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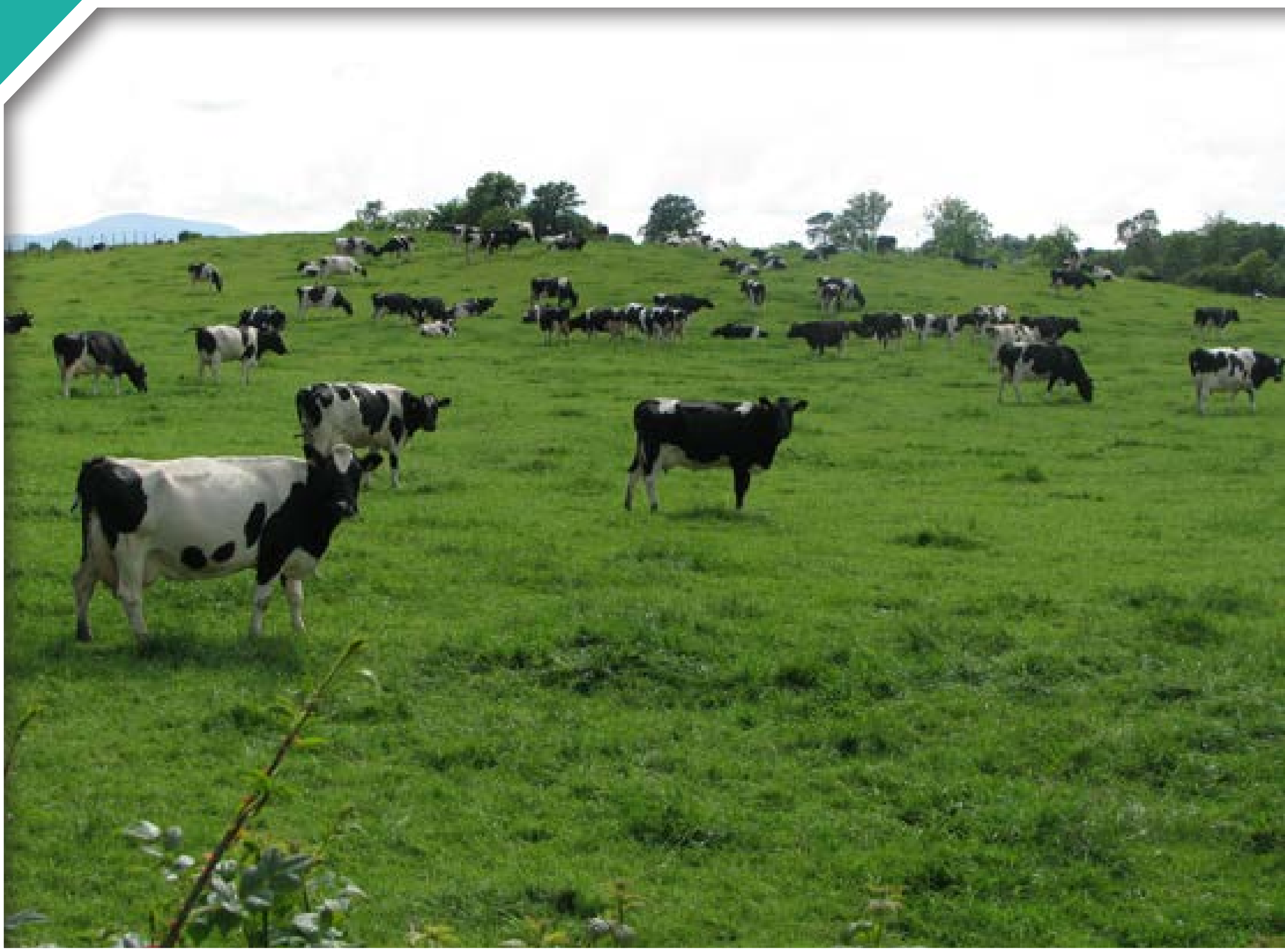
AgImpact Project: A Systematic and Participatory Review of Research on the Impact of Agriculture on Aquatic Ecosystems in Ireland

Authors: Donnacha Doody, Paul Cross, Paul Withers, Rachel Cassidy, Cara Augustenborg, Andrew Pullin, Owen Carton and Seamus Crosse



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Environment, Community and Local Government

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EPA Research Programme 2014–2020

AgImpact Project: A Systematic and Participatory Review of Research on the Impact of Agriculture on Aquatic Ecosystems in Ireland

EPA Research Report

(2013-W-DS-13)

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by

The Agri-Food and Biosciences Institute

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The EPA Research Programme addresses the need for research in Ireland to inform policymakers and 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

The increasing world population and changing consumption patterns, as well as changes in the policy environment (e.g. the removal of the European Union milk quota in 2015) and the ambition of the agri-food industry to expand, will drive an increase in outputs from Irish agriculture in the future. With the spatial expansion of agriculture in Ireland limited by the availability of suitable land, increases in agricultural outputs will be driven by the intensification of farming in existing areas through improvements in technology and resource efficiency. This will have to be achieved within the context of the targets established by EU environmental directives, such as the Nitrates Directive (ND; 91/676/EEC) and the Water Framework Directive (WFD; 2000/60/EC). However, as Ireland seeks to implement the schedule of the WFD, there is significant uncertainty in relation to achieving these targets within the context of the ambitious growth targets for agriculture, as defined in the Food Wise 2025 and Going for Growth strategies. How and if sustainable intensification can be achieved is still unclear, as climate change and competition for land (e.g. bioenergy crops) are further increasing the pressures on both agriculture and the environment.

In facing challenges related to sustainable intensification, climate change and the increasing demand for the delivery of ecosystem services from rural environments, there is a need for a horizon-scanning approach to determine to what extent existing research will allow us to face these challenges and what further research is required to enable an appropriate balance between the protection of aquatic ecosystems and agriculture into the future. The aim of the Environmental Protection Agency (EPA)-funded AgImpact Project was to develop an evidence base for further research on mitigating the impact of agriculture on aquatic ecosystems in Ireland. This was completed through a systematic and participatory review process of relevant environmental and behavioural research findings in Ireland.

The review is based on the primary research question used in the systematic mapping process: "What evidence exists to link agricultural practices with ecological impacts in Irish water bodies?". The systematic map of the literature provided a preliminary list of research gaps that were then presented to stakeholders for consideration. As a result of the feedback provided through an online survey and workshop, the project team revised this preliminary list of research gaps to reflect stakeholder input on the priority research areas. The final list of research gaps was then considered in the context of the available Irish and international literature, which forms the basis of the review provided in Chapter 3 of this report. It should be noted that this is not a comprehensive review: it focuses only on priority research gaps identified through a combination of the systematic mapping process and stakeholder engagement.

