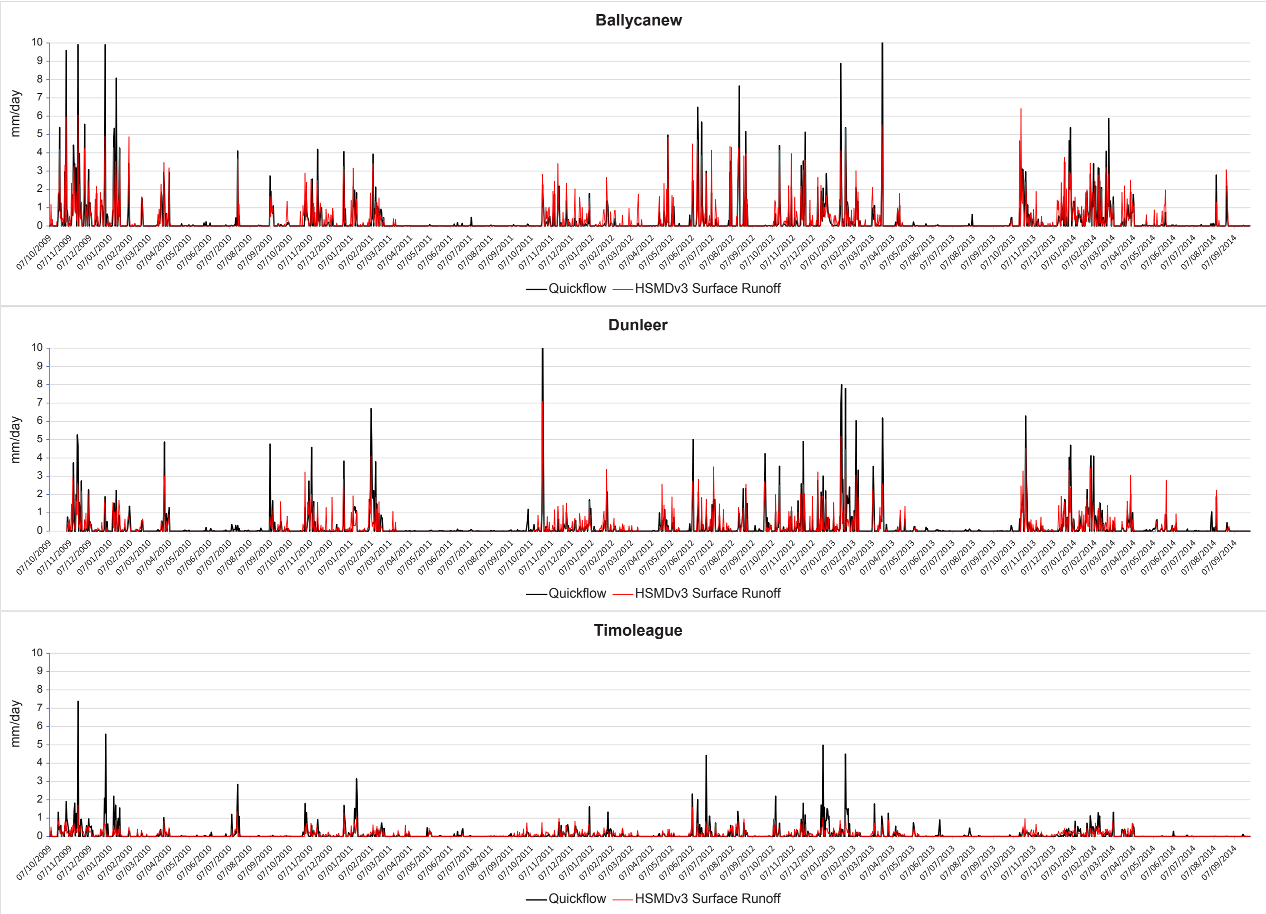

Figure 2.5. Comparisons between estimated quickflow (from hydrograph separations at Teagasc ACP catchment outlets) and predicted surface runoff (from HSMDv3), from October 2009 to September 2014 (all in mm day^{-1}).

Table 2.2. Comparison of annual water balance predictions between HSMDv3 and measurements from ACP catchments (mm y^{-1})

Quantity	Ballycanew	Dunleer	Timoleague
Total rainfall	1144.7	910.1	1113.6
Total effective rainfall	637.6	420.5	669.0
Total estimated quickflow	105.7	85.5	42.5
Total mSMD	498.5	308.9	101.6
Total predicted surface runoff	116.9	75.8	32.0
Missing surface runoff	–11.2	9.7	10.5
Total predicted percolation	520.7	344.7	637.0

mSMD and surface runoff depth. The results were spatially joined to the Soils Hydrology Ireland map, as HSMDv3 provides results for each soil drainage class at any location nationally. As expected, the proximity to the North Atlantic Current and topographic position had a significant influence on weather patterns, with the western coastal and mountainous areas having higher effective rainfall, lower SMDs, higher mSMD and hence higher surface runoff depths (mm y^{-1}) than lowland areas.

2.3.4 *Maps of flow pathways, hydrological connectivity and DEM products*

To model surface runoff flow pathways, the national 5-m NEXTMap DEM was hydrologically corrected by burning in the National Waterbodies map and filling flow sinks. A national slope map was then generated, which was used to define flow directions and flow accumulation to downslope waterbodies. The hydrological correction process was then reversed to extract flow sinks nationally, as shown in Figure 2.7. These are features such as hedgerows or pits that can topographically capture surface runoff and hydrologically disconnect pathways.

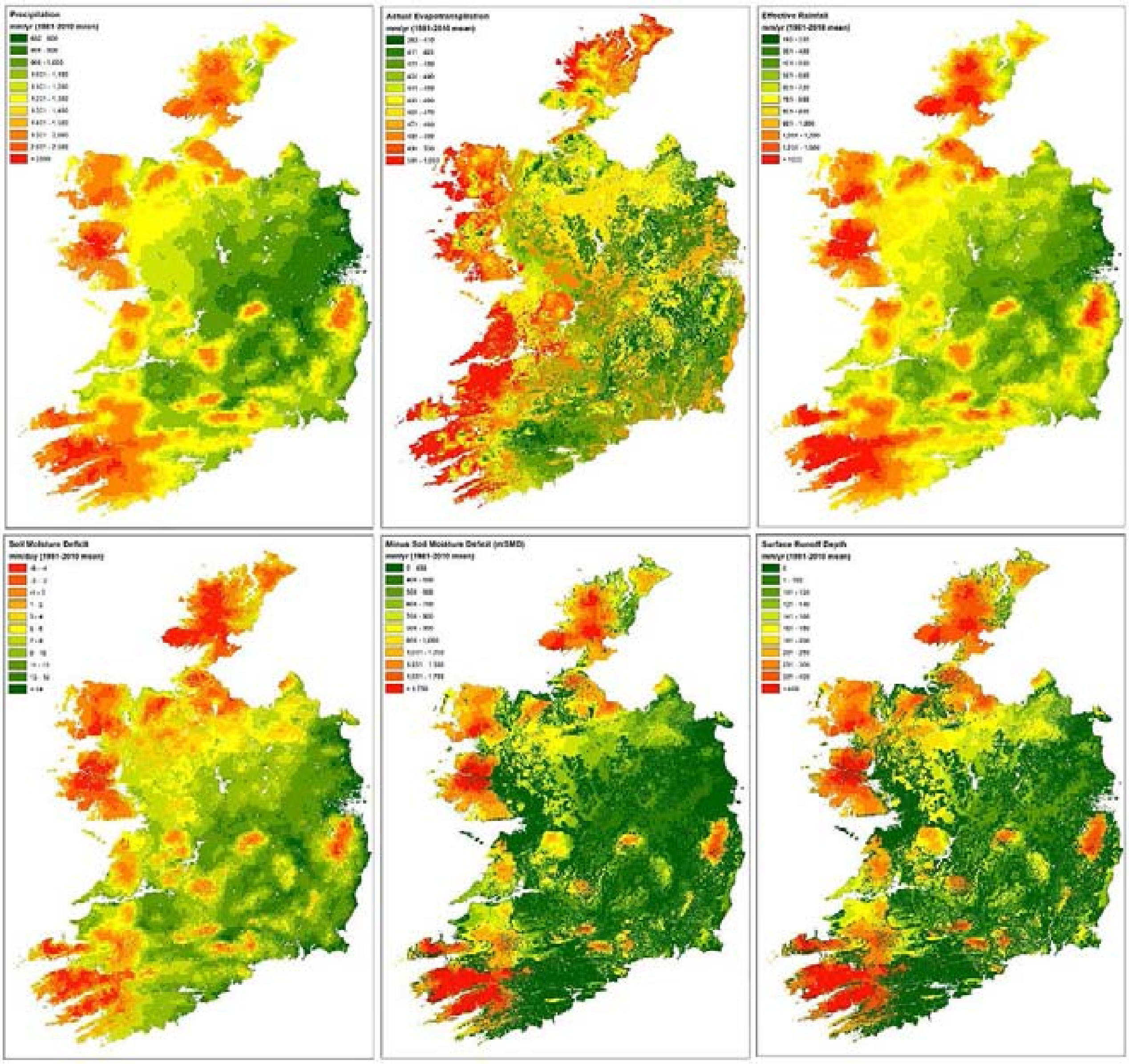

Figure 2.6. National HSMDv3 modelled 30-year annual means of hydrological variables. Clockwise from top left, the variables are precipitation, actual evapotranspiration, effective rainfall, SMD, mSMD and surface runoff.

Evapotranspiration and/or percolation of this trapped surface runoff results in the deposition of dissolved and entrained pollutants at the bottom of the flow sink, preventing them from reaching the waterbody via surface pathways. Flow sinks were prolific in the agricultural landscape, but locations were spatially variable nationally, with the majority being in lowland areas and valley bottoms, particularly in the Midlands and karst areas. Because the DEM resolution does not capture some microtopography, such as hedgerows, it misses some flow sinks that would be picked up if a higher-resolution LiDAR DEM was used, particularly behind hedgerows (see Thomas *et al*., 2016a). The national TWI and STI maps were used as additional information, particularly when topographically driven surface runoff is occurring (see section 2.3.7). The STI was initially used to predict surface runoff risk but, following comparisons with CSAs in aerial imagery (shown as gleyed soils adjacent to waterbodies), it was clear that, although showing the pathways correctly, it tends to overpredict surface runoff risk in well-drained soils with high flow accumulation, and to underpredict risk in poorly drained soils with low flow accumulation.

The national surface runoff volume map, derived from the depth map for each 5-m grid cell, is shown

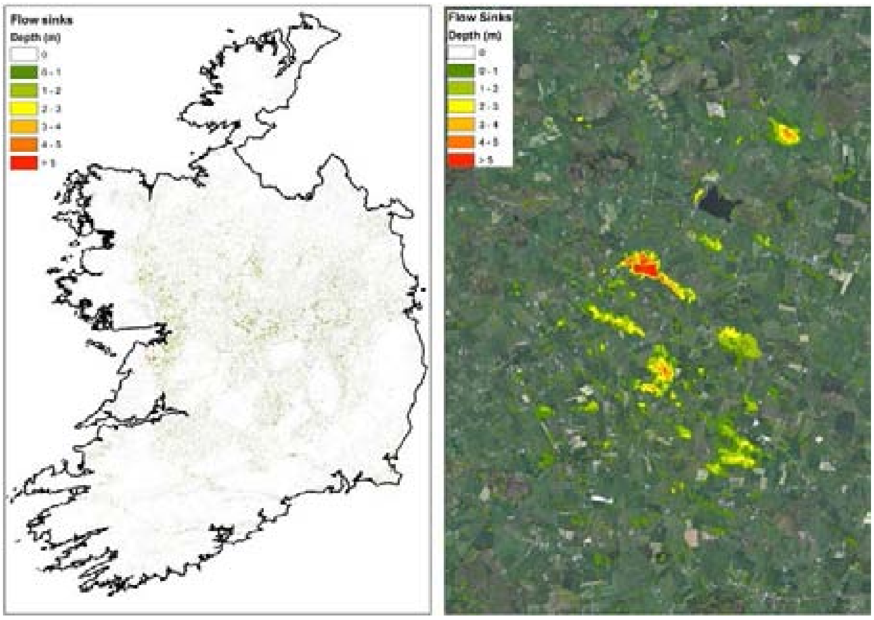

Figure 2.7. Flow sinks (pits and depressions) nationally (left) and a close-up example (right). © Ordnance Survey Ireland, license 2019/OSi_NMA_074.

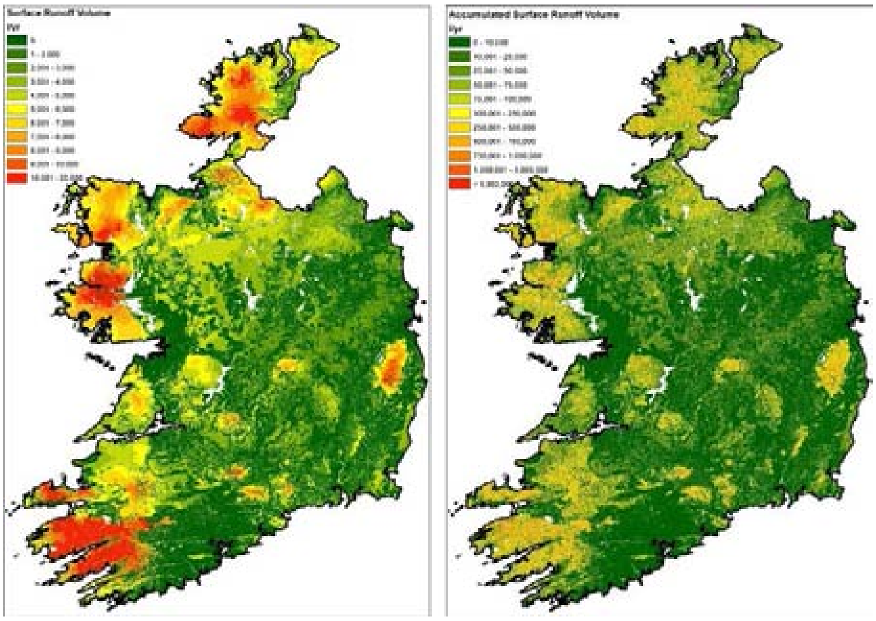

Figure 2.8. Surface runoff volume (left) and accumulated surface runoff volume (right) along multiple flow directions, both derived from 30-year annual average surface runoff depth.

in Figure 2.8. Volumes range from 0 to 15,500 l y^{-1}. Accumulation of surface runoff volumes downslope along multiple flow directions based on the hydrologically corrected 5-m NEXTMap DEM is also shown in Figure 2.8. The amount of accumulation as water flows downslope is clearly shown, with values ranging from 0 to >5 million l y^{-1}. Individual surface runoff pathways are clearly identifiable from source (upslope) to receptor (downslope) (see close-up in Figure 2.9), which is crucial in identifying where diffuse agricultural pollutants enter waterbodies. Importantly, these are also pathways of multiple diffuse agricultural pollutant losses, including P, nitrogen, sediments, pesticides, other chemicals and pathogens. The national map of accumulated volume that is interacting with the soil (set as 5% of accumulated flow from upslope drainage areas plus the surface runoff generated at the grid cell) is also shown in Figure 2.9.

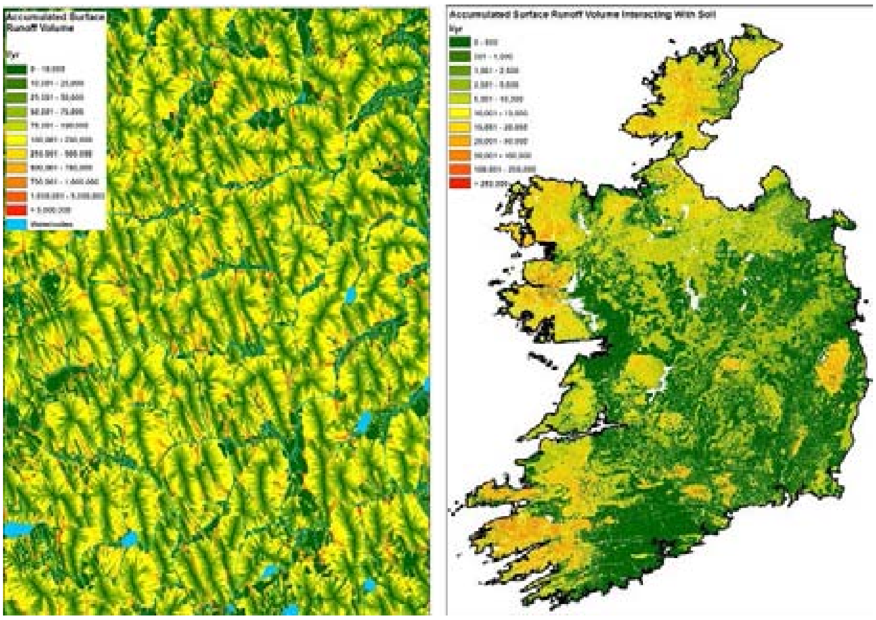

Figure 2.9. Close-up example of surface runoff accumulation showing individual pathways (left) and accumulated surface runoff volume interacting with soil (right).

Upslope drainage area boundaries are shown in Figure 2.10, based on discrete areas of flow accumulation. There were over 3 million UDA polygons, dividing the country into hydrologically discrete areas of accumulation of flow (and diffuse pollution pathways) to waterbody segments. These were used in the surface runoff P load (UDA scale) map. HSAs generating surface runoff (i.e. imperfectly and poorly drained soils, alluvium and peats), which act as surface pathways of diffuse pollution, are shown in Figure 2.11, with polygons intersected by UDAs.

2.3.5 Maps of mitigation locations

National maps of breakthrough points and delivery points for each P index scenario were produced and a sample close-up is shown in Figure 2.12. Breakthrough points, at field boundaries, are shown

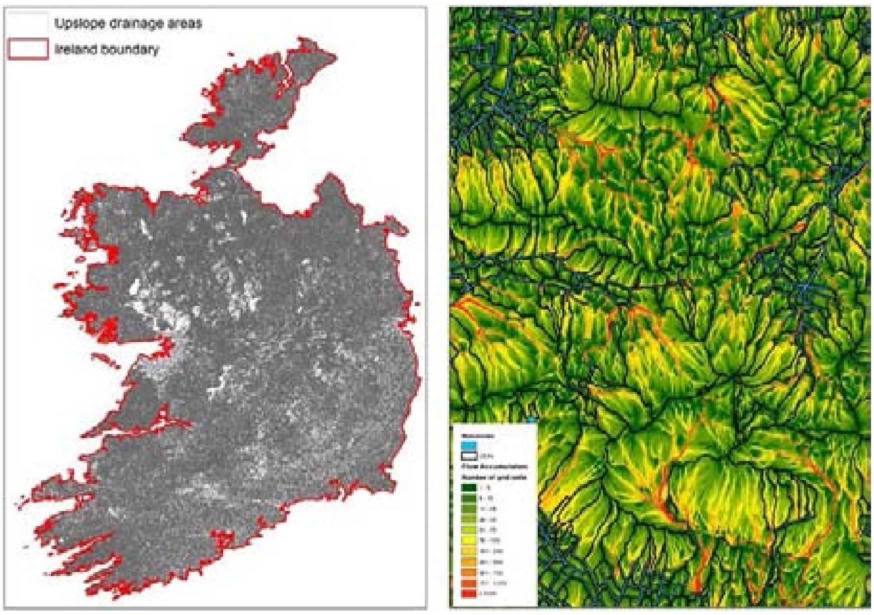

Figure 2.10. Upslope drainage areas nationally (left) and a close-up (right).