The review was prepared in the context of the need to elucidate environmental and behavioural heterogeneity in order to reduce uncertainty in the pressure–impact relationship between agriculture and aquatic ecosystems. This is central to achieving both the sustainable intensification of agriculture and the delivery of multiple ecosystem services within catchments, and the rationale for this is outlined in Chapter 3. Five broad topics and related sub-topics that require further research are also addressed (see Chapter 4):

- the transformation and attenuation of contaminants:
  - subsurface hydrological pathways;
  - natural water retention features;
- water body response:
  - multiple stressors;
  - emerging contaminants;
  - waterbody sensitivity;
- climate and weather patterns;
- predicting sources and responses:
  - source apportionment;
  - targeting measures;
- behavioural heterogeneity:
  - stakeholder engagement;
  - stimulating collective behaviour.

The report concludes that land use interventions to improve water quality, evaluated under "representative" conditions and scales, often do not respond as predicted if applied over a wide geographical area. Natural heterogeneity is the foundation that underpins the environment's ability to deliver multiple ecosystem services to society. The ability to accurately predict

the heterogeneity of catchment characteristics and ecosystem responses is central to achieving both the sustainable intensification of agriculture and the delivery of multiple catchment ecosystem services. However, although the issues of heterogeneity, uncertainty and targeted interventions must be addressed, there is still a general need to reduce source pressures, such as agricultural nitrogen and phosphorus inputs, and to promote best management practices at farm scale.

The report highlights the fact that a major difficulty in achieving sustainable intensification lies in the need to reverse agricultural impacts rather than prevent them, and the associated uncertainties related to ecological recovery trajectories, the continued impact of historical land use practices, lag times and the reluctance to jeopardise agronomic productivity. If agri-environmental solutions are to be pre-emptive in the future, then the ability to account for natural heterogeneity within and between catchments is essential. Evidence to support the need for targeted interventions has steadily increased in recent years, with an acknowledgement that an over-reliance on a “one-size-fits-all” approach is inadequate if the objectives of the WFD are to be achieved. Although scientists have highlighted uncertainty due to heterogeneity and have advocated a need for a targeted approach to interventions, there remain significant challenges with regard to providing agri-environmental solutions that are acceptable to end-users and that account for variability in catchment characteristics, farming systems and farmer behaviour. Addressing the research gaps detailed in this report will enable significant progress to be made towards providing these solutions.

Advances in decision-making processes, knowledge exchange and technology will be central to addressing many of the research gaps identified in this review. Advances in technology, related to monitoring, imaging, remote sensing, sensors and analytical instrumentation, will facilitate both a greater understanding of catchment processes and the ability to predict heterogeneity over wider geographical areas. Accounting for behavioural heterogeneity in both decision-making processes and knowledge exchange will also play a key role in the application of scientific knowledge, in order to provide effective solutions on farms. A research focus on heterogeneity will also facilitate a move away from policymaking based on the precautionary principle and support a greater focus on evidence-based decision making at the catchment scale.

1 Introduction

The increasing world population and changing consumption patterns, as well as changes in the policy environment [e.g. the removal of the European Union (EU) milk quota in 2015] and the ambition of the agri-food industry to expand, will drive an increase in outputs from Irish agriculture in the future. With the spatial expansion of agriculture in Ireland limited by the availability of suitable land, increases in agricultural outputs will be driven by the intensification of farming in existing areas through improvements in technology and resource efficiency. This will have to be achieved within the context of the targets established by EU environmental directives, such as the Nitrates Directive (ND) (91/676/EEC) and the Water Framework Directive (WFD) (2000/60/EC). Since the implementation of the EU WFD, estimates of the likelihood of achieving the standard of "good ecological status" in impaired water bodies by 2027 have largely focused on the uncertainty in lag times between the implementation of programmes of measures (POMs) and improvements in water quality (Schulte *et al*., 2010; Fenton *et al*., 2011; Doody *et al*., 2012a). However, as Ireland seeks to implement the schedule of the WFD, there is also significant uncertainty in relation to achieving these targets within the economic climate of agri-food industry buoyancy and the important and ambitious growth targets for agriculture defined in the Food Wise 2025 strategy (DAFM, 2015). How and if sustainable intensifications can be achieved is still unclear, as climate change and competition for land (e.g. bioenergy crops) are further increasing the pressures on both agriculture and the environment.

The Environmental Protection Agency (EPA) report *Ireland's Environment 2012 – An Assessment* (EPA, 2012) details the status of water quality in 2012 and concludes that agricultural diffuse and small point sources of pollution were the main cause of 50% and 13% of river water and groundwater pollution, respectively. Over the past 20 years, there has been significant investment in research related to the impact of agriculture on water quality in Ireland. Four large-scale research projects have, to date, given rise to three seminal reports on agriculture and water quality (EPA, 2000, 2008; Teagasc, 2013) and a fourth report arising from the Pathways Project (EPA, 2016a). Research reported by Tunney *et al*. (2000) and Carton *et al*., (2008) has been instrumental in defining the strategy for the implementation of the WFD and ND in Ireland, while the Agricultural Catchments Programme (ACP) (Teagasc, 2013) has played a central role in testing the effectiveness of measures implemented through the Nitrates Action Programme (Teagasc, 2013). The Pathways Project focused on developing a catchment management tool to aid with the targeting of measures within catchments (EPA, 2016a); the outputs of this project are expected to play a significant role in managing the impact of agriculture on water quality in the future.

As a result of the significant investment in research, there is now a good understanding of the factors controlling the impact of agriculture on water quality in Ireland. However, in facing the challenges of sustainable intensification, climate change and the increasing demand for the delivery of ecosystem services from rural environments, there is now a need for a horizon-scanning approach to determine to what extent existing research will enable us to face these challenges and what further research is required to enable an appropriate balance between water quality protection and agricultural production into the future.

1.1 Aim and Objectives

The aim of the AgImpact Project was to develop an evidence base to inform further research on the mitigation of the impact of agriculture on water quality in Ireland in the context of achieving the targets of, inter alia, the EU WFD, and the Food Wise 2025 and Going for Growth reports. This was completed through a systematic and participatory review process of relevant research, carried out in Ireland, on the impact of agriculture on aquatic ecosystems, while a standard review methodology was used to examine the related research on farmer engagement and the behavioural-change (i.e. socioeconomic) aspects of reducing the impacts of agriculture on water quality. To achieve this aim, the three key objectives of the project were:

1. to conduct a systematic review of the research using a standardised (socioeconomic) and peer-reviewed methodology (biophysical), in order to

identify gaps and drivers of future research on agriculture and water quality;

2. to elicit the input of a wide range of experts in this field of research with the aim of identifying future research needs;
3. to develop recommendations to inform the prioritisation of research on mitigating the impact of agriculture on water quality.

The systematic review approach taken to identify a preliminary list of research gaps is outlined in section 2.1. Input on this preliminary list of research gaps was obtained from a wide range of stakeholders through an online survey and a stakeholder workshop (as outlined in section 2.2). Section 2.3 of this report describes how the stakeholder input was combined with the knowledge obtained during the review process, which led to the identification of areas for further research and the research priorities for the future.