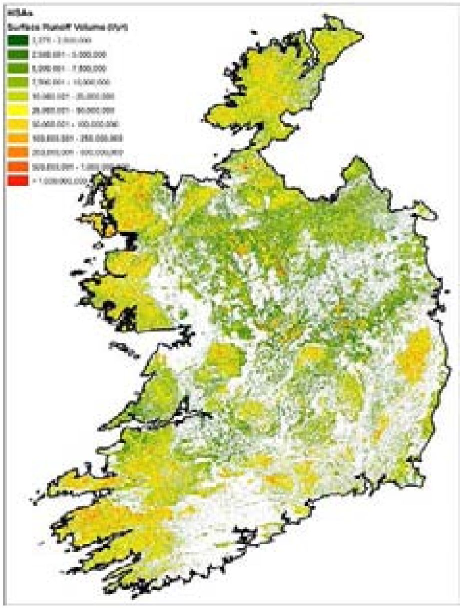

Figure 2.11. National HSAs map showing polygons of surface runoff-generating areas and volumes.

in pink and are nearer the sources of pollution than delivery points, shown in red, which are at lower hillslope positions, adjacent to watercourses, where mitigation measures (see NFGWS, 2020) could be more cost-effectively targeted. The example map highlights the highly diffuse nature of agricultural pollution, although this varies across the country and is controlled by topography and soil drainage.

2.3.6 National results

National summary statistics of predicted surface runoff amounts, breakthrough lines and delivery lines are shown in Tables 2.3 and 2.4. These indicate that only a relatively small fraction of land is generating a significant proportion of diffuse P pollution. This indicates that a relatively small fraction of national field boundaries or waterbody lengths need mitigating measures. To put this into context, Ireland's Office of Public Works (OPW) is responsible for about 11,500 km of river channels, including about 800 km of embankments, and local authorities are responsible for maintaining about 4600 km of drainage districts (OPW, 2020). Furthermore, using P index 3 as an example, these national results show that the breakthrough lines and delivery lines with the highest surface runoff P loads (e.g. > category 10) account for, respectively, approximately 34% and 57% of total loads, but only 4% and 11% of total length. This pattern is also found with other P index scenarios. These "low-hanging fruits" should be the focus of national mitigation policy, as the results also show that lower category breakthrough and delivery lines are significantly more numerous and longer and therefore much harder to mitigate. The results also show policymakers how aiming for different soil P indices affects pollutant loads nationally and the relative increase or decrease in mitigation measures required.

2.3.7 Model evaluation

Checking model with LAWPRO ground-truthing

Key to the success of water quality improvements needed in Ireland is LAWPRO, which carries out local on-the-ground catchment assessments and promotes the implementation of targeted mitigation measures to improve water quality at local levels. The DiffuseTools team and EPA Catchments Unit met with LAWPRO and decided that visual observations and photographic evidence would be collected during field and river walks, ideally during storm events when HSAs and surface runoff pathways would become active, and observations would be compared with model results.

As LAWPRO focuses assessments on PAAs, the map of accumulated surface runoff volume was clipped to 11 PAAs of its choice, which were familiar to LAWPRO from its fieldwork. The clipped maps were then overlaid onto PIPv3 maps in development by the EPA Catchments Unit, which account for stocking rates from LPIS 2018, and an interpretation guide was produced (E. Mockler, EPA, July 2020, personal communication; see section 2.4.4 and Figure 2.18). The combined map for each PAA was then provided to LAWPRO, which utilised the PIPv3 rankings to screen subcatchment areas to visit, and then used the overlaid DiffuseTools delivery paths and points to hone specific field and subfield locations to visit where pollution hotspots and delivery could be occurring. Initial feedback on comparison of field observations (e.g. from LAWPRO) with model predictions were positive and additional observations can be used to improve the model further.

Checking model with measured EPA water quality data

OSPAR. Model results were compared with EPA water quality data. First, results were compared with

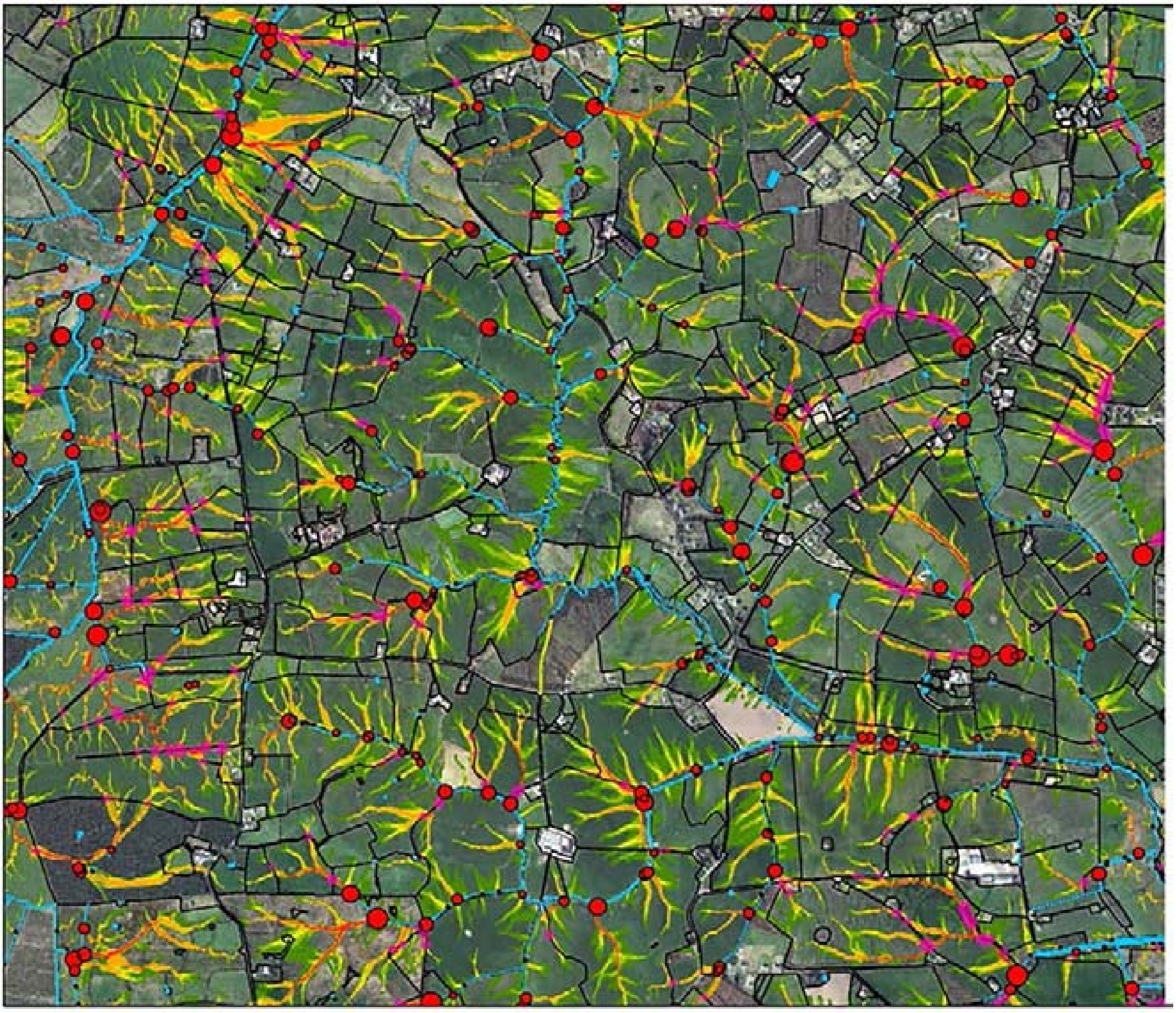

Figure 2.12. Close-up example of the surface runoff P delivery map showing locations of breakthrough points at field boundaries (pink circles) and delivery points to waterbodies (red circles). The size of the circles indicates magnitude of delivery, so measures could be further targeted at the largest points to achieve cost-effective reductions in P loss to waterbodies.

estimates of annual total P (TP) loads ($t\,y^{-1}$) from 19 major Irish rivers to estuarine and coastal waters, monitored between 1990 and 2018 as part of the Oslo Paris Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) (O'Boyle *et al*., 2019). The OSPAR calculations indicated that TP loads from major Irish rivers in the period 1990–2018 ranged from approximately 1000 to $4600\,t\,y^{-1}$, with a mean TP load of $2150\,t\,y^{-1}$. Extracting the DiffuseTools modelled TP loads from the major Irish rivers for the period 1990–2010 gave a very similar range of estimates of TP load, $1200–4600\,t\,y^{-1}$, with a mean of $2475\,t\,y^{-1}$. However, these TP loads differ from total reactive P (TRP) load, as TP includes particulate P and P from all sources, as well as agriculture. Furthermore, the sampling frequency of the measured data may not capture the high temporal variability during storm events, when a large proportion of P losses can occur.

SLAM. Mockler *et al*. (2017) applied the Source Load Apportionment Model (SLAM) (Mockler *et al*., 2016b) to 16 catchments dominated by diffuse nutrient sources that were part of OSPAR to disaggregate P sources. These monitoring stations in these catchments were largely located at tidal limits and generally upstream of large wastewater treatment plant discharges from coastal towns and cities.

Table 2.3. National summary statistics of surface runoff P losses and mitigation measure locations (breakthrough and delivery lines) for P index 3 scenario

Statistic description and unit	Value
Total Ireland area (km^2)	70,273
Total Ireland utilised agricultural area (UAA; km^2)	44,811
Total Ireland rasterised LPIS 2018 perimeter length (km)	1,555,121
Total Ireland rasterised waterbody perimeter length (km)	494,327
Total breakthrough line length (km)	36,520
Total breakthrough line length (% of Ireland rasterised LPIS 2018 perimeter length)	2.3
Total breakthrough line length with surface runoff P load ≥ 1 kg y^{-1} (km)	14,301
Total breakthrough line length with surface runoff P load ≥ 1 kg y^{-1} (% of Ireland rasterised LPIS 2018 perimeter length)	0.9
Total delivery line length (km)	13,161
Total delivery line length (% of Ireland rasterised waterbody perimeter length)	2.7
Total delivery line length with surface runoff P load ≥ 1 kg y^{-1} (km)	6050
Total delivery line length with surface runoff P load ≥ 1 kg y^{-1} (% of Ireland rasterised waterbody perimeter length)	1.2

Table 2.4. National analysis of breakthrough and delivery lines[a]

	Breakthrough lines			Delivery lines		
Surface runoff P load group	Number	Total length (km)	Percentage of total delivery	Number	Total length (km)	Percentage of total delivery
0.1 to < 1.0	740,284	22,219	18.8	738,906	7112	14.6
1.0 to < 2.0	110,206	5850	11.9	119,180	1803	9.8
2.0 to < 3.0	44,600	2638	8.3	50,614	971	7.2
3.0 to < 4.0	24,308	1485	6.4	28,675	621	5.8
4.0 to < 5.0	15,118	987	5.1	18,406	437	4.8
5.0 to < 6.0	9984	631	4.2	12,740	323	4.1
6.0 to < 7.0	6954	440	3.4	9306	248	3.5
7.0 to < 8.0	5277	370	3.0	6926	197	3.0
8.0 to < 9.0	3893	257	2.5	5375	162	2.7
9.0 to < 10.0	3076	209	2.2	4352	136	2.4
10.0 to < 25.0	14,537	1083	16.4	21,157	773	18.4
25.0 to < 50.0	2970	252	7.7	4676	231	9.1
50.0 to < 75.0	698	43	3.2	974	61	3.4
75.0 to < 100.0	256	20	1.7	332	23	1.7
> 100.00	386	36	5.1	583	62	9.4
Totals	982,547	36,520	100	1,022,202	13,160	100

[a]Table based on results for P index 3 soils.

SLAM estimated that, from 2012 to 2014, annual P emissions from these catchments from pasture and peatlands amounted to 996 t y^{-1}. Again, this figure is TP load rather than TRP load. Total surface runoff P loads predicted nationally from DiffuseTools ranged from 718 to 1536 t y^{-1} from P index 2 and 3 scenarios, bracketing the SLAM estimate. Mockler *et al*. (2017) also reported midranging P export rates of 0.25–0.5 kg ha^{-1} y^{-1}, coinciding with agricultural lands with poorly draining soils, compared with a range (for the 16 catchments) of 0.24–0.52 kg ha^{-1} y^{-1} from DiffuseTools P index 2 and 3 scenarios, for LPIS 2018 agricultural fields with > 50% poorly drained soils (poorly/very poorly/alluvium/peat). However, DiffuseTools model results do not implicitly account for the hydrological disconnection of P load pathways, via the slowdown and impedance of flow from topographic flow sinks or vegetation, although the national flow sink depth map (Figure 2.7) facilitates future analysis. Actual total surface runoff P loads (summed over the entire country) may be lower than the DiffuseTools model estimates.

Checking model with Teagasc ACP water quality data

Measured annual total TRP loads in quickflow from four Teagasc ACP catchment outlets (Ballycanew, Dunleer, Timoleague and Castledockrell) from 2010 to 2017 were compared with total DiffuseTools surface runoff P loads (5-m scale) values calculated for each catchment. The catchments are described in Sherriff *et al*. (2015). Average measured ACP TRP loads (kg y^{-1} and kg ha^{-1} y^{-1}) were comparable to DiffuseTools total surface runoff P loads for all catchments and scenarios 2 and 3, except for Timoleague. This is expected because analysis of national Teagasc soil samples shows that the mean soil Morgan P concentrations in all Irish counties and sectors are typically between P index 2 and 3 (Teagasc, 2020). In the Timoleague catchment, where monitored results (showing higher P loads) did not match modelled risk well, the Soils Hydrology Ireland map indicates that the catchment is dominated by well-drained soils and the majority of the small pockets of poorly drained soils do not coincide with agricultural fields. Thus, HSDMv3 results predict only very small areas of surface runoff generation in the catchment, which do not typically coincide with agricultural soil P concentrations, and hence modelled P loads were very small. In such catchments, dominated by intensive dairy farming, the higher observed P loads could be caused by farmyard pollution to headwater streams (Harrison *et al*., 2019) or poaching/cattle access to watercourses (Kilgarriff *et al*., 2020).

Checking flow pathway evidence from aerial imagery

Aerial imagery was used to find visual evidence of CSAs, surface runoff pathways, delivery points and flow sinks from soil gleying, soil erosion and gullies, vegetation changes, spatial patterns and other visual cues. Maps of modelled flow sinks, surface runoff P loads (5-m scale) and accumulated surface runoff volumes were then overlaid to see whether or not the model correctly identifies locations and spatial extents (Figures 2.13–2.15). The model shows very good agreement with surface runoff pathways (delivery paths of soil P) (Figure 2.13) and good agreement with the locations of CSAs (Figure 2.14), although with the latter the spatial extents were often partially missing, attributed to the Soils Hydrology Ireland polygon extents of poorly or very poorly drained soils. Thus, local field surveyed soil maps should be used where possible to check model accuracy. As a workaround in areas where CSAs are observed or suspected but are missed by the model, the TWI map, which predicts runoff risk from topography (shallow slopes and high flow accumulation), could be used to aid decision-making and provide additional information (see Figure 2.16 as an example). Flow sinks were accurately identified from the model (Figure 2.15), as shown by surface water accumulation, wetland vegetation patterns and soil discolouration indicating waterlogging.

Checking water balances using GSI recharge estimates

To sense-check annual water balances derived from the HSMDv3 and attempt to close water balances nationally, a conceptual annual water balance model (mm y^{-1}) was developed for all flow pathways, differentiated by aquifer type. This conceptual model integrates surface runoff and percolation (drainage) depths predicted from HSMDv3 with groundwater recharge predicted by GSI modelling. The latest groundwater recharge map (mm y^{-1}) from GSI (T. Hunter Williams, July 2020, personal communication, unpublished) used the Soils Hydrology Ireland map and 30-year (1981–2010) annual average effective rainfall (mm y^{-1}) calculated by Werner *et al*. (2019), although this was derived from HSMDv2 (using different parameter values) and so there are some differences. The conceptual model was applied nationally by subtracting groundwater recharge from the 30-year annual average percolation map developed by HSMDv3, to calculate subsurface flow, which combined interflow, transition zone flow and shallow groundwater flow (mm y^{-1}). The GSI recharge maps are for minimum dependable amounts of recharge to deep groundwater layers only (i.e. estimates tend to be conservative at their lower limits). Thus, it was not possible to disaggregate and quantify shallow groundwater flow from transition zone flow and interflow, and hence to separate pollutant losses

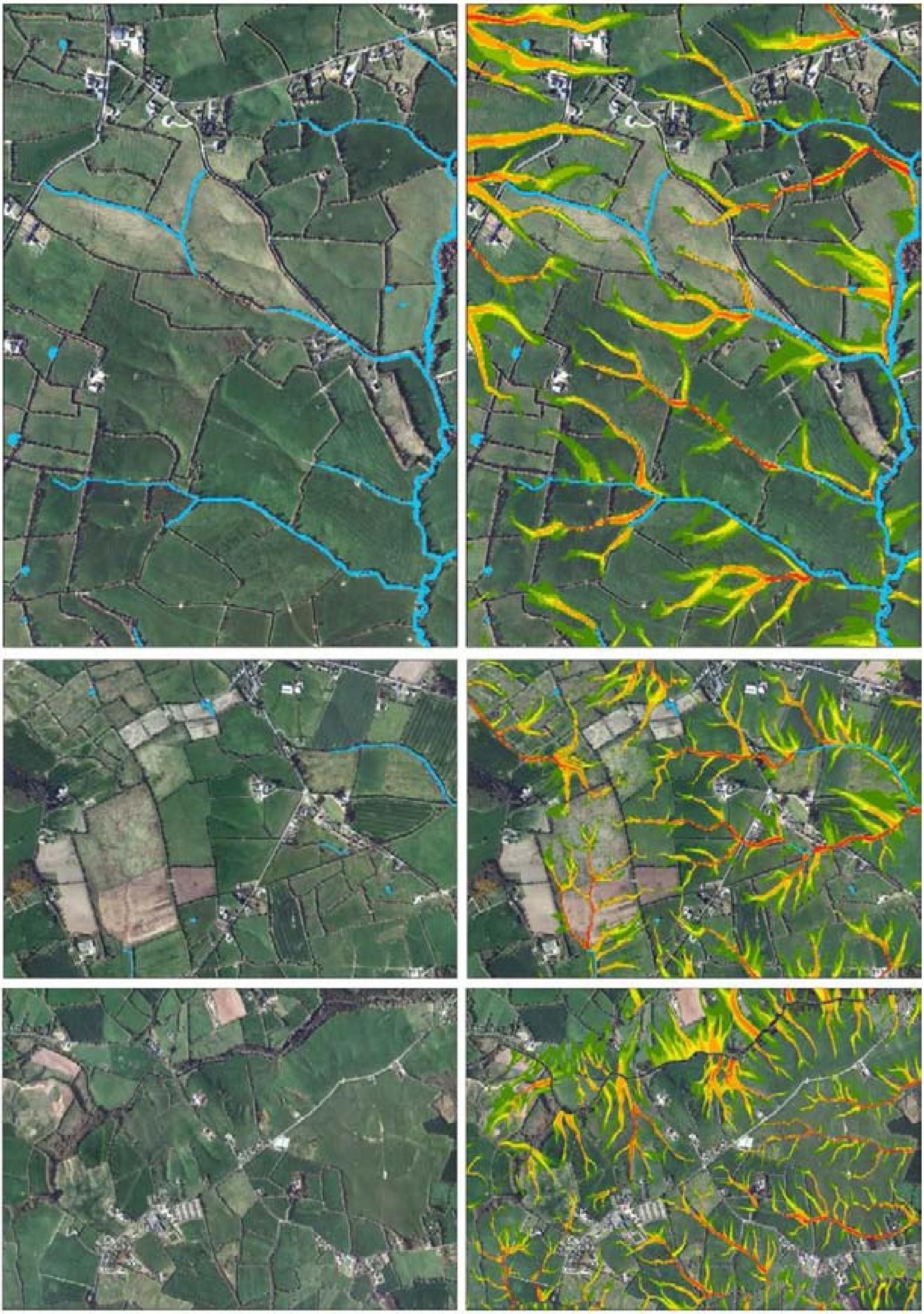

Figure 2.13. Aerial imagery showing visual evidence of gullies from surface runoff pathways (blue, left-side images) and surface runoff P loads (accumulation) overlaid (right-side images), with redder areas accumulating higher surface runoff P loads. © Ordnance Survey Ireland, licence 2019/OSi_NMA_074.