A second report arising from the AgImpact Project, entitled "Identifying Approaches to Improving Knowledge Exchange (KE) in the Irish AgriFood Sector using Expert Opinion" (EPA, 2016b), and a related workshop summary report are available on the EPA Secure Archive for Environmental Research (SAFER) website (http://erc.epa.ie/safer/).

2 Methods

2.1 Systematic Map

Although recent publications on agriculture and water quality have provided useful reviews of research in this area in Ireland, there is a need for a review process that is more systematic and applies scientific rigour and objectivity to the selection and interpretation of scientific/research literature, with the aim of minimising bias and the lack of reproducibility that can arise in more traditional, narrative-based literature reviews. A systematic review is a robust and transparent method of providing an evidence base for the identification of gaps and recommendations for further research requirements. The systematic review conducted in this project followed a standardised method developed by the Collaboration for Environmental Evidence (CEE; www.environmentalevidence.org), which is the leading authority on this methodology in environmental sciences. This method has been adopted by UK government departments as an effective approach to evidence-based policymaking. The systematic mapping protocol developed for this project was peer reviewed in the *Journal of Environmental Evidence* (Doody *et al*., 2015).

In addition, the multidisciplinary nature of this research area means that the prioritisation of research gaps in this report benefited from the input of a wide range of experts. Therefore, by making this review process participatory in nature, it was possible to utilise the knowledge and experience of a range of experts.

2.1.1 Objective of the systematic map

The objective of this systematic map was to develop an evidence base to inform the selection of a preliminary list of gaps in the research, on the mitigation of the impact of agriculture on water quality in Ireland, that could be presented to stakeholders during the stakeholder consultation process. The development of this list of research gaps was undertaken in the context of, inter alia, achieving the targets of the EU WFD and the Food Wise 2025 report. The EU WFD aims to protect and improve water quality using a river basin approach. Ireland's Food Harvest 2020 strategy (DAFM, 2010) aims to increase volumetric production in the dairy sector by 50% and beef output by 40%, and the report also details production targets for the sheep, pig, food and energy, forestry and marine agricultural sectors. The challenge of trying to achieve these production targets of Food Harvest 2020 and, subsequently, Food Wise 2025, while complying with obligations under the EU WFD in Ireland, has not yet been addressed. As the objective of this research is broad in scope, the systematic map was restricted to the island of Ireland to allow a full investigation of all studies related to the impact of agriculture on water quality.

2.1.2 Primary research question

Conducting a systematic map first requires the identification of a central research question. The question for this systematic map is:

- "What evidence exists to link agricultural practices with ecological impacts in Irish water bodies?"

In this context, Irish water bodies comprise all surface, groundwater and estuarine water bodies, including wetlands, coastal bodies and turloughs, on the island of Ireland. Agricultural practices include all farming activities and agricultural land use related to the pig, poultry, beef, dairy, sheep and arable farming sectors that potentially impact on water quality. In addition, studies that focus on agricultural mitigation measures to reduce the impact of agriculture on aquatic ecosystems were included. Ecological impacts include all outcomes related to changes in the ecological, chemistry and hydromorphology of water bodies including nutrients, sediment, microorganisms, hydrology and macroinvertebrates.

Secondary questions (sub-questions)

There are two related sub-questions which focus specifically on the delivery of pollutants and the effectiveness of mitigation measures:

1. "Has the delivery of agricultural pollutants to groundwater and surface waters been demonstrated?"
2. "Has the impact of mitigation measures on ecological and chemical water quality been demonstrated?"

Sub-question 1 was included because it is implicit in the primary review question that there is a link between chemical and ecological water quality, and both of these factors need to be addressed when determining the impact of agriculture on water bodies. As such, the first step in evaluating the evidence for a link between agricultural practice and ecological status was to determine if any research has demonstrated that agricultural pollutants are being delivered to water bodies. Sub-question 2 was included because mitigation measures will play a central role in achieving the objectives of the WFD in the context of the growth of the agricultural industry.

2.1.3 Methods and design

Data mapping and presentation

The systematic map outputs were recorded in the form of a database of studies, which describes the nature of the evidence on the review topic and the corresponding study locations. This database is easily searchable and freely accessible via the Irish EPA website (http://erc.epa.ie/safer/) and the CEE website (www.environmentalevidence.org). Once the systematic mapping process had been completed, the findings were used to identify research gaps, and research from other countries, particularly the UK, was used to inform the selection of these research gaps. An online survey was circulated to key experts in the areas of water quality and agriculture in Ireland and the UK. These experts were asked to evaluate the research gaps based on a number of criteria (detailed in section 2.2.2). Subsequently, the experts were invited to a workshop to further discuss and prioritise the areas of research. The systematic map, online survey and workshop outputs then formed the basis of a final list of recommendations for future research related to agriculture and water quality in Ireland.

Searches

Prior to the selection of search terms, review articles covering the impact of agriculture on water quality were consulted. Search terms were identified based on the *population*, *exposure* and *outcomes* components of the primary question. In addition, the search term "Ireland" was included in the search criteria in order to geographically restrict the search to just the island of Ireland. Because of the broad extent of the research question, the geographical scope of the systematic map was limited to the island of Ireland, in order to ensure that the number of studies included in the map was manageable and directly relevant. The *comparator* component of the primary question was not covered by the search terms, and the inclusion of spatial and temporal replication formed a key part of the subsequent selection process. A preliminary list of search terms was circulated within the project team. After a number of iterations, the list of search terms was then circulated to members of the project steering committee that oversaw this project. The final list of search terms used in this systematic mapping process is detailed in Table 2.1. The language was restricted to English, as this is the primary language used in scientific publications in Ireland.

Databases and other sources

When conducting the searches, a wide range of databases and sources were utilised. The use of the online databases detailed below was based on a review of previous systematic maps/reviews that covered topics similar to this study. Eight online databases were identified:

1. Web of Science (All databases), which covers:
   (a) Web of Science Core Collection;
   (b) CAB Abstracts – CABI;
   (c) FSTA – Food Science and Technology Abstracts;
   (d) MEDLINE;
   (e) SciELO Citation Index;
2. ScienceDirect;
3. DOAJ – Directory of Open Access Journals (including PloS ONE);
4. Copac;
5. Agicola;
6. CSA Iliumina/Proquest;
7. GreenFile;
8. Google Scholar.

The search strings presented in Table 2.1 are in the optimum format for Web of Science, but these were altered to suit other systems (e.g. Google Scholar). Consistent with guidance from the CEE, the first 100 hits from Google Scholar were selected for further review.

Table 2.1. Search criteria used in the systematic map of the impact of agriculture on the ecological status of Irish water bodies

Search terms[a]		Web of Science hits
Population	*"*surface water*" or "drainage water" or waterbod* or river* or lake* or estuar* or stream* or turlough* or groundwater or "ground water" or groundwater* or wetland* or canal* or pond* or spring* or catchment* or watershed or coast* or transitional or drain* AND*	**2,402,700**
Exposure	*farm* or agricul* or pig* or poultry or dairy or beef or sheep or livestock or cattle or tillage or arable or grassland* or grazing or slurry or manure* or fertil* or "nutrient management" or "stocking rate*" or "stocking densit*" AND*	**372,539**
Outcome	*micro* or "e. coli" or "Escherichia coli" or "water quality" or phosphorus or nitrogen or nitrate* or ammon* or sediment* or "organic matter" or nutrient* or pesticide* or herbicide* or "sheep dip*" or "endocrine disruptor*" or toxicant* or toxin* or antibiotic* or hydrology or hydrogeomorphology or ecolog* or macroinvertebrate* or pollution or fish or ecology* or status* AND	**237,512**
Location	Ireland	**965**

[a]In the search strings, '*' denotes a wildcard search operator.

A selection of specialist websites from Ireland and Northern Ireland were also searched for relevant publications that may not have been identified in the online database searches. The following websites were searched:

- "SAFER-Data: Access to Printable EPA/ERC Reports" (http://erc.epa.ie/safer/reports);
- the EPA Research Database (http://erc.epa.ie/smartsimple/);
- the Department of Agriculture Food and the Marine website (http://agriculture.gov.ie/);
- the Agriculture and Food Development Authority website (www.teagasc.ie);
- the Northern Irish Department of the Environment website (http://www.doeni.gov.uk/niea/);
- the Northern Irish Department of Agriculture, Environment and Rural Affairs website (http://www.dardni.gov.uk/);
- the Sustainable Water Network (SWAN) website (http://www.swanireland.ie/);
- the Department of Housing, Planning, Community and Local Government (formerly known as the Department of Environment, Community and Local Government) website (http://www.environ.ie/en/);
- the Engineering Village website (http://www.engineeringvillage.com);
- the EPA SAFER website (http://erc.epa.ie/safer), that is, not just the printable EPA reports;
- the Water Joint Programming Initiative (JPI) website (http://www.waterjpi.eu), which contains links to several projects and filters can be used to include those related to only Ireland; there are also links to projects other than those that are EPA funded
- the RIAN website (http://rian.ie/), which is the new portal for open access Irish research publications;
- the Water Matters website (http://www.wfdireland.ie).

Search comprehensiveness assessment

The number of Web of Science hits for each set of search terms is shown in Table 2.1. Several approaches were taken to ensure that a comprehensive list of articles was identified when using these search terms. A list of 32 key documents, collated based on the bibliographies of existing reviews of Irish research, were used to test the comprehensiveness of the search terms used in Web of Science (Appendix 1). The search terms were adjusted until all 32 key documents were found. These documents were selected based on knowledge of existing research on agriculture and water quality in Ireland and included studies on source, mobilisation impact pathways, the nutrient transfer continuum and key contaminants. A range of publication dates, from 1974 to 2013, and a variety of authors were also included in the selection. The reference lists of previous Irish review articles and the project participants' personal archives of Irish articles related to this area of research were also compared with the articles identified using the search criteria detailed in Appendix 1. After the removal of

erroneous, irrelevant articles, the full list of references was sent to a range of researchers, across Ireland, that have published regularly on this topic; these researchers were asked whether or not any relevant articles had been omitted. In addition, authors were emailed to ask if any omitted article was unattainable through the normal channels.

Study inclusion criteria

This systematic map protocol aimed to identify the evidence that exists to link agricultural practices with ecological impacts on Irish water bodies. For inclusion in the study, the following core criteria were assessed:

- **The relevant population**: Only studies of Irish water bodies, which comprise all surface, groundwater and estuarine water bodies, including wetlands, coastal bodies and turloughs, within the island of Ireland, were included.
- **The types of exposures/intervention**s: All studies on farming activities and agricultural land use, related to the pig, poultry, beef, dairy, sheep and arable farming sectors, that potentially impact on water quality were considered. In addition, studies that focused on agricultural mitigation measures to reduce the impact of agriculture on aquatic ecosystems were included.
- **The types of comparators**: Studies that assessed water quality before and after changes in farming practices and/or implementation of mitigation measures (temporal comparator), or comparisons with similar areas of zero-/lower-intensity agriculture or areas in which no mitigation measures had been implemented (spatial comparator) were permitted.
- **The types of outcomes**: Only studies that assessed ecological impacts related to changes in the ecological, chemical and hydromorphology of water bodies, including changes in nutrients, sediment, microorganisms, hydrology or macroinvertebrates, were included.
- **The types of study**: As this review is focused on the ecological impacts of agriculture, only studies that were carried out at a greater-than-field scale and with multiple spatial or temporal replication were included in the systematic map. Although studies carried out at a smaller scale than field scale are invaluable for understanding natural processes, they do not directly evaluate the impact of the delivery of pollutants or their impact on water bodies. However, if an included document made reference to smaller scale studies, this document was reviewed, if necessary, to elucidate the large-scale processes occurring at field or catchment scale.

It was hypothesised that searching for studies conducted using the term “Ireland”, in order to restrict the geographical scope, may have excluded studies conducted in Northern Ireland if such studies had used the term “United Kingdom” or “Great Britain” to define their geographical scope. The extent of this problem was investigated and was found to be insignificant: only 1 out of every 1000 articles examined, which relate to studies that included Northern Ireland as part of larger scale investigations in the UK or Great Britain, was not picked up using the search criteria in Table 2.1.

In order to identify a core list of articles for inclusion in the systematic map, a hierarchical screening process was undertaken. This hierarchical process covered the title, abstract and full text, and this screening was based on the predetermined criteria detailed above. If doubt over the validity of a document arose during the screening process, it was noted and assessed further. The title of a paper was screened first to remove erroneous articles unrelated to water quality. Subsequently, the abstracts were reviewed to determine whether or not the article relates to water quality and agriculture in Ireland. General review articles were excluded at this stage of the screening process. Reports that did not contain abstracts were automatically progressed to the full-text screening stage, which was based on the criteria shown in Table 2.2.

Data extraction strategy

Table 2.2 details the coding criteria that were used to extract information from individual studies. The coding criteria were initially identified during a review of similar systematic map studies and subsequently revised through an iterative process involving the project team and steering committee. The coding criteria were first applied to a selection of key papers on the impact of agriculture on water quality, and then revised again based on the outcome of that scoping study. The data extracted from each document were recorded in a Microsoft Access database.