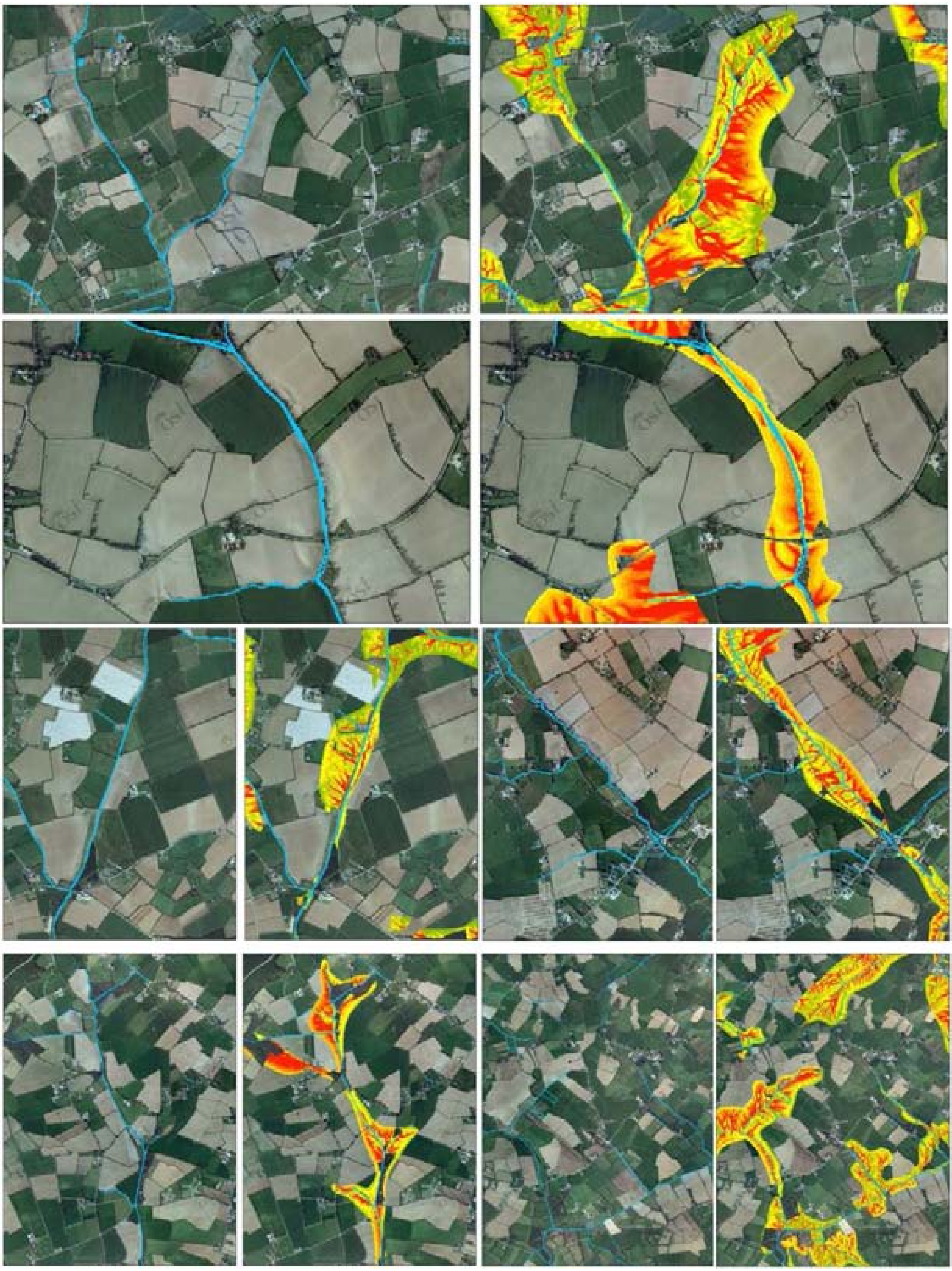

Figure 2.14. Aerial imagery showing evidence of CSAs (gleyed soils and gullies) (left) and surface runoff P loads (5-m scale) map overlaid (right), with redder areas generating higher surface runoff P loads. © Ordnance Survey Ireland, licence 2019/OSi_NMA_074.

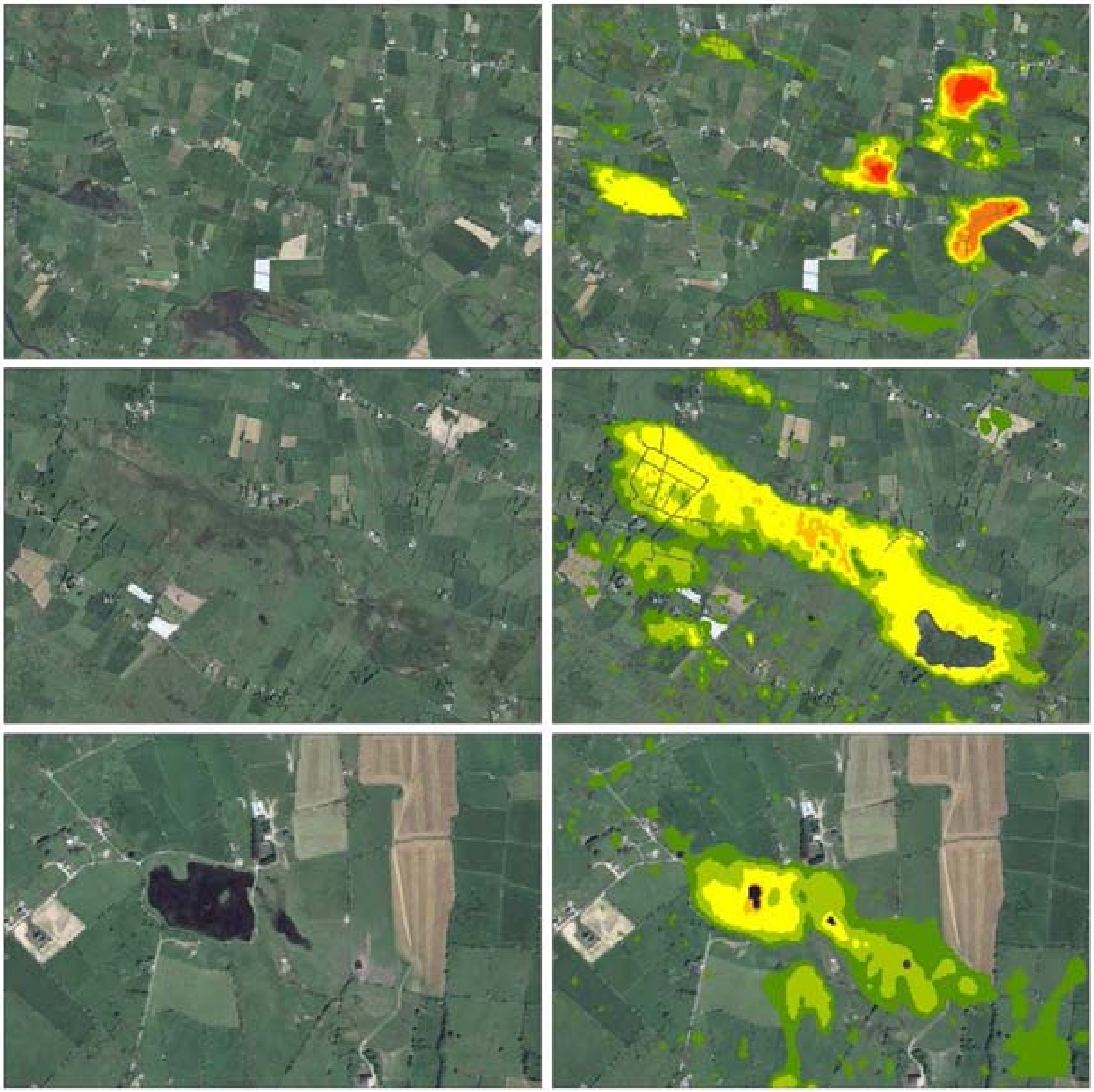

Figure 2.15. Aerial imagery of evidence of flow sinks (left) and modelled flow sinks overlaid (right), with redder areas being higher flow sink depths. These features can hydrologically disconnect surface runoff pathways to waterbodies. © Ordnance Survey Ireland, licence 2019/OSi_NMA_074.

within each pathway. The resulting subsurface flow map (mm y^{-1}), shown in Figure 2.17, indicates that water balances from both models (HSMDv3 and GSI groundwater recharge) correspond well. Exceptions include some small areas with "negative balances" (< 0 mm y^{-1}) typically located in waterbodies and small karst features, where recharge coefficients were 85% and hence very high groundwater recharge was predicted; such features are not considered separately in calculations in HSMDv3.

2.4 Discussion

2.4.1 Model assumptions

There are several model assumptions in the approach described above that are explicitly stated below:

- Soil P concentrations (in mg l^{-1}) are assumed to equate to soil P load (in kg ha^{-1}) using an assumption of 10-cm sampling depth and 1 g cm^{-3} bulk density across the country.

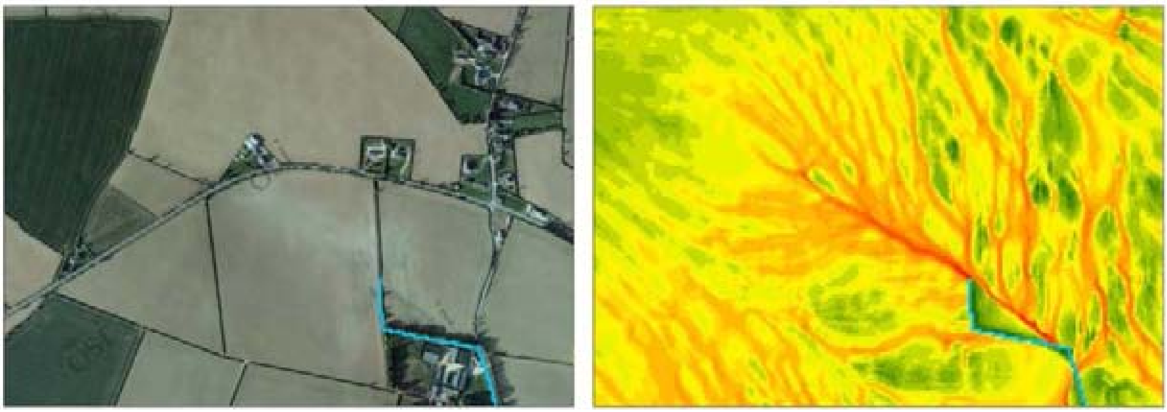

Figure 2.16. Example of a CSA (gleyed soils and soil erosion; left) not accurately modelled in the surface runoff P loads (5-m scale) map. The difference is due to it being defined as well-drained soil in the Soils Hydrology Ireland map. The TWI map (right) shows high topographic risk of surface runoff generation due to low slope gradients and high flow accumulation (redder areas). © Ordnance Survey Ireland, licence 2019/OSi_NMA_074.

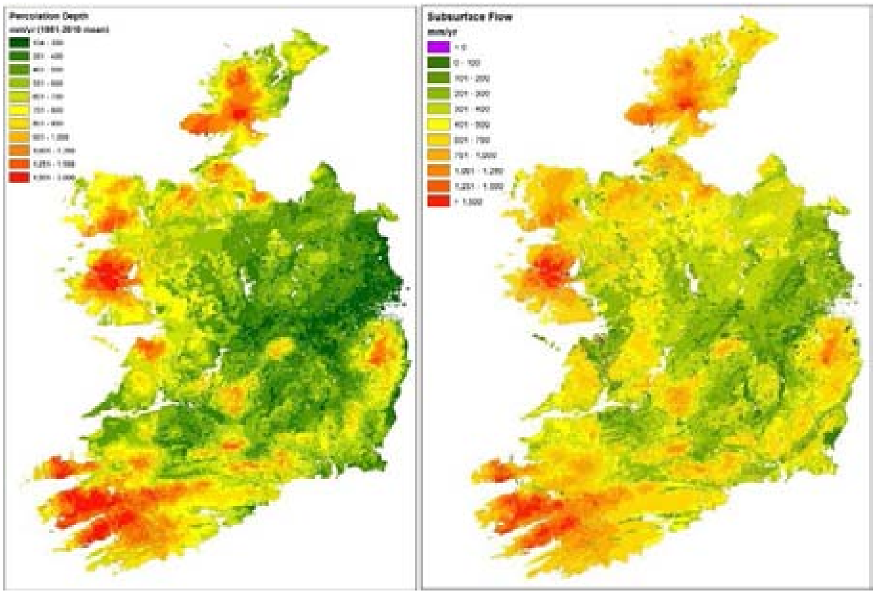

Figure 2.17. DiffuseTools subsurface flow map (mm y^{-1}; right), lumping interflow, transition zone flow and shallow groundwater flow, based on 30-year annual averages (1981–2010). Note that a small number of purple areas (< 0 mm y^{-1}) are an artefact of subtracting GSI groundwater recharge from HSMDv3 percolation depth (left).

- The model is run for four scenarios, each with a soil P concentration at the midpoint of the P index range.
- Only 5% of accumulated upslope surface runoff volume (plus 100% of the overland flow volume generated at the cell itself) is assumed to interact immediately with the soil (and soil P).
- Peats are assumed to act like poorly drained soils.

2.4.2 Limitations and uncertainties

All input datasets have some limitations, errors, uncertainties and/or data gaps, which are addressed below grouped by the nutrient transfer continuum components.

Nutrient sources:

- Spatial variation in livestock densities, fertiliser applications and excreted P is not considered.

- LPIS 2018 may be spatially inaccurate, and the dominant land use in a parcel is used in this model.
- Within-field nutrient local source hotspots are not mapped and so are not modelled.
- Point sources of agricultural nutrients are not considered.

Mobilisation:

- The model does not account for differences in soil P mobilisation potential from different soil types due to soil geochemistry (e.g. soil organic matter, pH, aluminium, iron, calcium, degree of phosphorus saturation).
- The model does not account for source-limited changes in soil P mobility and availability for loss.

Pathways:

- The national DEM used has vertical and horizontal accuracies of 1 m and 2 m, respectively, and the 5-m grid resolution is too coarse to account for some microtopographic flow diversions.
- The model does not account for surface runoff from local features such as poaching/surface sealing/soil compaction, infiltration–excess overland flow, subsurface return flow (e.g. springs) or subsurface drains.
- The HSMDv3 model uses five soil drainage classes instead of individual soil types.
- Quickflow data used to calibrate HSMDv3 are estimated with associated uncertainties.
- HSMDv3 used three Teagasc ACP catchments for calibrating parameter values. Although chosen as representative of typical Irish agri-environmental conditions nationally (Fealy *et al*., 2009), they nonetheless represent only a small proportion of the total agricultural land area, and applying results nationwide introduces some uncertainties.
- MÉRA weather data are modelled (not measured) and have inherent uncertainties and area at 2.5 km^2 resolution.
- Well-drained soils are assumed to generate no surface runoff in the HSMD, but some well-drained soils will generate runoff following intense rainfall, particularly during the winter months.
- Estimates of surface runoff P losses ($kg\,ha^{-1}\,y^{-1}$) were capped so that they could not exceed the initial soil P concentration, which would not be realistic.
- The clipping of datasets by WFD river sub-basin, catchment or river basin district (RBD) for the purposes of computer processing can sometimes result in "over the cliff" artefacts at these sharp borders.
- The model does not consider subsurface pathways.

Delivery to receptors:

- Some waterbodies may be missing or spatially inaccurate, and 5-m rasterisation exaggerates stream widths.
- The r.watershed algorithm used in QGIS GRASS to accumulate flow and P loads downslope is not able to account for well-drained soils acting as sinks. Although the final maps erase accumulations within well-drained soils, the accumulations downslope in other soil drainage classes are still shown and the results may be overpredicted.
- The model focuses on annual average dissolved P losses from diffuse soil P sources and does not consider particulate P or incidental losses following fertiliser applications.

2.4.3 Interpreting the maps

Surface runoff volume and P load maps are available at five different scales, each providing unique information and being more suited to specific categories of users/stakeholders.

1. Surface runoff (5-m scale): this scale map shows surface runoff estimated at each 5-m raster grid cell, and hence identifies CSAs as areas generating higher amounts of runoff. Values are found only within agricultural fields.
2. Surface runoff (accumulation): this scale map shows the accumulated surface runoff from the entire UDA that passes through the 5-m grid cell on its way downslope to the waterbody. This map can be used to identify delivery paths of pollution and where it "ends up" outside the field, and hence was used in pinpointing breakthrough and delivery points/lines.
3. Surface runoff (field scale): total surface runoff volume and P load is estimated within each LPIS 2018 field parcel. The feature class attribute table

also includes mean slope and soil drainage class per cent coverage.

4. Surface runoff (UDA scale): total surface runoff volume and P load estimated to come from each UDA.
5. Surface runoff (WFD river sub-basin scale): total surface runoff estimated to be generated within each WFD river sub-basin. The feature class attribute table also includes LPIS utilised agricultural area.

These maps were generated at different scales to suit different stakeholders. For example, catchment managers and LAWPRO may find the 5-m scale and accumulation maps more useful, particularly for pinpointing where to go during on-the-ground assessments, understanding where diffuse pollution enters the waterbodies, and determining where to target individual source, mobilisation or transport mitigation measures to reduce losses. Farmers or Teagasc ASSAP staff may find field-scale maps more appropriate, as this is the farm management scale they work with day to day. In addition, users focusing on particular stretches of watercourses may want to know the pollution generated in the entire UDA of that segment of watercourse, which may encompass many fields or parts of fields within the hillslope. Finally, for displaying more visually interpretable results nationally, and for focusing policies on PAAs, WFD river basin-scale maps may be more appropriate. It is important to state that all maps are DSTs that can be used to aid decision-making, but they do not replace local on-the-ground knowledge, which can contradict or enhance interpretation, and they should not be used to make decisions on their own (see Djodjic *et al*., 2018).