Coding criteria 1–19 in Table 2.2 relate to the characteristics of the identified study and the study site; these

Table 2.2. Coding criteria for the systematic map of the impact of agriculture on the ecological status of Irish water bodies

Criterion no.	Coding variable	Details/examples
1	Author(s) and affiliation(s)	
2	Full reference	
3	Publication type	Book chapter, journal paper, report
4	Holding institution	Organisation holding access to the document
5	Document access issue	Open access or subscription only
6	Funding agency	
7	Study start and end dates	Date(s) on which study was carried out
8	Study length	Duration of study
9	Study description	Brief overview of the study
10	Study scale	Regional/catchment/farm/field/plot/laboratory
11	Experimental design	
12	Water body type	Estuary/lake/river/groundwater; WFD category, e.g. coastal
13	Description of water body	Brief description of water body
	Status of water body	How affected (ecologically, chemically and hydromorphologically) is the water body under investigation?
14	Study location	Where within Ireland
15	Farming type	Dairy, livestock, sheep, arable, mixed
16	Soil description	Classification, permeable/impermeable, soil fertility
17	Land use type (dominant and other land use)	Pasture, rough grazing, arable, commonage, mixed
18	Description of exposure	Description of the agricultural pressure (e.g. farming type, intensity, derogation, etc.) the water body is being exposed to
19	Exposure time period	Length of time water body has been exposed
20	Mitigation description (if any)	Description of exposure mitigation
21	Mitigation time period (if any)	Length of time mitigation measure has been in place
22	Comparator description	For example, brief description of spatial and/or temporal comparator
23	Comparator type	Spatial and/or temporal
24	Replication	Number and unit of replication
25	Methodological detail	The level of detail provided in the method description (i.e. low, medium, high or obvious detail missing)
26	Outcome focus	Which aquatic variables have been measured (e.g. physico-chemical, ecology, hydrology, geomorphology)
27	Dominant hydrological pathway of export	Overland flow, drain flow, interflow, groundwater, transition zone
28	Measured outcomes	All outcome terms detailed in Table 2.1
29	Impact on biological status	Details of impact on macroinvertebrates, fish, macrophytes, algae, etc.
31	Impact on chemical status	Details of impact on phosphorus, nitrogen, conductivity, biological oxygen demand, etc.
32	Impact on hydromorphological status	Details of impact on riverbed substrate, hydrology, riparian zone, etc.
33	Evidence linking exposure/mitigation to outcome	Yes/no
34	Strength of evidence for impact	Low/medium/high
35	Description of evidence linking exposure/mitigation to outcome	Brief description
36	Scale at which dominant source area identified	Point location, field, farm, land use type, sub-catchment, catchment
37	Other compounding sources reported	Septic tanks, WWTPs, point sources, urban runoff
38	Other compounding stressors reported	Invasive species, lack of a species pool, hydromorphology
39	Policy relevance	WFD, ND, Phosphorus Regulations, etc.

WWTP, wastewater treatment plant.

were used to evaluate the validity of each study in relation to assessing the impact of agriculture on ecological water quality. In addition, in the future, these criteria will also provide end-users of the database with the ability to search the database on the basis of these specific criteria. The remaining coding criteria detailed in Table 2.2 were designed to extract information from each document on the impact of agriculture on ecological water quality. In addition, the criteria were designed to extract information on the impact of mitigation measures and whether or not the delivery of agricultural pollutants to water bodies has been demonstrated. All the information was extracted and used as the input for the database was then used to perform a final assessment of the strength of the evidence.

Critical appraisal of a study's internal validity

Unlike a systematic review, a full critical appraisal is not necessary for systematic mapping. However, to allow users to evaluate the internal validity of each study, the following coding variables were used: *study length*, *study scale*, *experimental design*, *comparator type*, *replication* and *methodological detail*.

2.2 Stakeholder Engagement

On completion of the systematic mapping process and standard review of the behavioural-change aspects of mitigating the impact of agriculture on water quality, a preliminary list of research gaps (see section 2.2.1) was identified by the project team based on the information gathered. This formed the basis of an online survey that was sent out to a wide range of expert stakeholders. This survey listed 24 research gaps for stakeholders to evaluate; these were divided into five categories (Parts A–E of the survey; see section 2.2.1). Survey participants were asked to evaluate and rank the relative importance, in their own opinion, of the research gaps on the basis of five evaluation criteria (detailed in section 2.2.2) . Further space was provided for participants to suggest additional research gaps that they felt were not covered in the list provided (Appendix 2). Because this evaluation involved the relative ranking of each of the research gaps identified, stakeholders had to familiarise themselves with the full list of research gaps before commencing evaluation. The research gaps and the evaluation criteria are detailed below.

2.2.1 Preliminary list of research gaps included in online survey

Part A – Minimising the sources and mobilisation of contaminants (nutrients, sediment, pesticides, endocrine disrupters, microbial, etc.)

- A1: Assessments of cost effectiveness and the feasibility of approaches for targeting management strategies at critical source areas (CSAs) within catchments;
- A2: Methods to apportion the relative ecological impacts of different contaminant (legacy and contemporary) inputs to freshwaters;
- A3: The integrated use of microbial source tracking (MST) with hydrological and hydrochemical data to identify sources and evaluate the impact of management strategies.

Part B – Understanding and intercepting the pathways of contaminant transport

- B1: Optimising natural water/contaminant retention features within catchments (e.g. wetlands, riparian zones, ditches);
- B2: Estimating the attenuation of contaminants in different subsurface hydrological pathways;
- B3: Reduction in the uncertainty associated with the catchment-scale models used for the identification of the hydrological pathways responsible for contaminant export in catchments.

Part C – Impact of agricultural pressures on aquatic receptors

- C1: The long-term evaluation (≥10 years) of the effectiveness of agricultural point and diffuse management strategies at catchment scales;
- C2: The reduction in the uncertainty associated with the catchment-scale models used for the evaluation of the effectiveness of management strategies;
- C3: Evidence to support the linkages between agricultural practices and ecological change in aquatic ecosystems;
- C4: Disentangling the impact of multiple stressors on aquatic ecology;
- C5: The identification of synergies between contaminants and their role in determining the ecological state of aquatic ecosystems;

- C6: The identification of catchment-specific thresholds, for multiple stressors, at which ecological recovery of aquatic ecosystems may occur;
- C7: The evaluation of the relative impacts of sediment and hydromorphology on ecological status compared with other stressors;
- C8: Extending existing research on the impact of agricultural impacts on groundwater to a broader range of geological contexts (e.g. bedrock, soil, weathering history) and aquifer typologies;
- C9: Disentangling the effect of management strategies from natural denitrification and recharge processes.

Part D – Impacts of climate change

- D1: The impact of climate change on the effectiveness of management strategies to mitigate the impact of agriculture on water quality;
- D2: The impact of climate change on aquatic ecosystem responses to agricultural land use practices;
- D3: The impact of climate change on agricultural water-use efficiency and ecological flows (Eflows) in water bodies.