Thus, it is important to select the results of the appropriate P index scenario for the area of interest; for example, P index 1 for low-intensity farms such as commonage in the Wicklow mountains or P index 3 or 4 scenarios for high-intensity, derogated dairy farms in Cork. Analysis of national Teagasc soil samples over the last few years shows that the mean soil Morgan P concentrations in all Irish counties are typically between P index 2 and 3 (Teagasc, 2020). Thus, it is expected that surface runoff P loads would typically range between those predicted in P index 2 and 3 scenarios over subcatchment to catchment scales. However, Teagasc national soil sampling results show that soil P concentrations have increased significantly from 2018 onwards, reversing a decreasing trend up until 2017, resulting in the percentage of soil samples being P index 3 and P index 4 increasing from 21% to 24% and from 16% to 26%, respectively, driven mainly by increases in dairy and drystock. Thus, the results for P index 3 and 4 scenarios are becoming more relevant as soil P source pressure trends continue to increase following intensification and increases in the national livestock herd from a low of 6.49 million cattle in 2011 to a record high of 7.36 million in 2017 (+13.4%) and currently 7.31 million in 2020 (provisional; CSO, 2020).

2.4.4 Pollutant Impact Potential for phosphorus (PIP-P)

The latest EPA PIP-P v3 map update utilises LPIS 2018 data (instead of LPIS 2012 data, as in PIPv2) and a new map of artificially drained grassland for the border, Midlands and western regions (O’Hara *et al*., 2020), and overlays the DiffuseTools overland flow paths from the surface runoff (accumulation) map (Figure 2.18). The map is now made of the following three layers that are combined to visualise the movement of P losses across the landscape:

1. *PIP-P: CSA map.* A phosphorus CSA occurs where there is a diffuse source of P from agricultural areas and the land is susceptible to losses. Source loading data for cattle, sheep and crops are based on 2018 farm management data from DAFM. A “high PIP” (rank 1, 2 or 3) area is typically due to the presence of poorly draining soils and moderate/high livestock intensity.
2. *PIP: focused delivery flow paths*. Focused delivery flow paths are the areas of converging runoff, determined by this project [i.e. the surface runoff (accumulation) map], which results in an increasing accumulation of flow. The red flow paths have the highest accumulations of surface runoff. Where these cross “high-PIP” areas, expect higher P losses. The map can highlight areas to target phosphorus pathway interception actions, for example hedgerows.
3. *Focused delivery points.* Focused flow delivery points occur where focused flow paths enter a watercourse. The size of the point indicates the relative volume of flow delivered to water. It is important to consider the available source of

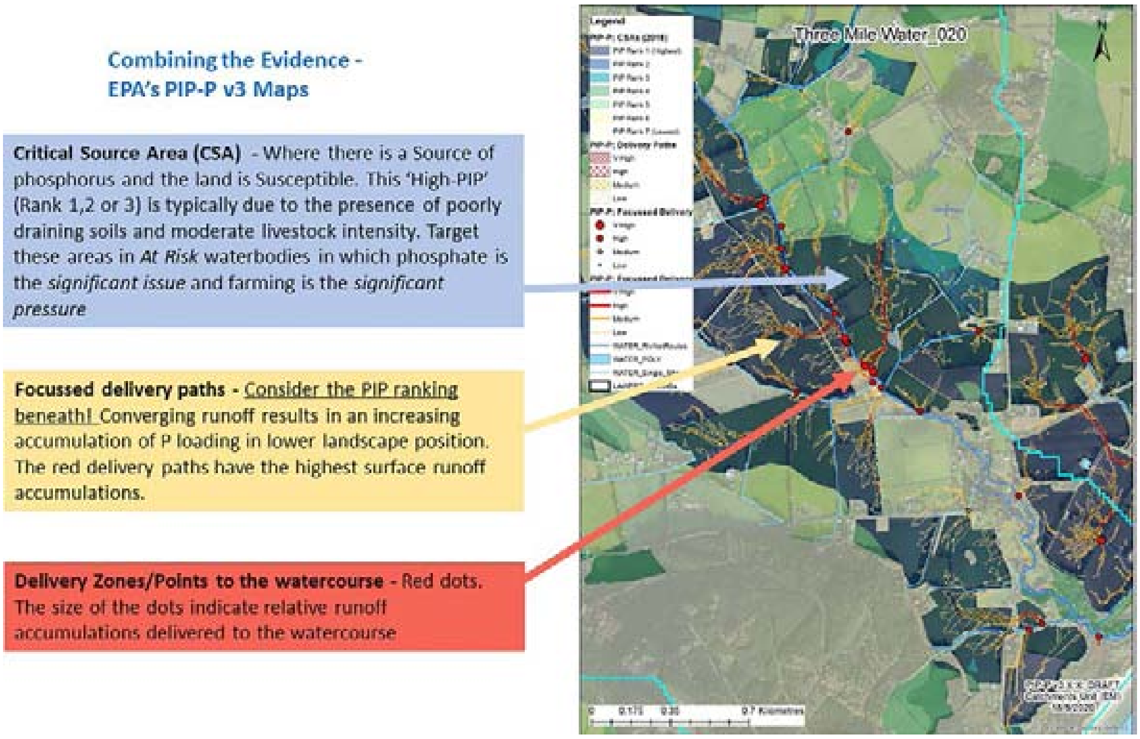

Figure 2.18. Focused delivery paths and points at watercourses overlaid onto the new EPA PIPv3 maps (https://gis.epa.ie/EPAMaps/Water) that account for LPIS 2018 livestock densities. © Ordnance Survey Ireland, licence 2019/OSi_NMA_074.

phosphorus in the upslope contributing areas. The map can highlight areas to target phosphorus pathway interception actions, for example riparian/buffer zones, woodlands, engineered ditches.

Livestock densities are accounted for in the latest PIPv3 maps for phosphorus under development by the EPA Catchments Unit, and hence our DiffuseTools results can be overlaid with these to provide additional information about delivery points, and these were provided to LAWPRO for checking. These maps allow stakeholders to inform changes to farming practices (e.g. where not to spread slurry) and to pinpoint measures in a "treatment-train" approach that targets different stages of the source–mobilisation–transport–hydrological connectivity–delivery continuum (Thomas *et al*., 2016b). Targeting measures at breakthrough points located upslope nearer the in-field source, or at delivery points downslope adjacent to waterbodies, maximises the size of UDAs being mitigated (particularly delivery points), so is likely to be more cost-effective than blanket implementation, minimises the area of land taken out of agricultural production and minimises disturbances to farming practices. Importantly, this mitigation approach facilitates a multi-pollutant framework, which could be applied to other diffuse pollutants transported in surface runoff (e.g. nutrients, sediments, pesticides, chemicals and pathogens) to mitigate losses (Bloodworth *et al*., 2015; Thomas *et al*., 2016b). The maps could also be used within WFD RBMPs, agri-environmental schemes (e.g. GLAS) and species conservation schemes (e.g. freshwater pearl mussels), and support environmentally sustainable intensification goals set out by Food Wise 2025.

3 Modelling Diffuse Source Sediment Pollution

3.1 Introduction

The traditional approach for sediment modelling focuses on the prediction of sediment export from the catchment (modelling sediment flux or annual yields), and such models can range from complex and process-based numerical models, through conceptual catchment-based models, to those based on simple regression and lumped prediction equations. The latter, even with their reduced complexity and reduced data requirements, can provide a general indication of sediment loads and can contribute to the assessment of water resources at larger scales (Rymszewicz *et al.* 2018).

Obtaining measurements of flow and sediment concentration for sediment flux determination are resource intensive and challenging. The use of sediment yield and transport models is an attractive alternative to estimating loadings to river catchments, particularly in those that are ungauged or where data records are insufficient to produce meaningful estimates of sediment flux or yield.

Models can be broadly divided into four main types:

1. physically based models;
2. empirical models;
3. conceptual/empirical models;
4. semiquantitative models.

Type 1, 2 and 3 models are well reviewed in the literature (Jetten *et al.*, 2003). Semiquantitative models (type 4) have been described in de Vente and Poesen (2005). For brevity, only types 2 and 3 are discussed below.

3.1.1 Empirical catchment scale models

Empirical models are based on the analysis of data. Empirical models are often single equations and the most frequently used equation for estimating sediment generation from land surfaces, the revised universal soil loss equation (RUSLE), is one example (Foster *et al.*, 2001). Annual soil loss using this equation is described by five multiplied factors:

$$A = R*K*LS*C*P \tag{3.1}$$

where A indicates mean annual soil loss ($\mathrm{t\,ha^{-1}\,y^{-1}}$), R is the rainfall erosivity ($\mathrm{MJ\,mm\,ha^{-1}\,h^{-1}\,y^{-1}}$), K is the soil erodibility ($\mathrm{t\,ha\,h\,ha^{-1}\,MJ^{-1}\,mm^{-1}}$), LS is a dimensionless slope length, steepness factor, C, is a dimensionless parameter that reflects crop cover and management, and P is a dimensionless support practice factor that accounts for variations in soil loss for different surface conditions.

The EPA-funded project SILTFLUX showed that, in principle, the approach could be applied in Ireland, but that RUSLE, as it currently stands, overestimates sediment amounts, and considerable work would be required to adjust its factors for Irish conditions (Rymszewicz *et al.*, 2014). However, the SILTFLUX project developed an empirical equation to directly estimate annual suspended sediment yield in rivers draining small agricultural catchments (Rymszewicz *et al.*, 2018). From assessment with an Irish dataset, it had a Nash–Sutcliffe (NS) coefficient above 0.7 in both calibration and validation. However, additional work is needed to develop additional equations for larger catchments and for different types of land use.

3.1.2 Conceptual models

Conceptual models rely on a simplified representation of the processes and factors that describe soil loss and sediment delivery to the watercourses. However, for sediment, these models are typically conceptual in relation to their hydrological components, but their sediment modelling is usually based on various adaptations (including the revised form) of the empirical (R)USLE, (Renard *et al.*, 1997), which is the most widely used empirical formulation for estimating soil erosion.

To obtain sediment yields, conceptual models combine information on soil loss with transport capacity equations or with values of sediment delivery ratio (SDR). The main disadvantage of (R)USLE-based models is that, in most cases, they do not account for mass movement, or gully and bank erosion. However, both physically based and conceptual models can be spatially distributed, providing potential information on sediment sources and sinks within the catchment, and this represents a distinct advantage over other models.

3.1.3 High-resolution modelling using SCIMAP and the Network Index

The aim of the Sensitive Catchment Integrated Modelling and Analysis Platform (SCIMAP) is to identify where, within a large river catchment, diffuse pollution is most likely to come from. This permits spatial targeting of more detail investigations of soil erosion and the implementation of mitigation measures within the catchment at locations where they are most likely to be effective. SCIMAP uses a detailed geospatial approach with landscape extent coupled with subfield detail, to capture how the mosaic of land use, topographic detail and hydrological flow paths integrate to give rise to diffuse pollution problems. The SCIMAP methodology is described in Reaney *et al*. (2011).

3.1.4 Catchment-scale modelling including SimplyP

In addition to high-resolution, slope or field-scale sediment modelling, catchment-scale models are also used, as described in section 2.1.4. Here we test the usefulness of the SimplyP (Jackson-Blake *et al*., 2017) model as an example of the conceptual modelling approach. It is a dynamic (daily time step) semi-distributed model that simulates three separate water pathways from the land to the river: (1) quickflow, (2) soil water flow and (3) groundwater flow (Figure 3.1). In its conceptualisation, dissolved P can be transported only via soil water flow and groundwater, and particulate P and sediment are transported only via quickflow. SimplyP can have up to 45 parameters, but only five require adjustment or calibration; the others can be set from characteristics of the catchment. SimplyP also includes an internal simple hydrological model to estimate flow along each of the three water pathways modelled.

3.2 SCIMAP Estimates of Erosion Risk

The SCIMAP approach estimates soil erosion risk at fine scales (typically at subfield scale) and can integrate the results to coarser scales as required for managing the risk. The methodology is described in Reaney *et al*. (2011) and here it is used with the 5-m DEM to estimate the following for Irish catchments:

1. hydrological connectivity;
2. mobilisation potential;
3. erosion export potential.

The calculation was performed in two stages. The first stage of the preprocessing burnt the channels and other linear flow features into the surface to ensure

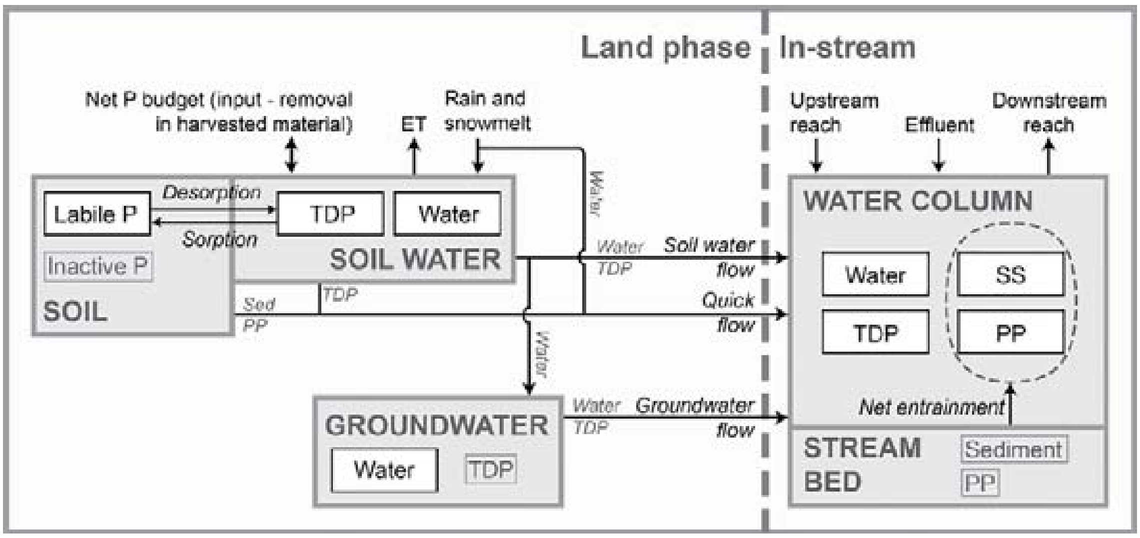

Figure 3.1. SimplyP model schematic. PP, particulate phosphorus; SS, suspended sediment; TDP, total dissolved phosphorus. Reproduced from Jackson-Blake *et al*. (2017), with permission from Wiley & Sons, Inc.

the correct hydrological routing at the catchment scale. The second stage was to remove hydrological blockages resulting from data errors within the digital terrain model. This was achieved using the “Deepen Drainage Routes” options on the “Fill Sinks” tool within SAGA GIS. This algorithm was selected because, compared with algorithms that fill the sink, it results in much less geomorphic change within the topography.

3.3 SCIMAP Results

3.3.1 SCIMAP erosion risk map

The SCIMAP national erosion risk map is provided as raster files for each RBD for use in a GIS. As an example, the South Eastern RBD is shown in Figure 3.2, and a close-up of a section of the map is shown in Figure 3.3. In these maps, the red colour shows surface waterbodies or areas of land directly connected to surface waterbodies and thus having higher risk, and the darkest green colour shows areas with least connectivity and lower risk.

3.4 SCIMAP Discussion

Within this application of the SCIMAP approach, the focus was on the relative risk within each of the catchments for the export of sediment and nutrients. A set of detailed mobilisation potential, hydrological connectivity and erosion potential maps have been produced for Ireland. These potential maps have been produced with constant land cover risk weights and

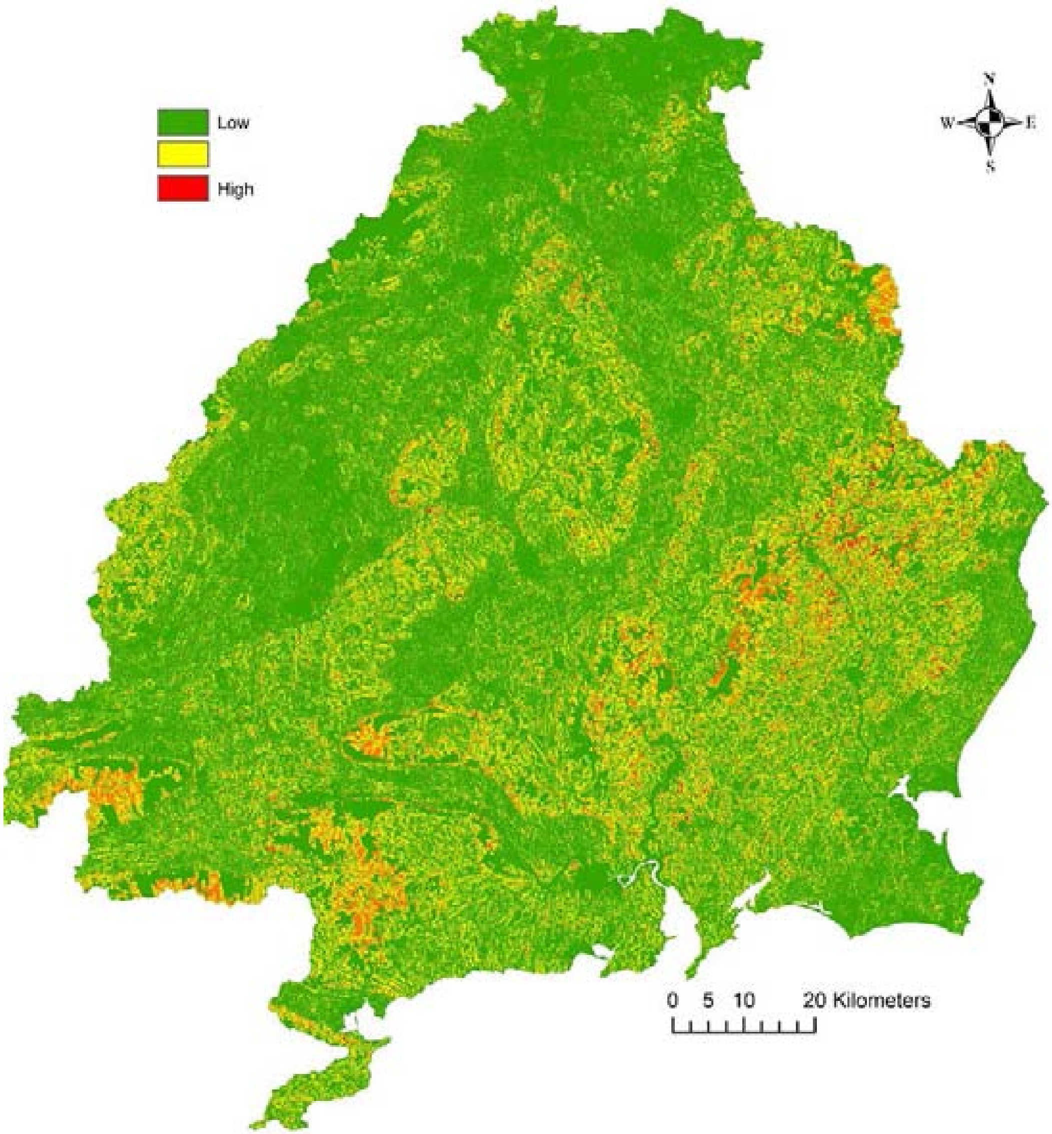

Figure 3.2. Example of the SCIMAP erosion risk map for the South Eastern RBD.

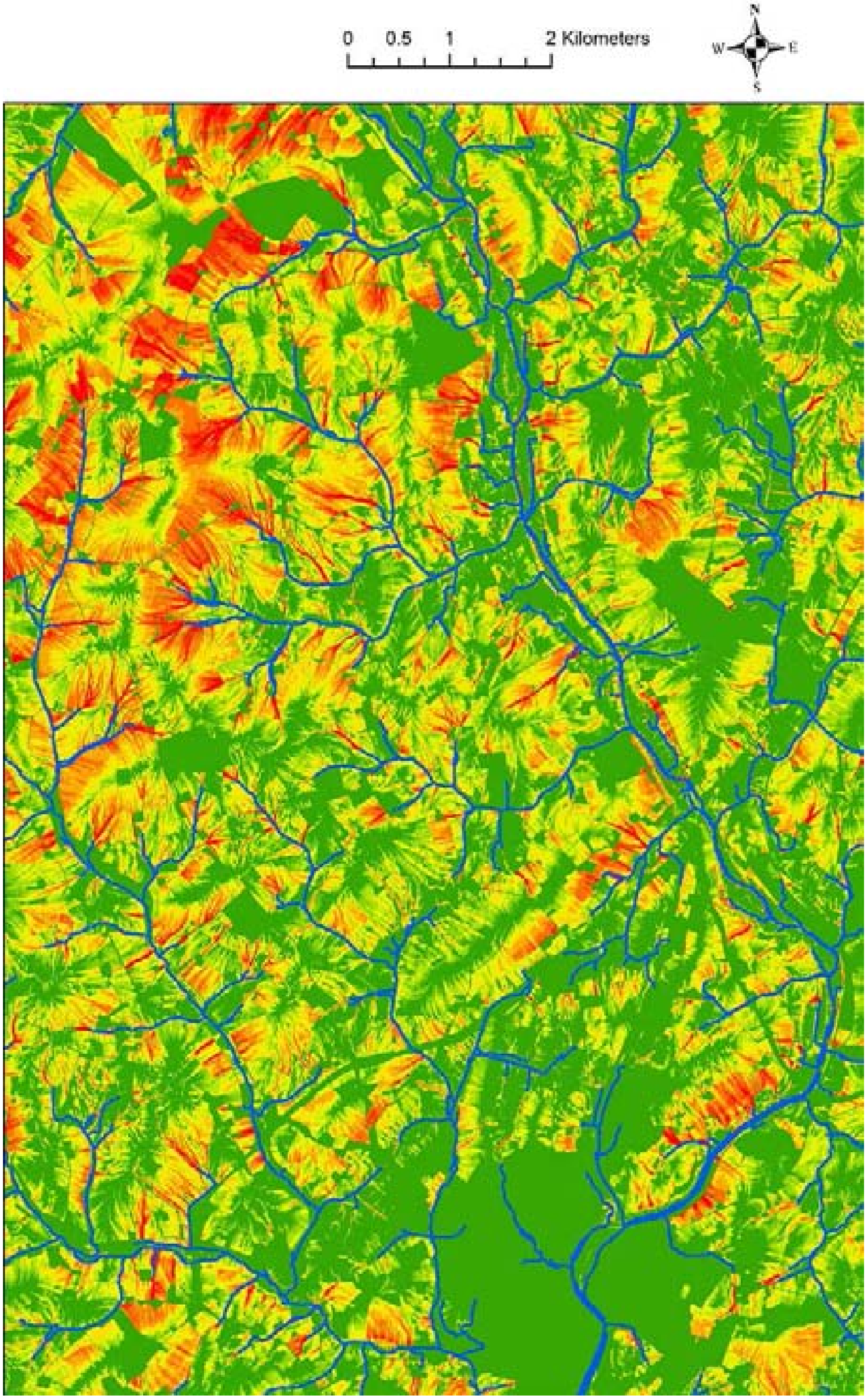

Figure 3.3. Close-up of section of the SCIMAP erosion risk map of the South Eastern RBD at 5-m grid scale (areas of higher erosion risk are shown in red and areas of low risk are shown in green).

constant rainfall. The risk mapping could be extended and refined with the use of the observed spatial pattern of rainfall across Ireland and the land cover information to give different risk weightings.

The SCIMAP approach uses the topographic wetness index (TWI) (Beven, 1979) to make predictions of the spatial pattern of soil moisture within the catchment. Normally, this index is applicable only within surface water catchments and has limited predictive power within groundwater catchments. However, it is important to consider the hydrological conditions and soil moisture status during large storm events that result in sediment, and particularly particulate P, being exported to the river channels. Under these conditions, most catchments export this material along surface pathways; therefore, the SCIMAP mapping does have value and information even for catchments with large groundwater base flow contributions.

3.5 Catchment-scale Modelling: SimplyP Methodology and Results

3.5.1 Test catchments

The SimplyP model was applied to 5 years of daily input data for rainfall and potential evapotranspiration (PET) for the Ballycanew catchment and 4 years of similar data from the Timoleague catchment. Most of the required parameters were determined directly from a digital terrain model, river reach and land use maps for the catchment using ArcGIS. SimplyP simulates flows, suspended sediment (SS), total dissolved P and particulate P concentrations, but here we concentrate on SS. The catchment was treated as a single unit (lumped) with only minor adjustments of initial values of parameters. A comparison of model output (red lines) with measured values (grey lines for flows and grey points for SS concentrations) in Ballycanew is shown in Figure 3.4. These are without automatic calibration and show a moderately good fit to the flow record, particularly the recessions and low flows. However, the model tends to substantially underestimate the peak flows of storm events. The fit of the uncalibrated model to the SS data is less good, with the model having the appropriate dynamic response to rainfall, but greatly underestimating the higher SS concentrations. This is understandable given the underestimation of the peak flows that are likely to be associated with heavy rainfall, and which have greater ability than low flows to mobilise and transport sediment. The underestimations of the high flows and SS are closely related, as the model links sediment transport exclusively with quickflow. The NS coefficient for the fit of flows is 0.72 and for SS is 0.35, and the model bias in these quantities is 21% and –3.5%, respectively.

The calibration facility of the Mobius implementation of SimplyP was then used to adjust the parameters to obtain a better fit to the measured data. The results for discharge are shown in Figure 3.5. The NS coefficient for the fit of flows declined marginally to 0.71, but the fit to the measured SS series improved substantially (Figure 3.6), showing a better attempt at modelling peak concentrations in higher flows than the uncalibrated case, with the NS coefficient increasing considerably to 0.61, which is just satisfactory for SS (Rovira and Batalla, 2006).

A similar comparison of model output (red lines) with measured values (grey lines for flows and grey points for SS concentrations) for Timoleague is shown in Figure 3.7. This shows a moderately good fit to the flow record, particularly the recessions and low flows. However, as for Ballycanew, the model also tends to underestimate the peak flows of storm events. The fit to the SS data is not good, with the model having the appropriate dynamic response to rainfall, but greatly underestimating the higher sediment concentrations and overestimating the lower concentrations. The NS coefficient for the fit of flows is 0.65, but for SS is 0.26, which is not good. The model bias in both these quantities is about 4%. The underestimations of the high flows and SS are closely related. Using the

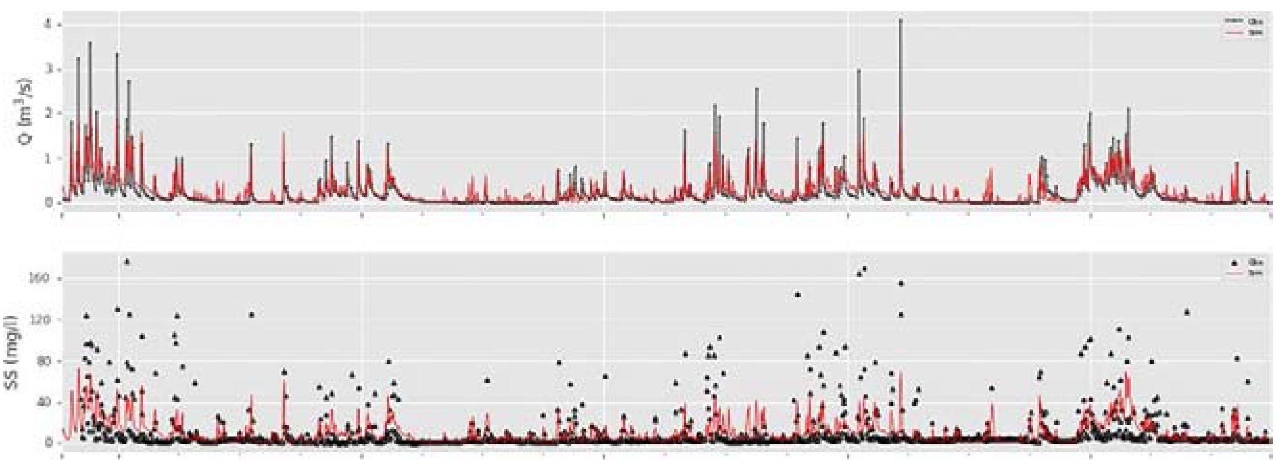

Figure 3.4. SimplyP flow (top) and SS concentration (bottom) for Ballycanew. Uncalibrated red line is model output; grey line and black points are measured data.

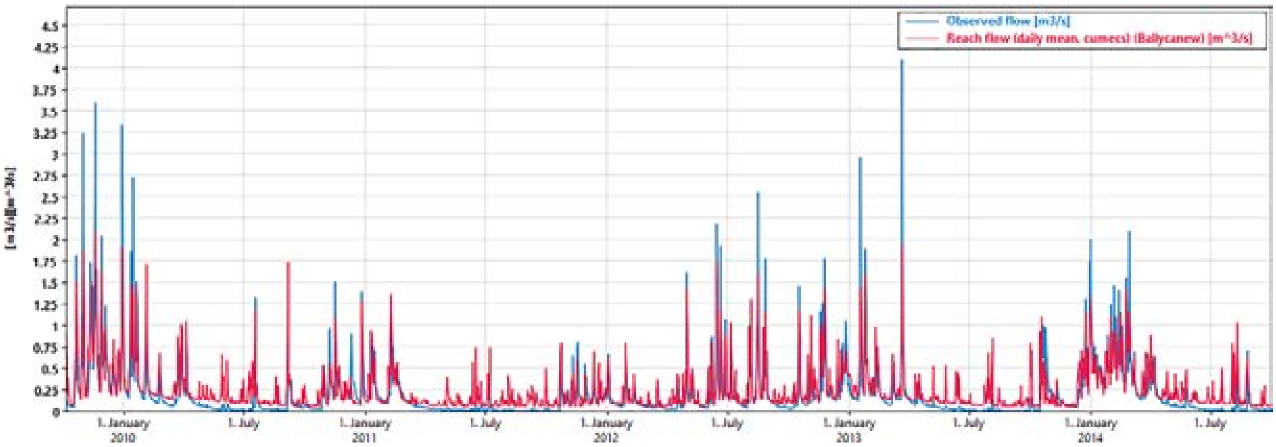

Figure 3.5. SimplyP discharge fit from automatic optimisation (Mobius) for Ballycanew.

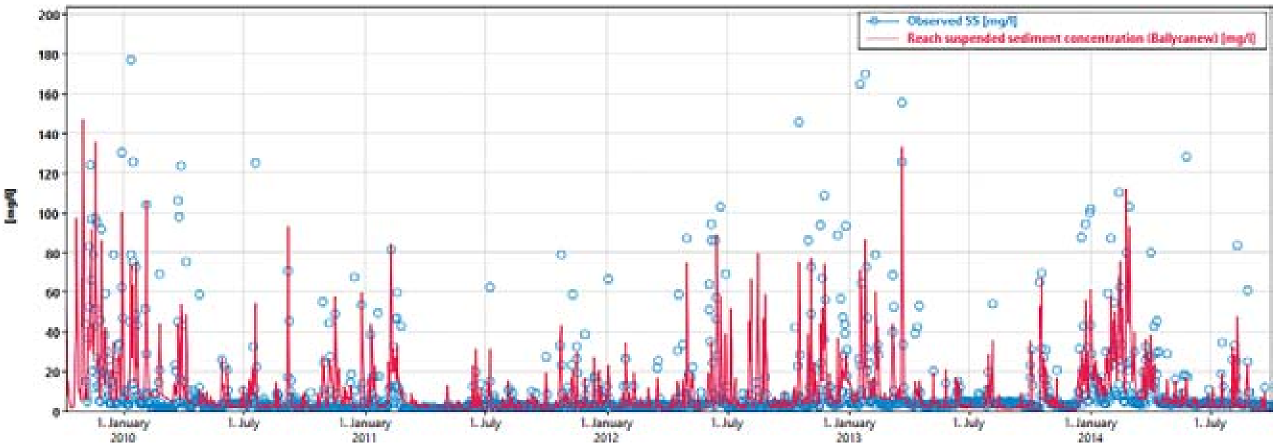

Figure 3.6. SimplyP SS concentration fit from automatic optimisation (Mobius) for Ballycanew.

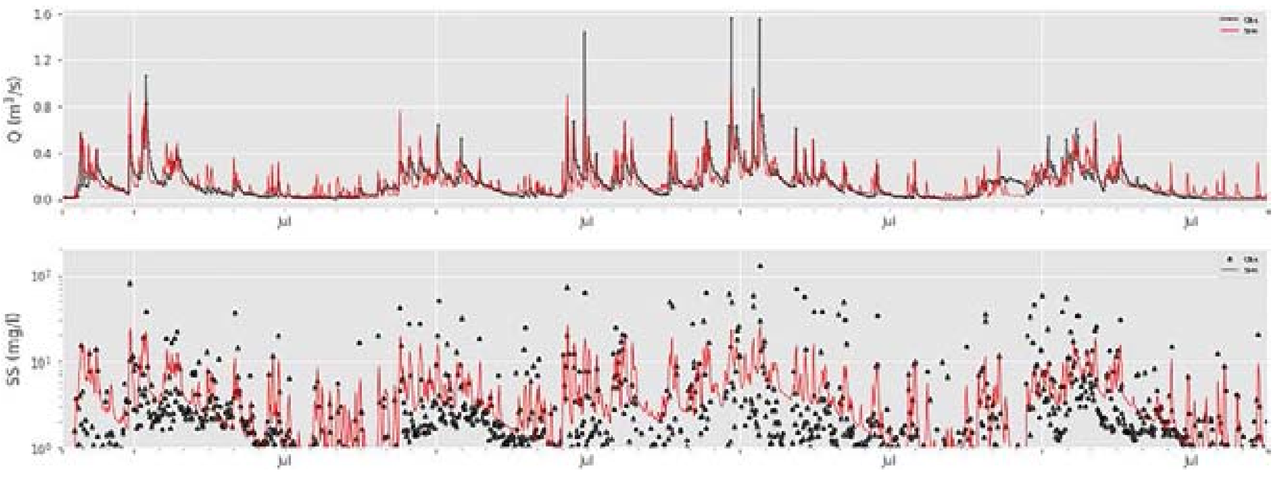

Figure 3.7. SimplyP flow (top) and SS concentration (bottom) for Timoleague. Uncalibrated red line is model output; grey line and black points are measured data.

automatic optimisation, the fit to flows improved, to a NS coefficient of 0.74 (Figure 3.8); however, in the case of SS concentrations (Figure 3.9), although the NS coefficient improved to 0.35, the fit was still not good.

The two test example catchments shown previously demonstrate that, while a simple conceptual model can achieve satisfactory fits to measured SS data in some catchments, it cannot be relied on for all catchments. Moreover, this type of model requires sediment and flow data with sufficient temporal resolution to resolve specific events for calibration to each catchment and such an approach, while usable for such catchments, is not suitable for use in all catchments, most of which do not have requisite measurements of sediment concentrations. The weakness of the SimplyP model, and of many other simple conceptual hydrological models, is a difficulty in correctly estimating peak flow magnitudes, and this carries through to the modelling of sediment (and other contaminants) mobilised by quickflow. Estimates of sediment load with sufficient temporal resolution, as well as corresponding flows and precipitation, are needed to improve event-based modelling and also to characterise sediment loads to support modelling for annual estimates.

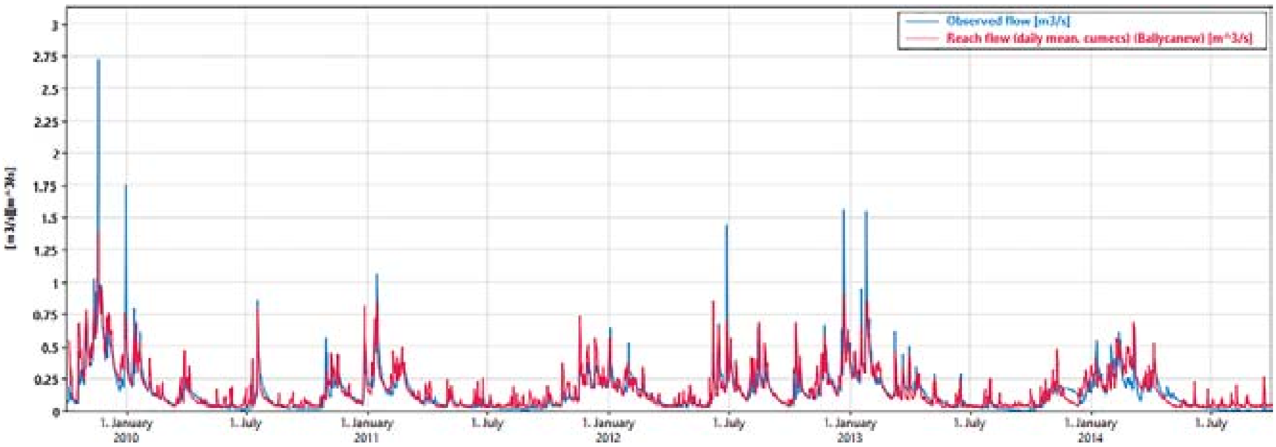

Figure 3.8. SimplyP discharge fit from automatic optimisation (Mobius) for Timoleague.

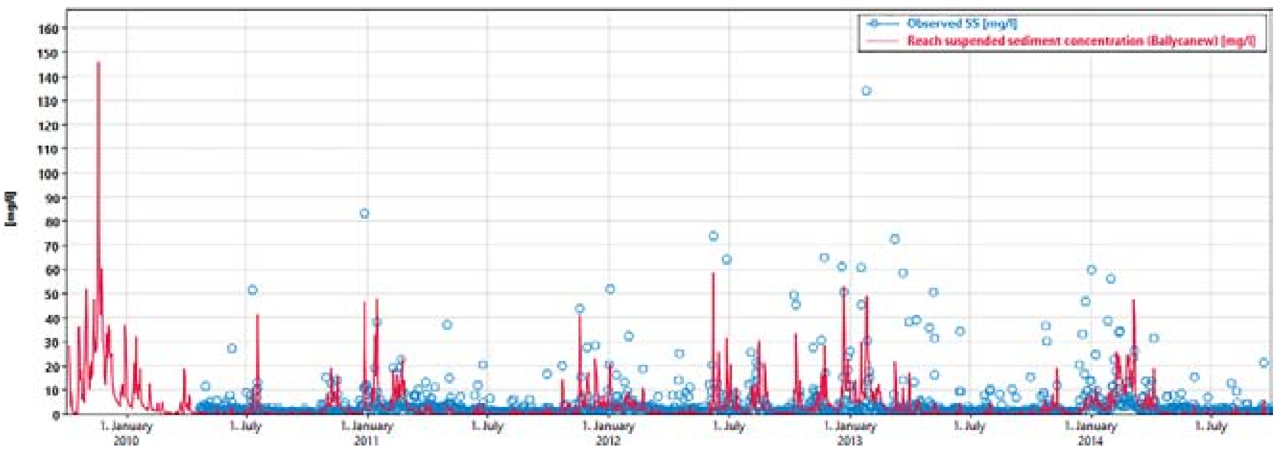

Figure 3.9. SimplyP SS concentration fit from automatic optimisation (Mobius) for Timoleague.