Part E – Socioeconomic research

- E1: The development of approaches to facilitate "payment by results" through agri-environmental schemes (i.e. ecosystem services) rather than payments for prescribed activities (i.e. current agri-environmental schemes);
- E2: The evaluation of approaches for increasing farmer learning and awareness of the links between management practices and successful conservation;
- E3: Approaches to stimulating collective conservation behaviour by farmers for the protection of water quality;
- E4: Approaches to empowering land owners to engage in agri-environmental decision making at local, regional and national scales;
- E5: Catchment-specific approaches to the delivery of extension services and knowledge transfer for the protection of water quality;
- E6: Determining a farmer's capacity to adapt to the changes required for the protection of water resources.

2.2.2 Evaluation criteria

Survey participants were asked to rank the importance of the above-listed research gaps on the basis of the evaluation criteria detailed in the following sections. Appendix 3 provides the overall ranking of the top 10 most important research gaps, as determined by the evaluations carried out by the stakeholders who responded to the online survey.

Criterion 1 – Importance with regard to understanding the fundamental relationship between agriculture and water quality

Some research may not be directly or immediately relevant to policy or regulation. However, it may be a key step in improving our fundamental understanding of the processes controlling the relationship between agriculture and water quality. Rank the question as None, Low, Moderate or High Priority. A rank of "No Priority" means that the research gap is not a priority for understanding the fundamental relationship between water quality and agriculture, while a rank of 'High Priority' means that it is a high research priority to elucidate this understanding. Where a research gap is given a 'High Priority' ranking, please provide a brief justification for this rank.

Criterion 2 – Importance with regard to achieving the objectives of the Water Framework Directive

The central objective of the WFD is to achieve good status or maintain high status in all waterbodies by 2027. Rank the question as None, Low, Moderate or High Priority. A rank of 'No Priority' means that a research gap is not a priority for achieving the objective of the WFD, while a rank of 'High Priority' means that it is a high research priority to achieve these objectives. Where a research gap is ranked as 'High Priority', please provide brief justification for this rank.

Criterion 3 – Importance with regard to achieving sustainable agricultural intensification in Ireland

Achieving the objectives of the WFD is made more challenging when the expected expansion of the agricultural industry in Ireland is taken into consideration. This criterion is focused on identifying research to enable sustainable agricultural growth in Ireland. Rank the question as None, Low, Moderate or High Priority. A rank of "No Priority" means that a research gap is not

a priority for achieving sustainable intensification, while a rank of "High Priority" means that it is a high research priority to achieve these objectives. Where a research gap is given a "High Priority" ranking, please provide a brief justification for this rank.

Criterion 4 – Difficulty in closing the research gap

The capacity to close certain research gaps may be limited by the availability and/or feasibility of techniques, methodologies and understanding of the underlying processes. Rank this question as None, Low, Moderate, or High Difficulty. In this context a rank of "No Difficulty" means there are available techniques, methodologies and/or knowledge to fill this research gap. If you feel a research gap has already been filled by research in Ireland or elsewhere, please select "No Difficulty" and indicate in the comments section where this research can be found. A rank of "High Difficulty" means that a gap cannot be filled until further techniques, methodologies and/or knowledge are developed. Please indicate, where possible, the techniques and/or knowledge that are required to conduct research in this area.

Criterion 5 – Timescale

In the context of your assessment of criteria 1 to 4, please, at the end of this survey, we ask you to select up to 10 research gaps which should be tackled in the short term (i.e. 3–4 years) and up to 10 research gaps which should be tackled in the medium term (i.e. 5–10 years).

2.2.3 AgImpact workshop

After completion of the survey and analysis of the results, a 1-day workshop was held, on 29 January 2015 in the O'Callaghan Alexander Hotel (Merrion Square, Dublin), to which stakeholders (i.e. from research, the agri-food industry, farming groups and knowledge transfer groups, and policymakers) were invited. The workshop took the following format (see also the workshop agenda in Appendix 4 and the EPA SAFER website):

- After an introduction, a brief presentation was given by the project team on the AgImpact Project and the findings of the survey.
- After questions, the participants were divided into four breakout groups, the make-up of which was predetermined in order to ensure that different types of stakeholders were in each group. Three breakout groups focused on the research gaps that were deemed to be priorities, while the fourth group focused their discussions on the challenges that will face the agricultural industry if the sustainable intensification of agriculture is achieved. Each group was assigned a chairperson who was asked to facilitate the discussions, and a note taker to record the conversations. The first three breakout groups were provided with a template on which to record their rationale for selecting a topic as a research gap (see Appendix 5).
- Over lunch, the AgImpact team collated the responses of the breakout groups into a matrix. The afternoon session was facilitated by Professor Rogier Schulte of the Agriculture and Food Development Authority Teagasc, who, first of all, presented a framework for the ensuing discussions. During the discussions, each breakout group provided feedback on their selection and rationale for specific research gaps, and Rogier Schulte probed and examined how each research gap fitted into the conceptual model.
- After feedback from the breakout groups, in the final part of the workshop, members of the sustainable development breakout group were given the opportunity to provide input for a general discussion on the way forward. All of the afternoon discussions were recorded by a stenographer.

The information collected during the systematic review, survey and workshop was used to identify a final list of research gaps, which are detailed in the following sections. Although "improving the exchange of knowledge between research, policy and practice for the protection of aquatic ecosystems in the context of sustainable intensification" was identified as major research priority, this issue subsequently formed the basis of an extension of the AgImpact Project and, therefore, is the subject of a separate report (EPA, 2016b).

2.3 The Context of the Literature Review

The literature review based on the primary research question ("What evidence exists to link agricultural practices with ecological impacts in Irish water bodies?") used in the systematic mapping process is presented in Chapter 3.

The systematic map of this literature provided a preliminary list of research gaps, which were then presented to stakeholders for consideration. As a result of the feedback provided through the online survey and workshop, the project team revised this preliminary list of research gaps to reflect stakeholder input on the priority areas. In Chapter 4, the project team considers this list in the context of the available Irish and international literature. At the end of each section in Chapter 4 addressing a specific research gap, a list of key knowledge gaps are summarised. Each section is not intended to provide a comprehensive review of each topic, but instead to provide a rationale for the suggested further research. In this regard, it should be noted that the review is not all inclusive, but rather focuses on research gaps deemed to be priorities through a combination of the systematic mapping process and stakeholder engagement. Other potential research topics are listed in section 2.2.1 and Appendix 1.

3 Literature Review

3.1 Introduction

Much of the research related to water quality in Ireland over the past 40 years has focused on understanding, quantifying and/or mitigating the contribution that agriculture has made to the decline in chemical water quality. Since the implementation of the WFD in 2000, research efforts have increasingly focused on determining how agricultural practices are linked to ecological water quality (Donohue *et al*., 2006). However, the accurate source apportionment of pollutant inputs to water bodies remains a significant challenge because of the multiple pressures that contribute to these impacts, the lag times in responses, the differences in the bioavailability of sources and their timing of delivery, and the large number of physical and biological factors that influence chemical water quality (Hamilton *et al*., 2012; Jarvie *et al*., 2013). The difficulty of linking agricultural pressures to ecological impacts has, consequently, presented a complex and holistic problem for research (Moss, 2008; Page *et al*., 2012). Despite this complexity, threshold targets for some pollutants, necessary to limit or reverse chemical and ecological impacts, have been identified under the WFD and ND. In Ireland, these threshold targets relate to phosphorus (P) and nitrogen (N), as these are the primary controls over chronic eutrophication and subsequent loss in ecological biodiversity, and risks to human health (Smith and Schindler, 2009). There is also increasing emphasis on the wider provisioning, regulating and cultural services that aquatic ecosystems provide for societal well-being, and water quality regulation contributes to the delivery of a number of these services, from recreation to human health (Keeler *et al*., 2012).