4 Pesticide Losses to Surface Waters

4.1 Pesticide Transport at Catchment Scales: Monitoring, Analysis and Modelling

In this chapter we address the issue of pesticides in rivers, with a particular focus on the pesticides used in Ireland. The separate technical report on pesticides (Bruen *et al.*, 2021) includes a literature review of the topic and a description of modelling approaches. Here, we analyse herbicide concentrations from three separate sources:

1. the ACP;
2. the EPA's own sampling and analyses;
3. data provided by the Animal and Plant Health Association (APHA).

All test for the most common herbicides in Ireland but also adopt very different approaches to either timing or spatial coverage. The purpose is to assess the factors involved in herbicide export to rivers to inform model development.

4.2 Herbicide Data Analysis and Modelling Implications

Here, measurements of herbicide concentrations are analysed to obtain information on the factors that can inform efforts to model pesticide export to rivers. Three major sources of data were available to the DiffuseTools project.

1. One year of 14-day average herbicide concentrations in two ACP catchments, collected as part of the EU Horizon 2020 WaterProtect project, together with local daily precipitation and flow measurements. Using a Chemcatcher instrument, this provides a picture of the trends in 14-day average concentrations over the year, which were compared with contemporaneous river flows and rainfall.
2. WFD monitoring: EPA data on the concentrations of 19 herbicides in rivers. The dataset analysed contained a total of 15,463 analytical results from 162 surface water sampling sites.
3. Monitoring of herbicide concentrations undertaken by the AHPA in priority areas, as defined by the National Pesticide and Drinking Water Action Group, namely Deel, Feale, Lough Forbes, Nore and Upper Erne.

Full details of the analyses of this data are given in the technical report on pesticides (Bruen *et al.*, 2021) and only a summary of key results and conclusions is given here. There is no environmental quality standard (EQS) in Ireland for pesticides, so here, as an illustration, we use the EU drinking water standard for most pesticides of 0.1 mg l^{-1} transposed into Irish law by S.I.122.[1]

4.2.1 Analysis of Agricultural Catchments Programme data

Introduction

As part of the EU Horizon 2020 project WaterProtect, a 1-year time series of 14-day average herbicide measurements was collected by Teagasc in the outlet of two ACP catchments, Ballycanew (dominated by grassland on poorly drained soils) and Castledockrell (dominated by arable land on well-drained soils) covering the period from 6 November 2018 to 21 November 2019. These catchments were not chosen for being "problem areas" for herbicides, but because they are hydrologically different from each other and have different land uses. The average annual runoff for that period was similar for both catchments: 719 mm for Ballycanew and 694 mm for Castledockrell. However, the annual rainfall totals were different (1180 mm for Ballycanew and 1316 mm for Castledockrell), indicating naturally different hydrological behaviour between the catchments. The herbicide data were fortnightly averages of

1 S.I. No. 122/2014 – European Union (Drinking Water) Regulations. Availabe online: http://www.irishstatutebook.ie/eli/2014/si/122/made/en/print (accessed 18 August 2021).

herbicide concentrations obtained via a Chemcatcher integrating sensor.

The mean values for all herbicides were well below the S.I. 122 limit (0.1 mg l^{-1}), but the maximum concentration measured was well above this limit in both catchments for fluroxypyr. In addition, the maxima for MCPA and triclopyr exceeded the limit in Ballycanew and for mecoprop in Castledockrell. In all cases (except one) the mean is substantially greater than the median owing to a positively skewed distribution of measured values with a small number of very high measurements, with maximum values typically over an order of magnitude greater than the means or medians. There could be many reasons for this, apart from soils and hydrology, including different application rates and/or practices, including accidental spills.

A key question for the DiffuseTools project is to what extent the delivery of herbicides to rivers is via the hydrological pathways (surface runoff, interflow and baseflow), as this is key for the modelling approach. To examine this question, the 14-day measured pesticide concentrations were plotted against the flows (mean and maximum), precipitation (mean and maximum), PET (mean and maximum) and estimates of the SMDs (mean and maximum) for the corresponding 14-day periods.

Relationship with catchment hydrology in Ballycanew

Looking at Ballycanew first, Figure 4.1 shows how the 14-day averages of MCPA measurements vary with the corresponding 14-day mean flows (Q_{mean}) and maximum daily precipitation (P_{max}) (the maximum precipitation was chosen as an indicator of storm events with possible quickflow runoff – the technical appendix gives the corresponding results for other herbicides). High average concentrations of all herbicides are associated with low mean flows and precipitation. There are no high 14-day average herbicide concentrations associated with high average river flows. However, plotting herbicide concentrations against contemporaneous PET (the third row of Figure 4.1) shows that high herbicide concentrations

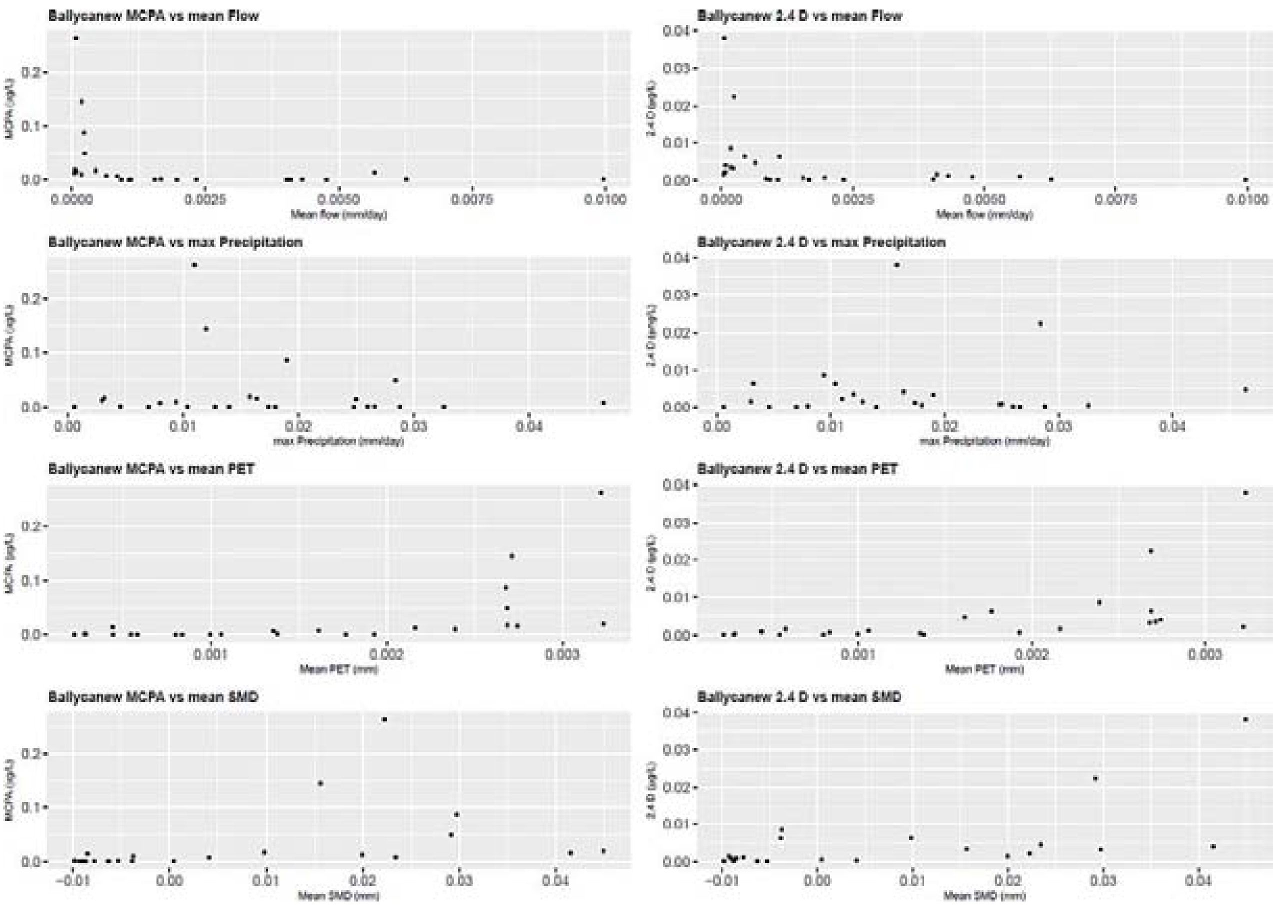

Figure 4.1. Ballycanew: relationship of MCPA (left) and 2,4-D (right) with hydrological quantities.

are often associated with high values of average PET. The relationship with the mean 14-day SMD is similar, with low herbicide concentrations associated with low SMDs. Although it appears that a small amount of herbicide is exported through subsurface flows, it is curious that there is no strong indication in the data of surface runoff due to high-intensity rain. Dilution in the resulting higher or intermediate flows occurs and may be a factor. It is also likely that herbicides are leached to groundwater (it was present in many private drinking water wells in the catchments) and transferred to the streams via shallow groundwater. In addition, there may be stores in sediments that are released under certain conditions. Note that Chemcatcher data are averages over 14 days and that the impact of very short-duration events is reduced by this averaging.

Relationship with catchment hydrology in Castledockrell

The same analysis was carried out for Castledockrell (Figure 4.2), with similar conclusions, although the relationships are not as strong as those observed for Ballycanew. Mecoprop and fluroxypyr show high 14-day average concentrations primarily associated with low average flows and lower peak rainfalls (i.e. pointing to some impact from other non-hydrological influences). However, high triclopyr measurements are more strongly associated with higher peak rainfalls (but not the highest) irrespective of the flow in the river, and high concentrations of triclopyr occur only infrequently during high river flows. The lack of a unique positive relationship between herbicide concentrations and hydrological fluxes shows that the high summer herbicide concentrations cannot be completely explained only by rain or high-flow events.

4.2.2 Analysis of EPA monitoring data

Introduction

An important question is whether or not the ACP results described previously, derived from the two intensively monitored ACP catchments, are representative of the herbicide situation in Ireland.

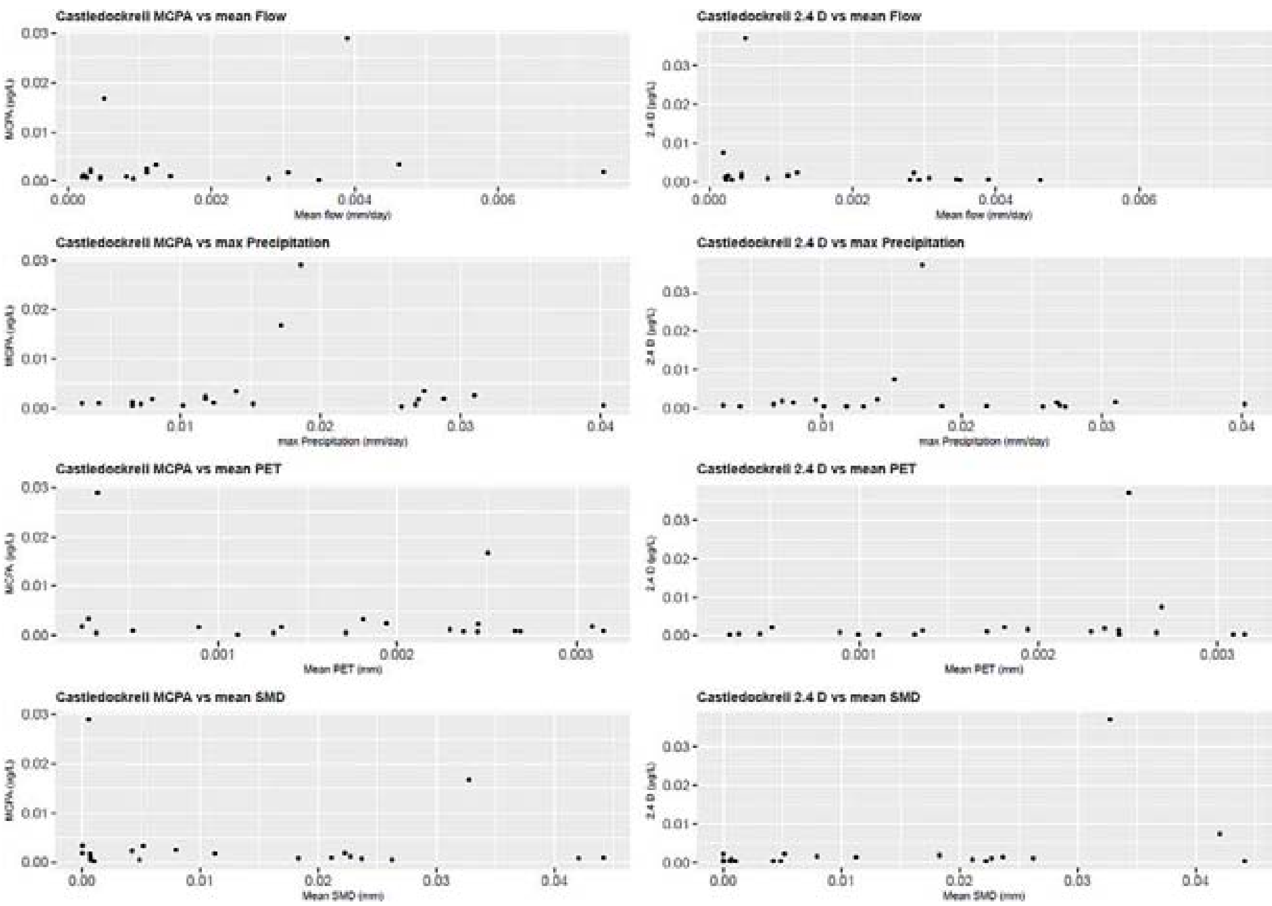

Figure 4.2. Castledockrell: relationship of MCPA (left) and 2,4-D (right) with hydrological quantities.

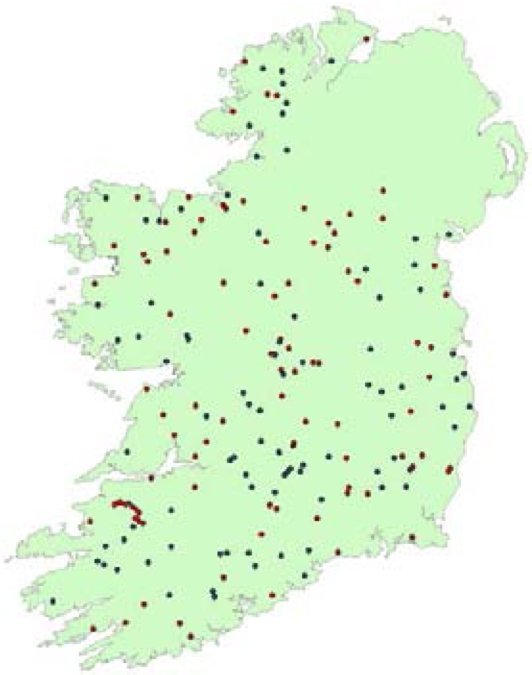

Figure 4.3. Location of EPA pesticide monitoring points. Red dots indicate sites where the S.I. 122 limit was exceeded by one or more sample and green dots indicate sites where no samples exceeded the S.I. 122 limit.

To answer this, a more spatially extensive dataset is required. The EPA monitors water quality at 9399 sites across Ireland for purposes that include their WFD-related responsibilities. Data for pesticides from the period 2013–2019 include samples (typically one per month) from 194 of these stations on rivers or streams, located across the country, with slightly better coverage in the Midlands than in the west and south of Ireland (Figure 4.3). These were analysed for one or more of 19 different pesticides (2,4-D, 2,6-dichlorobenzamide, AMPA, atrazine, clopyralid, dichlobenil, dichlorprop, diuron, glyphosate, isoproturon, linuron, malathion, MCPA, mecoprop, simazine, triclopyr, dieldrin, isodrin and terbutryn). Note that 2,6-dichlorobenzamide is a metabolite of dichlobenil. Fewer than 100 analyses were carried out for clopyralid, dichlorprop, triclopyr, dieldrin and isodrin, so these were not included in this analysis. Glyphosate and AMPA were also excluded. Less than 3% of the tests for the remaining 12 pesticides exceeded the S.I. 122 limit, except for MCPA, for which less than 8% exceeded that limit (Figure 4.4). There were no exceedances of the S.I. 122 limit for simazine and terbutryn. In all of these, the

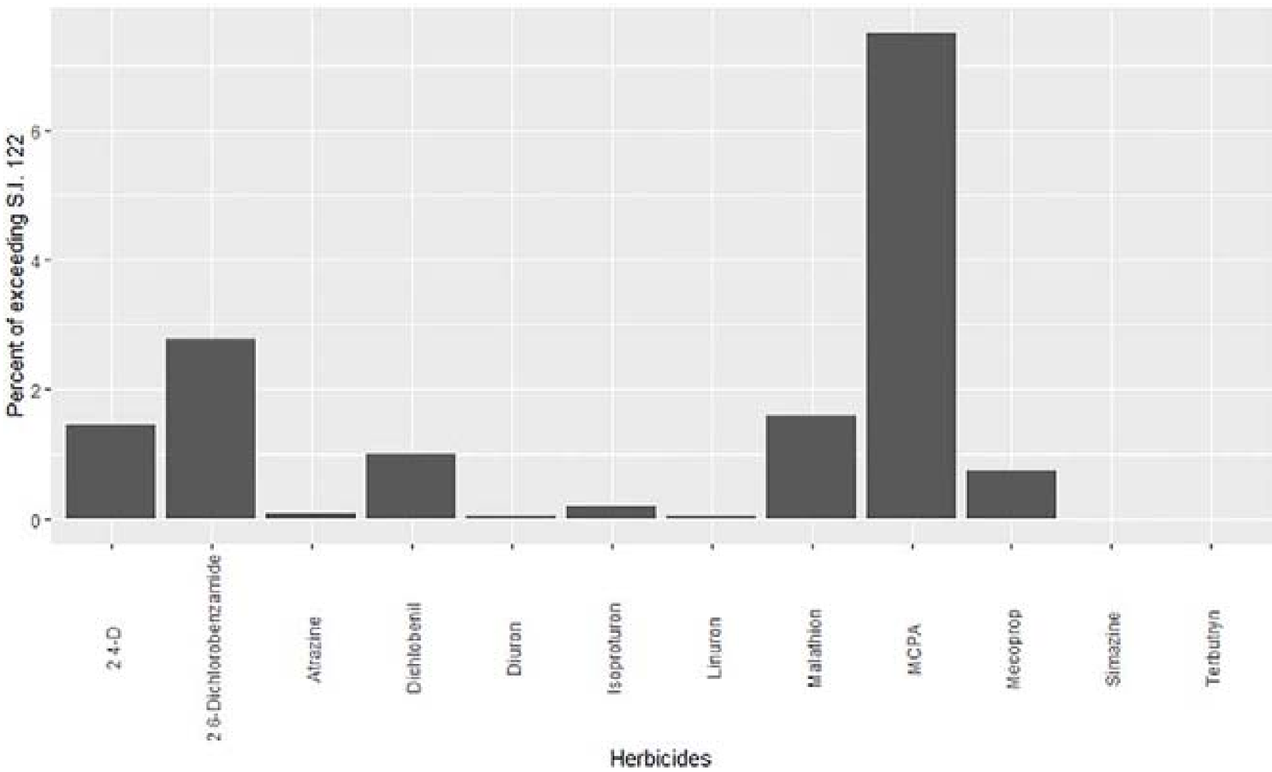

Figure 4.4. Percentage of EPA pesticide tests exceeding the S.I. 122 limit.

concentration median and 75% quartile are well below the limit, and the exceedances are typically an order of magnitude (or more) greater than the median, suggesting that they are caused by exceptional events. The locations of the 96 monitoring stations that have recorded one or more exceedances show particular clusters in the middle of the country, the north-west of Connaught, west Limerick, east Clare and in parts of the south-east of the country (Figure 4.3).