Research studies in Ireland have demonstrated a clear relationship between agricultural intensification and changes in aquatic ecosystems at a number of different scales. Foy and Lennox (2006) investigated agricultural intensification in the Lough Neagh catchment, and demonstrated that changes in P inputs and exports have occurred in the catchment since 1925. Inputs of P increased from 1176 kg P/km^2 per year in 1925 to 3823 kg P/km^2 per year in 2000, and 85% of this increase predated 1975. However, they found that P exports to Lough Neagh increased by 238% between 1975 and 2000, despite a reduction in input from point sources, and concluded that these exports were associated with an increase in soil P levels in agricultural areas of the catchment. Donohue *et al.* (2006) found significant inverse relationships between a range of land use pressures, including agricultural intensity and cattle densities, and ecological status in 797 river catchments in Ireland. They identified a threshold value of 38% pasture, above which there was a significant decline in the probability of a site having a good ecological status (Donohue *et al*., 2006). Similar trends in N and P inputs to agricultural land associated with agricultural intensification have been observed in England and Wales (Withers *et al*., 2001), although it has been more difficult to establish a direct causal link between agricultural P use and water chemical status because of the ubiquitous presence of wastewater outflows (Withers *et al*., 2014). For example, in an analysis of the links between water quality and catchment characteristics across the UK, Davies and Neal (2007) found that although mean river nitrate concentrations were linked to land use types, mean river phosphate concentrations were linked more strongly to population density. Foy (2007) examined river P concentrations across Europe and found that urban populations were the main driver. Similarly, some recent notable improvements in chemical and ecological water quality in Great Britain and Europe (Vaughan and Ormerod, 2012) have, so far, been associated more with reductions in wastewater nutrient loadings than with reductions in diffuse agricultural inputs.

At the smaller sub-catchment scale, in the Colebrooke and Upper Bann catchments in Northern Ireland, Foy and Kirk (1995) and, more recently, Barry and Foy (2016) demonstrated clear links between agricultural intensity and both chemical and ecological impacts. Foy and Kirk (1995) reported a 1-unit decrease in the ecosystem class of fisheries with every increase of 0.6 dairy cow equivalent/ha. Dairy cow equivalent/ha was significantly positively correlated with maximum biological oxygen demand (BOD) and total N in the form of ammonium, and negatively correlated with dissolved oxygen concentration in 42 lowland streams in these two catchments. Barry and Foy (2016) demonstrated a significant relationship between manure N stocking rate and P flow-weighted mean concentrations (FWMCs)

in the same streams, concluding that the decline in P FWMCs between 1990 and 2009 was in response to the improved nutrient management in the catchments during this period (Barry and Foy, 2016). In England and Wales, catchment monitoring programmes under the Catchment Sensitive Farming (CSF) initiative, which is the main vehicle for delivering best management practice with regard to diffuse pollution, have found a significant reduction in river pesticide concentrations and an ecological response to reductions in sediment pressure as a result of CSF activities. The CSF phase 3 report suggests that there is, on average, a lag of 3 years before signs of water quality improvement become apparent (Natural England, 2014).

Conroy *et al.* (2016) observed local-scale impacts of cattle access on ecological water quality in Irish streams; these impacts were most pronounced during the summer and autumn months in rivers that have a high/good status. They reported significant changes in metrics, such as macroinvertebrate total richness and the Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) (EPT) Richness Index, downstream of access points in all four of the studied high-/good-status sites during summer and autumn. However, there were no consistent differences in sediment metrics, such as turbidity, between upstream and downstream cattle access points (Conroy *et al*., 2016). In contrast, a study in New Zealand demonstrated that turbidity and *Escherichia coli* concentrations increased considerably downstream of a dairy herd crossing point (Davies-Colley *et al*., 2004). Kibichii *et al.* (2015) examined the impact of water quality on river hyporheic invertebrate communities in six sub-catchments in Ireland that were dominated by agriculture. These authors found that a reduction in invertebrate density and richness was associated with higher agricultural intensities, management practices and reduced riparian buffering. The EPA-funded COSAINT project (https://www.teagasc.ie/environment/biodiversity--countryside/research/current-projects/cosaint-/) should provide further evidence on the impact of cattle access points on aquatic ecosystems and the effectiveness of mitigation strategies such as fencing off waterways.

Thompson *et al.* (2013) used sediment tracing techniques to identify the sources of sediment in three Irish catchments and, although river channel banks were the main source in two of the catchments, the proportion of sediment that originated from arable land was as high as 85% in the third catchment. In previous work by Evans *et al*. (2006) on the Bush catchment in County Antrim, a sediment fingerprinting technique was applied to identify the main sources of sediment inputs which had the potential to affect salmon spawning in the lower sections of the river. In addition to drainage maintenance work and forestry clearfelling, which together accounted for 61% of annual suspended sediment (SS) load and 32% of bed load in the river, bank erosion caused by cattle access (2% of annual SS load and 60% of bed load) and the poaching of ploughed arable land (37% of annual SS load and 8% of bed load) were also significant sources. Unlike chemical stressors, there are no standards for SS in Irish rivers (although a guideline exceedance limit of 25 mg/L is recommended for salmonid and cyprinid rivers) and few studies have made any link between the ecological impact of SS in terms of either concentration or the duration of exceeding SS load (Thompson *et al*., 2014). Work from countries other than Ireland suggests that siltation does have a major impact on aquatic ecology (e.g. Jones *et al*., 2012a,b). Using isotope labelling and infrared spectroscopy, Collins *et al*. (2013) recently found that particulate organic matter ingressing fish spawning gravels in the Blackwater catchment in England was derived from in-stream decaying vegetation (39%), damaged road verges (28%), septic tanks (22%), and farmyard manures and slurries (11%). These studies demonstrate the high degree of site-to-site variability in sediment sources and that it is not just agriculture that contributes to diffuse pollution, even in rural areas.