These are grab samples taken at specific times and include some local authority data. DiffuseTools analysed these data covering the years 2013–2019. For mecoprop, isodrin, simazine, triclopyr and dichloroprop there were a negligable number of values above the limit of reporting, so these chemicals were not considered here.

The percentages of analyses exceeding the S.I. 122 limit for each herbicide are shown in Figure 4.4. The most prominent chemicals are dichlobenil, MCPA, clopyralid, mecoprop and diuron. The percentage of analyses each month that exceeded the S.I.122 limit is shown in Figure 4.5. Despite sampling throughout the year, there is a clear tendency for most exceedances of the regulation to occur in the summer/autumn period, particularly for MCPA and mecoprop. The percentage of samples exceeding the limit is smaller for 2,4-D and 2,6-dichlorobenzamide than mecoprop and MCPA and they are more evenly spread over the year. Although only a small percentage of analyses exceed the S.I. 122 limit for malathion, a small late-summer peak is observed.

For some herbicides, the percentage of analyses below the S.I. 122 limit is larger in the EPA dataset than in the ACP data. One possible reason is that the continuous 14-day average produced by the Chemcatcher picks up short-duration pulses of herbicide that infrequent grab sampling may miss.

Relationship of herbicide concentrations with hydrological processes

Unfortunately, river flow and catchment rainfall data are not readily available for the herbicide monitoring stations. Therefore, an indirect approach is used to investigate the strength of the relationship between herbicide concentration and hydrological processes. It is generally accepted that sediment concentrations, and to some extent P concentrations, in rivers are related to heavy rain events and high flows and are associated with the quick-flow response of catchments. In contrast, nitrate concentrations are generally associated with less variable subsurface pathways. Nitrate pulses can also follow drought periods because the moisture stress means less root uptake and more leaching when the drought ends. If the processes resulting in high herbicide concentrations are similar to those resulting in high SS and P, this would support an argument that the

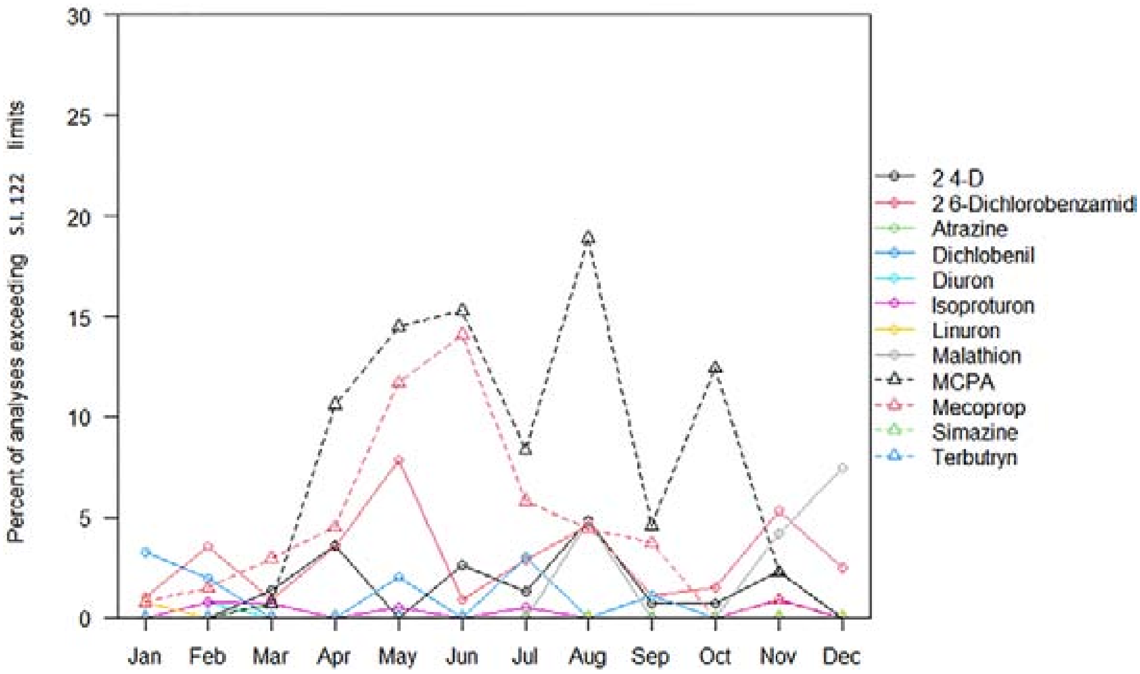

Figure 4.5. Monthly distribution of analyses exceeding S.I. 122 by month in EPA data (2013–2019).

Table 4.1. Summary of comparisons of herbicides with other water quality parameters

Case number	Herbicide	Water quality parameter	Number of monitoring stations	Interpretation
1	MCPA	SS	7	Considerable scatter but indications of inverse relationship, i.e. that high MCPA concentrations are associated with low SS and vice versa. See Figure 4.6
2	MCPA	OrthoP	34	More scatter than case 1, some indications of inverse relationship, but not in all cases
3	MCPA	TON	34	More scatter than case 1, some indications of inverse relationship, but not in all cases
4	2,4-D	SS	6	Considerable scatter. Some of the graphs are dominated by single outliers, so no conclusion possible
5	2,4-D	OrthoP	32	Considerable scatter, but general indication of inverse relationship, but not in all cases
6	2,4-D	TON	32	Some scatter but generally high TON associated with low herbicide concentrations

OrthoP, orthophosphate; TON, total oxidised nitrogen.

processes of mobilisation and transport were also similar. Alternatively, if the processes were similar to those resulting in high nitrate concentrations then the mobilisation and transport process would also be similar. To examine this, measurements of SS, P and nitrate for the same days and sites as herbicide measurements were extracted from the EPA dataset and correlated with the individual herbicide concentrations. This was carried out for all of the individual monitoring locations for which there are herbicide data. The results for MCPA and 2,4-D are summarised in Table 4.1 and examples are shown in the plots of Figure 4.6. There is considerable scatter in the data, understandable because of the many factors

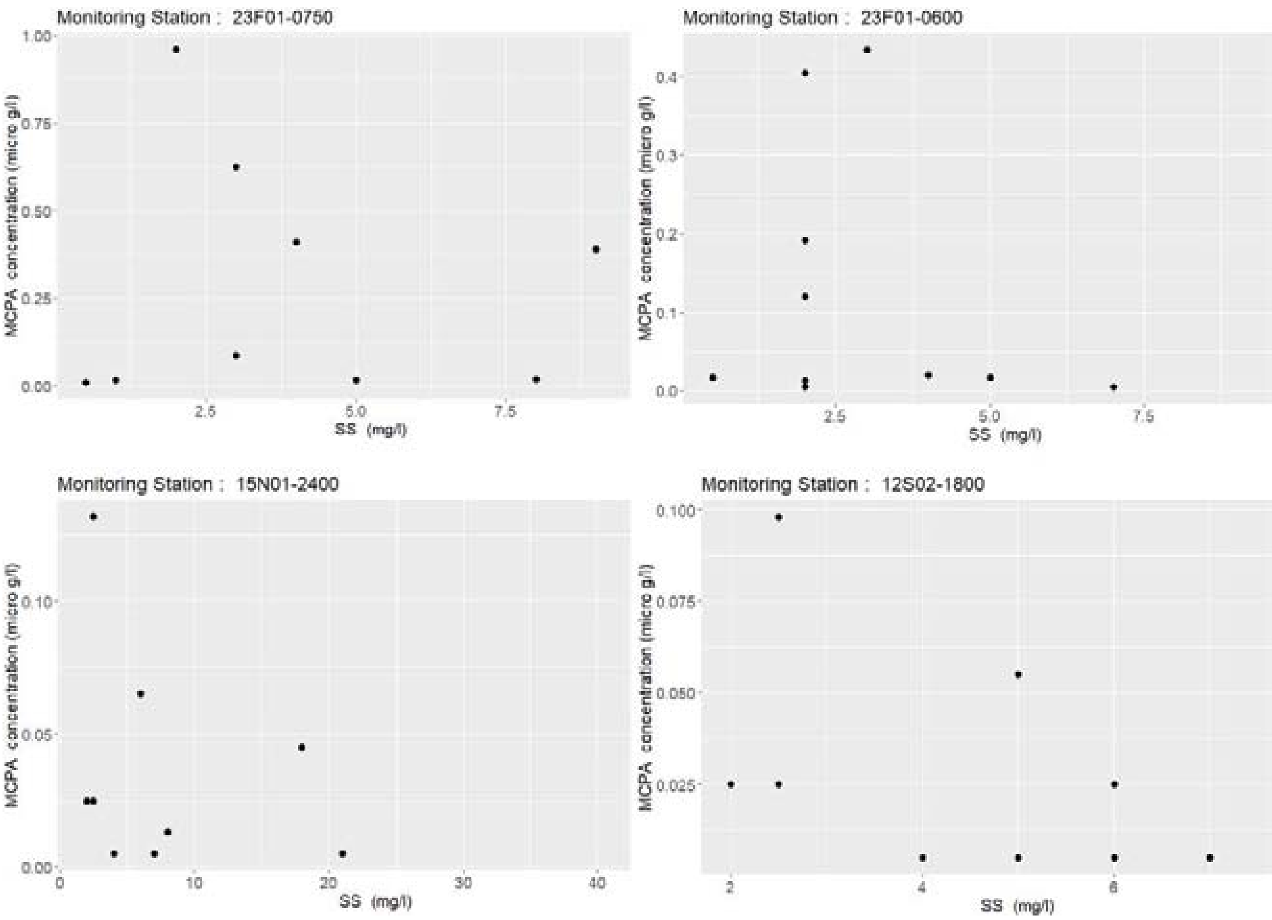

Figure 4.6. Examples of MCPA vs SS (EPA data 2013–2019).

that influence the concentrations of herbicide and other water quality parameters, and many graphs were not used if dominated by a single outlier. Nevertheless, there is an overall impression of an inverse relationship between high herbicide concentration and SS, orthophosphate and total oxidised nitrogen. This suggests that there are some differences in the generating processes producing high concentrations of each. In particular, the association of some high herbicide concentrations with dry weather and low flows from the analysis of the two ACP catchments seems to apply more widely.

4.2.3 Analysis of APHA data

The APHA is the representative body for manufacturers and sole distributors of veterinary medicines and agrochemicals, including pesticides and herbicides. Part of its remit is to provide information about these products and to inform policy and legislation in relation to their effects on the environment. The APHA has undertaken weekly monitoring of pesticides in waterbodies in five areas: Lough Forbes, together with the upper reaches of the Deel, Feale and Nore, were monitored mainly from week 14 to week 42 of 2019 and 2020, while sites near the upper Erne system near Belturbet were monitored only in 2020. The locations of these sampling points are shown in Figure 4.7. All samples were analysed for MCPA, MCPB, 2,4-D, 2,4-DB, mecoprop and dichlorprop 2,4-DP and the four with the highest percentage exceedences are summarised in Table 4.2.

Sampling for all sites typically took place from week 14 (mid-April) until about week 42 (end of September), corresponding to the period expected to be of highest risk. In the early part of this period, sampling was weekly, but later fortnightly. Figure 4.8 shows a box and whisker plot of the results by week. The overall median herbicide concentrations (shown as horizontal black lines), combining all analyses together for each week, were below the S.I. 122 limit in all weeks. However, there are many very high-concentration outliers (shown as red dots) in each week sampled, suggesting the episodic nature of the higher concentrations. The early weeks, mid-April to the end of May, tend to have higher exceedances than the later parts of the year. The Feale catchment has the highest median herbicide concentration, above the regulatory limit (Figure 4.9). All the other

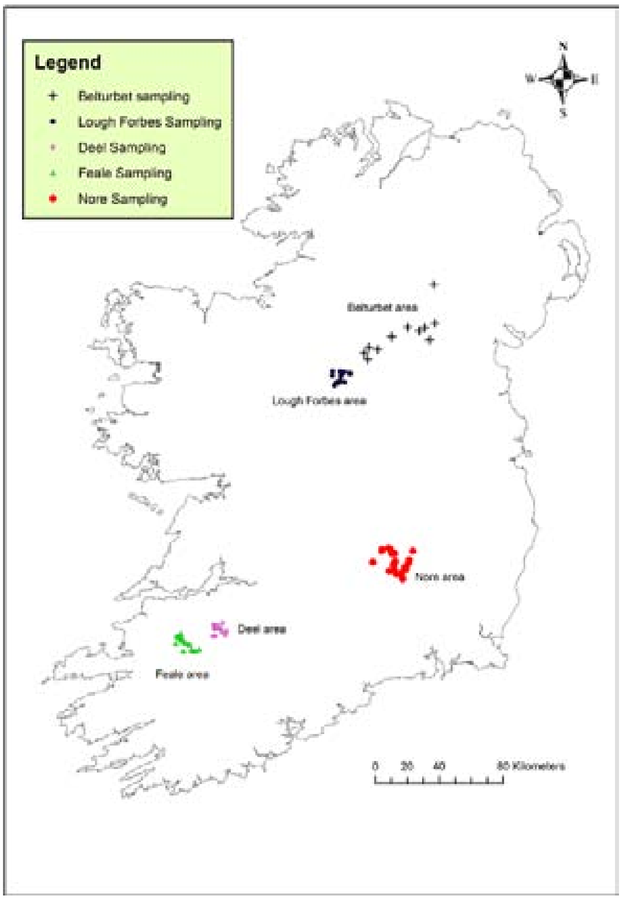

Figure 4.7. Locations of APHA sampling points.

catchments have medians at or below the limit, and Belturbet has the lowest median concentration. All catchments have a significant number of high outliers, although all medians are below the S.I. 122 threshold.

The median of the APHA data for each of the areas is below the S.I. 122 limit when all samples, including those below the limit of reporting, are included (values set to half of the limit). However, some samples

Table 4.2. Summary of APHA analyses for all five study areas (2019–2020)[a]

Herbicide	Analyses exceeding limit (%)
MCPA	15
2,4-D	2
Mecoprop	0.6
2,4-DB	0.2
Total	4.4

[a]Includes values below level of reporting.

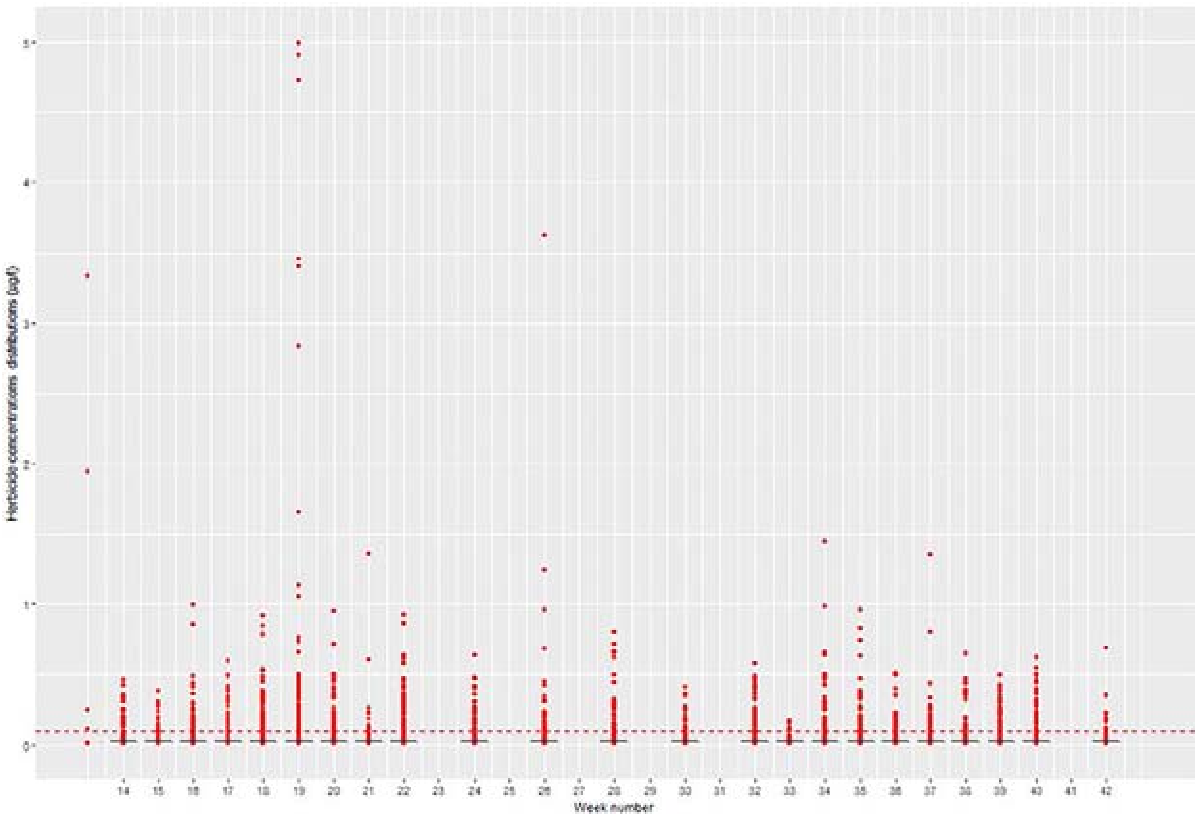

Figure 4.8. APHA concentration data by week (2018–2020). The dashed red line denotes the S.I. 122 limit, and the horizontal black lines are the median values.

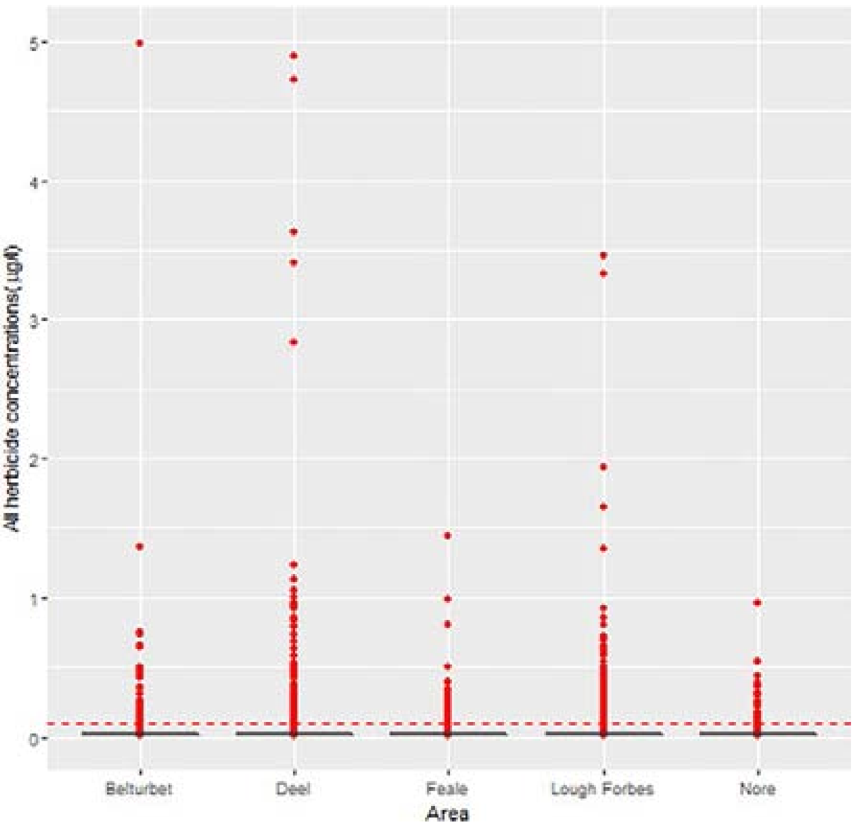

Figure 4.9. APHA data distributed by catchment (2018–2020). The dashed red line denotes the S.I. 122 limit, and the horizontal black lines are the median values.

from all areas have high herbicide concentrations, exceeding 1 mg l^{-1}, and some samples from three areas (Belturbet, Deel and Lough Forbes) exceeded 3 mg l^{-1}. However, there is considerable variation between the individual sampling points in each area (details can be found in the technical report).

The timing of high MCPA concentrations in samples tend to be similar, but not identical, for many stations in the region. This suggests similar timing of the drivers of these concentrations, whether it be the hydrological drivers of mobilisation and transport, management decisions or weather conditions.

5 Conclusions

New national maps of diffuse agricultural soil P losses showing estimated surface runoff and P mobilisation for four different soil Morgan P index scenarios were developed for Ireland. Accumulation of surface runoff and pollutants downslope to waterbodies were also modelled, allowing delivery zones and breakthrough and delivery points along surface pathways to be identified across the country. These can be overlaid with new PIPv3 maps in development by the EPA to allow catchment managers and policymakers to target mitigation measures more cost-effectively and to quantify potential changes to water quality from proposed measures. The outputs from this study can be used to underpin actions to improve water quality, functional land management and agricultural sustainability.

In relation to sediments, the project applied two approaches appropriate to different scales. At high-resolution hillslope scale, the project applied the SCIMAP methodology, as used in the UK, and produced maps of surface connectivity and erosion potential and risk with national coverage. At the catchment scale, the project showed that a simple conceptual model (SimplyP), with most parameters determined from known or mapped features of the catchment, could predict SS export from one of the two well-monitored agricultural catchments typical of tillage and grassland, but that prediction capability in the second catchment was limited. Model performance improved with calibration of the parameters; this is possible only for gauged and monitored catchments, so is not yet applicable nationally.

The project also analysed pesticide monitoring data from the ACP catchments, the EPA and APHA, and demonstrated that measured pesticide concentrations were usually within the S.I. 122 drinking water limit. However, extreme exceedances of this limit (sometimes by an order of magnitude), predominantly in the March–October period, which includes the main application period, are highly episodic in nature and not always linked with high rainfall or flows. It is likely that, in addition to hydrological and catchment-specific factors, there are other significant influences involved in many of these episodes, perhaps related to preparation and application (timing and method) of the pesticides that typically are not captured in traditional models.

For the ACP catchments higher 14-day average concentrations are coincident with low average flows and low rainfall amounts but do correspond to periods of high PET (triclopyr in the Castledockrell ACP catchment is the exception). This could be because (1) herbicides are preferentially applied during periods of good weather, and high concentrations are due to accidents when mixing or during application or to spray drift; or (2) there is less dilution of herbicide in base flows.

The Irish data show that, in many cases, occurrences of very high herbicide concentrations in rivers are intermittent, and are not always exclusively associated with high water fluxes. Rather, the temporal link between very high herbicide concentrations and periods of high PET seems to be due to the choice of good weather periods for application of the herbicides and the fact that application factors sometimes contribute substantially to the high-concentration episodes. This has implications for the modelling of herbicide concentrations in rivers, and suggests that a combination of a stochastic model for applications and related accidents linked together with a process-based hydrological model for transport, dilution and attenuation is indicated. The available data are not sufficient for the validation of such a model. In particular, the paucity of information on the timing of herbicide application by farmers, and the amounts applied, means that a stochastic loadings model cannot currently be calibrated. Future monitoring programmes should include data for estimating flows so that loads can be calculated.

6 Recommendations and Further Research

This project has identified a number of technical and policy recommendations and further research.

1. Riparian measures to mitigate P and sediment delivery to waterbodies should prioritise the major delivery and breakthrough locations identified in this research, with local solutions appropriate to the magnitude of the delivery. Spatiotemporal modelling of HSAs, CSAs and surface runoff volumes/P loads at daily, monthly or annual time steps (rather than 30-year average) can now be undertaken using the HSMDv3 outputs, as well as dynamic forecasting with real-time weather data (see Drohan *et al*., 2019) and predicting the effects of climate change.
2. Research is required to quantify the fraction of accumulated surface runoff that interacts with the soil and mobilises soil P.
3. Edge-of-field measurements of surface runoff and P concentrations are needed, targeted at modelled surface runoff pathways, delivery points and non-HSAs, to provide validation data on the mobilisation and deposition of P, free from the effects of point sources.
4. National field-scale soil P data are needed to better predict diffuse P losses from all pathways and impacts on water quality. Teagasc has a national database of soil samples from farms, and this could be a valuable resource for the research community. The national Tellus survey programme from GSI has soil samples on a regular fine-scale grid across half of the country, with the remainder to be completed in the next few years.
5. National cost–benefit analysis of targeting measures at breakthrough and delivery points, and identifying and prioritising those with the largest surface runoff P loads, is needed for informing stakeholders and policy decisions. The SMARTER_BufferZ and WaterMARKE projects have started to address this topic.
6. Further research into the hydrological connectivity of flow pathways is needed, specifically the slow-down, impediment and re-infiltration of overland flow on shallow slopes in heavily vegetated areas or at flow diversions. Research on natural water retention measures can start to address this need, for example the SlowWaters research project.
7. Future work is needed to calculate flow sink volume capacities and quantify whether or not they hydrologically disconnect daily surface runoff volumes entering the sink (following on from Thomas *et al*., 2016a), as well as “fill and spill” dynamics.
8. Research into actual soil P concentrations, their mobilisation and deposition within delivery paths is needed to evaluate the approach described in the points above.
9. Research is needed into surface runoff from Irish peatlands and the intensity of agriculture in those areas to better inform P loss modelling.
10. Whole-farm P balances could be calculated using environmental losses predicted in this study in combination with Teagasc National Farm Survey data (e.g. following on from Murphy *et al*., 2019).
11. Further model development should include the accounting of infiltration–excess overland flow, soil compaction hotspots, subsurface return flow and subsurface drains, and particularly with the last, the development of such datasets nationally.
12. Model development should also include estimating P loads in subsurface and groundwater pathways utilising the new subsurface flow map (Figure 2.17) and GSI groundwater recharge map.
13. A national LiDAR DEM would be useful to capture local microtopographic influences on surface runoff pathways. However, although 1- to 2-m grid resolutions are deemed optimal (Thomas *et al*., 2017), it increases visual noise and “thins” pathways considerably, making it less user-friendly. Furthermore, data processing requirements would increase considerably. Although flow pathways can be diverted by microtopography, hillslope-scale topographic controls tend to reroute flow back to similar flow pathways and directions as those predicted

by the 5-m DEM. Thus, for national modelling, 5-m-resolution DEMs may be a good compromise between accuracy, processing requirements and user-friendly products.

14. Model uncertainty (data, model structure and parameter values) should be evaluated and used to spatially map uncertainties and to identify areas requiring special investigation.

15. Modelling of sediment concentrations and loads is limited. Estimates of sediment load from measurements of concentration and flow with sufficient temporal resolution, as well as corresponding precipitation, are needed to (1) improve the event-based modelling and (2) characterise sediment loads to support modelling for annual estimates.

16. Strategies are needed to collect the full range of data needed to improve understanding and modelling of the episodic elevated herbicide concentrations in some Irish waterbodies. This will allow the further development and calibration of the new modelling approaches needed. This should include more factors than hydrological and chemical processes as it also requires a characterisation of the incidents that cause the high peak concentrations and possibly of decision-making and practice in relation to the application of pesticides.

17. Models that represent the subsurface flow components of contaminants, including pesticides, are required, particularly in areas where transition zone flow or karst flow pathways are important.

18. Although at most monitoring points the distributions of measured herbicide concentrations are heavily positively skewed, with mostly low values, there are some intermittent extremely high values. This necessitates a frequent monitoring regime to capture the full variability and to estimate loads, and to support understanding and modelling of both the episodic and baseline concentrations and loads.

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Abbreviations

ACP	Agricultural Catchments Programme
APHA	Animal and Plant Health Association
CSA	Critical source area
DEM	Digital elevation model
DST	Decision support tool
EPA	Environmental Protection Agency
GIS	Geographical information system
GLAS	Green Low-carbon Agri-environment Scheme
GSI	Geological Survey Ireland
HSA	Hydrologically sensitive area
HSMD	Hybrid soil moisture deficit
ICHEC	Irish Centre for High-End Computing
LAWPRO	Local Authorities Waters Programme
LPIS	Land Parcel Identification System
MÉRA	Met Éireann Reanalysis
mSMD	Negative or minus soil moisture deficit
NAP	Nitrates Action Programme
NS	Nash–Sutcliffe
NWP	Numeric weather prediction
OPW	Office of Public Works
OSI	Ordnance Survey Ireland
OSPAR	Oslo Paris Convention for the Protection of the Marine Environment of the North-East Atlantic
PAA	Priority Areas for Action
PET	Potential evapotranspiration
PIP	Pollutant Impact Potential
RBD	River basin district
RBMP	River Basin Management Plan
RUSLE	Revised universal soil loss equation
SCIMAP	Sensitive Catchment Integrated Modelling Analysis Platform
SLAM	Source Load Apportionment Model
SMD	Soil moisture deficit
SS	Suspended sediment
STI	Soil topographic index
TP	Total phosphorus
TRP	Total reactive phosphorus
TWI	Topographic wetness index
UDA	Upslope drainage area
WFD	Water Framework Directive

AN GHNÍOMHAIREACHT UM CHAOMHNÚ COMHSHAOIL
Tá an Ghníomhaireacht um Chaomhnú Comhshaoil (GCC) freagrach as an gcomhshaol a chaomhnú agus a fheabhsú mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ó éifeachtaí díobhálacha na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialú: *Déanaimid córais éifeachtacha rialaithe agus comhlíonta comhshaoil a chur i bhfeidhm chun torthaí maithe comhshaoil a sholáthar agus chun díriú orthu siúd nach gcloíonn leis na córais sin.*

Eolas: *Soláthraímid sonraí, faisnéis agus measúnú comhshaoil atá ar ardchaighdeán, spriocdhírithe agus tráthúil chun bonn eolais a chur faoin gcinnteoireacht ar gach leibhéal.*

Tacaíocht: *Bímid ag saothrú i gcomhar le grúpaí eile chun tacú le comhshaol atá glan, táirgiúil agus cosanta go maith, agus le hiompar a chuirfidh le comhshaol inbhuanaithe.*

Ár bhFreagrachtaí

Ceadúnú
Déanaimid na gníomhaíochtaí seo a leanas a rialú ionas nach ndéanann siad dochar do shláinte an phobail ná don chomhshaol:
- saoráidí dramhaíola *(m.sh. láithreáin líonta talún, loisceoirí, stáisiúin aistrithe dramhaíola)*;
- gníomhaíochtaí tionsclaíocha ar scála mór *(m.sh. déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta)*;
- an diantalmhaíocht *(m.sh. muca, éanlaith)*;
- úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe *(OGM)*;
- foinsí radaíochta ianúcháin *(m.sh. trealamh x-gha agus radaiteiripe, foinsí tionsclaíocha)*;
- áiseanna móra stórála peitril;
- scardadh dramhuisce;
- gníomhaíochtaí dumpála ar farraige.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil
- Clár náisiúnta iniúchtaí agus cigireachtaí a dhéanamh gach bliain ar shaoráidí a bhfuil ceadúnas ón nGníomhaireacht acu.
- Maoirseacht a dhéanamh ar fhreagrachtaí cosanta comhshaoil na n-údarás áitiúil.
- Caighdeán an uisce óil, arna sholáthar ag soláthraithe uisce phoiblí, a mhaoirsiú.
- Obair le húdaráis áitiúla agus le gníomhaireachtaí eile chun dul i ngleic le coireanna comhshaoil trí chomhordú a dhéanamh ar líonra forfheidhmiúcháin náisiúnta, trí dhíriú ar chiontóirí, agus trí mhaoirsiú a dhéanamh ar leasúchán.
- Cur i bhfeidhm rialachán ar nós na Rialachán um Dhramhthrealamh Leictreach agus Leictreonach (DTLL), um Shrian ar Shubstaintí Guaiseacha agus na Rialachán um rialú ar shubstaintí a ídíonn an ciseal ózóin.
- An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

Bainistíocht Uisce
- Monatóireacht agus tuairisciú a dhéanamh ar cháilíocht aibhneacha, lochanna, uiscí idirchriosacha agus cósta na hÉireann, agus screamhuiscí; leibhéil uisce agus sruthanna aibhneacha a thomhas.
- Comhordú náisiúnta agus maoirsiú a dhéanamh ar an gCreat-Treoir Uisce.
- Monatóireacht agus tuairisciú a dhéanamh ar Cháilíocht an Uisce Snámha.

Monatóireacht, Anailís agus Tuairisciú ar an gComhshaol
- Monatóireacht a dhéanamh ar cháilíocht an aeir agus Treoir an AE maidir le hAer Glan don Eoraip (CAFÉ) a chur chun feidhme.
- Tuairisciú neamhspleách le cabhrú le cinnteoireacht an rialtais náisiúnta agus na n-údarás áitiúil *(m.sh. tuairisciú tréimhsiúil ar staid Chomhshaol na hÉireann agus Tuarascálacha ar Tháscairí)*.

Rialú Astaíochtaí na nGás Ceaptha Teasa in Éirinn
- Fardail agus réamh-mheastacháin na hÉireann maidir le gáis cheaptha teasa a ullmhú.
- An Treoir maidir le Trádáil Astaíochtaí a chur chun feidhme i gcomhair breis agus 100 de na táirgeoirí dé-ocsaíde carbóin is mó in Éirinn.

Taighde agus Forbairt Comhshaoil
- Taighde comhshaoil a chistiú chun brúnna a shainaithint, bonn eolais a chur faoi bheartais, agus réitigh a sholáthar i réimsí na haeráide, an uisce agus na hinbhuanaitheachta.

Measúnacht Straitéiseach Timpeallachta
- Measúnacht a dhéanamh ar thionchar pleananna agus clár beartaithe ar an gcomhshaol in Éirinn *(m.sh. mórphleananna forbartha)*.

Cosaint Raideolaíoch
- Monatóireacht a dhéanamh ar leibhéil radaíochta, measúnacht a dhéanamh ar nochtadh mhuintir na hÉireann don radaíocht ianúcháin.
- Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as taismí núicléacha.
- Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta.
- Sainseirbhísí cosanta ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Faisnéis Inrochtana agus Oideachas
- Comhairle agus treoir a chur ar fáil d'earnáil na tionsclaíochta agus don phobal maidir le hábhair a bhaineann le caomhnú an chomhshaoil agus leis an gcosaint raideolaíoch.
- Faisnéis thráthúil ar an gcomhshaol ar a bhfuil fáil éasca a chur ar fáil chun rannpháirtíocht an phobail a spreagadh sa chinnteoireacht i ndáil leis an gcomhshaol *(m.sh. Timpeall an Tí, léarscáileanna radóin)*.
- Comhairle a chur ar fáil don Rialtas maidir le hábhair a bhaineann leis an tsábháilteacht raideolaíoch agus le cúrsaí práinnfhreagartha.
- Plean Náisiúnta Bainistíochta Dramhaíola Guaisí a fhorbairt chun dramhaíl ghuaiseach a chosc agus a bhainistiú.

Múscailt Feasachta agus Athrú Iompraíochta
- Feasacht chomhshaoil níos fearr a ghiniúint agus dul i bhfeidhm ar athrú iompraíochta dearfach trí thacú le gnóthais, le pobail agus le teaghlaigh a bheith níos éifeachtúla ar acmhainní.
- Tástáil le haghaidh radóin a chur chun cinn i dtithe agus in ionaid oibre, agus gníomhartha leasúcháin a spreagadh nuair is gá.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil
Tá an ghníomhaíocht á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóirí. Déantar an obair ar fud cúig cinn d'Oifigí:
- An Oifig um Inmharthanacht Comhshaoil
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Fianaise is Measúnú
- Oifig um Chosaint Radaíochta agus Monatóireachta Comhshaoil
- An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag comhaltaí air agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

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Catchment Models and Management Tools for Diffuse Contaminants (Sediment, Phosphorus and Pesticides): DiffuseTools Project

Authors: Ian Thomas, Michael Bruen, Eva Mockler, Christopher Werner, Per-Erik Mellander, Sim Reaney, Anna Rymszewicz, Gavan McGrath, Edith Eder, Andrew Wade, Adrian Collins and Berit Arheimer

Identifying Pressures

Eutrophication, often driven by phosphorus, is the most significant issue for inland surface waters in Ireland. 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 a major source of phosphorus pollution in Irish rivers; however, because of its combined diffuse and point source characteristics, it is also often the most difficult source to quantify and manage. Furthermore, although some sediment is a natural component of healthy rivers, too much can also have an impact on their morphological and biological status. Pesticides are a vital part of agricultural systems, but they also pose a threat to both human and animal health, and to water quality. The DiffuseTools project has addressed the characterisation and modelling of all three of these major pressures (i.e. phosphorus, sediment and pesticides) on Irish rivers.

Informing Policy

This research contributes to Ireland's response to the third River Basin Management Plan of the WFD and to the development and implementation of more sustainable and cost-effective agricultural policies. Identification of critical source areas and pathways of phosphorus export at high resolution (subfield scale) from agricultural areas will assist (1) Local Authority Waters Programme (LAWPRO) catchment managers in identifying the locations and scales of appropriate interception measures needed for diffuse pollutants in overland flow and (2) The Agricultural Sustainability Support and Advisory Programme (ASSAP) advisers in implementing Good Agricultural Practice regulations and achieving Food Wise 2025 goals. These outputs underpin actions to improve water quality, functional land management and agricultural sustainability.

Developing Solutions

New national high-resolution (5 m scale) maps of diffuse agricultural soil phosphorus losses in surface runoff pathways were developed for Ireland. Critical source areas, breakthrough points at field boundaries and delivery points to waterbodies were identified across the country. These can be overlaid with the new EPA Pollutant Impact Potential (PIP) v3 maps to allow catchment managers and policymakers to target mitigation measures more cost-effectively and quantify potential changes to water quality from proposed measures.

The project produced SCIMAP (Sensitive Catchment Integrated Modelling Analysis Platform) sediment risk maps for all river basin districts. It also showed that a simpler version of the INCA-P model (SimplyP) provides useful dynamic sediment export estimates, but only for some catchments and so is not yet applicable nationally.

The project demonstrated that measured pesticide concentrations in Irish rivers were usually within the S.I. 122 drinking water limit. However, extreme exceedances, sometimes by an order of magnitude and occurring mostly in the main pesticide application period, are probably linked to preparation and application practices in addition to hydrological and catchment factors. Thus, management of all these factors is required.

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