Agricultural nutrient inputs and soil P values have been used as indicators of agricultural intensity and the risk of eutrophication impacts in agricultural catchments, with Tunney *et al*. (2000) reporting that soil test-P concentrations in Ireland have increased 10-fold in the last 50 years. Jordan *et al.* (2000) examined the relationship between the soil and river P concentrations in 50 catchments and reported that there were 0.5 and 1.0 kg/ha increases in soluble reactive P and total P (TP) export, respectively, for every 10 mg/L increase in Olsen soil P concentration. At field scale, Kurz *et al*. (2005) recorded P loss in runoff from three agricultural fields that had Morgan's P concentrations of 4, 8 and 17 mg P/L; the corresponding average flow-weighted dissolved reactive P (DRP) concentration for each site ranged from 0.01 to 0.05 mg/L, 0.08 to 0.32 mg/L and 0.12 to 5.1 mg/L, respectively. This relationship was also highlighted by Watson *et al*. (2007) who demonstrated an increase in P loss in both overland flow and

subsurface drains in response to fertiliser P application, and Olsen's P accumulation in six hydrologically isolated drumlin field plots in Northern Ireland. Watson *et al*. (2007) found that TP export from the plots increased from 0.19–1.55 kg P/ha, for the plot receiving zero P, to 0.35–2.94 kg P/ha for the plot receiving 80 kg P/ha per year. On the same field plots, Watson *et al.* (2000) also demonstrated a linear relationship between fertiliser N application rates and nitrate loss in field drainage water, accounting for between 5% and 23% of the N applied. While none of these field-scale studies directly links these losses to chemical or ecological impacts on water bodies, they support the hypothesis that an increase in agricultural intensity increases the risks posed to aquatic ecosystems. In another study, McDowell *et al.* (2015) found elevated concentrations of P in the surface water and groundwater under dairy farms which had high soil P levels. However, other work suggests that direct links between lowering N and P surpluses and river N and P concentrations are not always apparent because of N emissions to air (Lord *et al*., 2002), soil P buffering (Cherry *et al*., 2008) or the contribution of other confounding nutrient sources (Oenema *et al*., 2005; Withers *et al*., 2014).

Research has also demonstrated the impact of incidental losses of organic and inorganic fertilisers on water quality. For example, Withers *et al*. (2003) concluded that if slurry is applied to the surface, it potentially contributes to 50–98% of the measured P lost in runoff. The rates of P loss varied spatially and temporally depending on the amounts of P applied, the properties of the manure and the timing of the application; the quantities of P lost varied from less than 1% to 25% of the TP applied (Withers *et al*., 2003). Following a 4-year study in Northern Ireland, Doody *et al*. (2010) highlighted the challenges posed with regard to identifying periods suitable for slurry spreading: soil moisture field capacity was exceeded on 50% of days in February and runoff was recorded, on average, on 33% of days in May. Preedy *et al*. (2001) reported that if slurry application coincides with rainfall, the TP concentration in runoff from a grassland plot was 4000 µg/L just 4 hours after slurry application and reached a peak of 7000 µg/L after 36 hours.

In Ireland, Kurz *et al*. (2005) found evidence for an increase in P concentrations in overland flow and subsurface flow after the application of slurry to an experimental grassland site. O'Rourke *et al*. (2010) demonstrated that there was still a risk of nutrient loss 48 hours after slurry application to 0.5-m^2 plots, thus extending beyond the current 48-hour guidelines. In fact, elevated nutrient concentrations were recorded for more than 9 days post application, with greater losses recorded in winter than in summer or spring (O'Rourke *et al.*, 2010). Doody *et al.* (2014a) demonstrated that slurry spreading on 0.2-ha field plots in Northern Ireland resulted in an accumulative P loss of between 0.73 kg/ha and 1.2 kg/ha over 2 years, despite adherence to best management practices including a 3.5-month closure period during the winter months. This contrasted with losses of 0.14 and 0.22 kg P/ha from the two plots that received no slurry during the same period. The majority of the P was lost during a number of large rainfall events, with losses occurring up to 20 days post application after dry periods (Doody *et al*., 2014a). Shore *et al.* (2016b) examined incidental losses at the catchment scale in Ireland and reported high FWMCs of P, and TP to SS ratios indicative of incidental losses, during wet summers and in autumn. Flynn *et al.* (2016) used MST techniques to demonstrate similar levels of water contamination at catchment scale after slurry application in the months following the closed period in the Mattock catchment.

In 2012, 85% of groundwater bodies in Ireland were deemed to be of good status under the WFD (EPA, 2012). The impact of agricultural practices on groundwater quality has been observed in a number of studies. EPA (2006) monitored the groundwater beneath an intensively managed dairy farm for 2 years and observed annual nitrate concentrations of 16.5 and 12.6 mg/L. The concentrations ranged from 3 to 31 mg/L depending on the location of the borehole on the farm and annual variation. Clear increases in concentration were associated with areas used for dirty water management and average values exceeded the target set in the ND and the Drinking Water Directive. However, this study was confined to a single site with free-draining soils overlying limestone bedrock, and, therefore, the results cannot be extrapolated to other landscape units or aquifer types. Richards *et al.* (2015) compared nitrate leaching for two different stocking rates and reported that N present as nitrate (NO_3-N) varied from 55 to 71 kg N/ha per year for the highest intensity grazing systems (210 kg organic N/ha), while for the lowest intensity grazing system (170 kg organic N/ha), NO_3-N loss was 15–20 kg N/ha per year. Ryan *et al.* (2006) monitored the changes in NO_3-N and N present as ammonium (NH_4-N) leaching from agricultural land on a free-draining soil

overlying fissured limestone and, although there was significant variation in NO_3-N concentrations between years, the 4-year mean NO_3-N and NH_4-N concentrations were 8.2 and 0.297 mg/L, respectively. Mellander *et al.* (2014) also reported significant annual variation in NO_3-N concentrations in two intensively farmed catchments and that events such as ploughing and reseeding resulted in short-term increases in NO_3-N concentrations of up to 23.9 mg/L to levels that are above the maximum acceptable concentration (MAC). Despite these peaks, the mean concentrations were below the current regulatory standard, and Mellander *et al.* (2014) concluded that monitoring programmes need to be cognisant of the occurrence of discrete land management events when interpreting sampling data, in order to distinguish between nitrate events and trends in groundwater bodies. Baily *et al.* (2011) used stable isotope techniques to identify sources of groundwater N on an intensive dairy farm in Ireland. After monitoring 24 wells across the farm, they found that organically derived N from manure was the main source of nitrate recorded in groundwater. Huebsch *et al.* (2013) examined evidence for the impact of changing N loads from an intensive dairy farm on nitrate concentration in a karst aquifer. The 11-year dataset demonstrates a decline in nitrate concentrations in the aquifer as a result of the implementation of best management practices, such as an improvements in N use efficiency and the timing of applications of organic and inorganic fertiliser. Most groundwater studies in the island of Ireland have focused on the south and south-east, which have different geologies and soil typologies from the north of the island. In a recent study covering the whole island, Orr (2014) highlighted the differences between the fate of NO_3-N in poorly productive aquifers in County Down and the regionally productive karst systems in County Kilkenny, and showed differential denitrification rates and a greater persistence of nitrate in karstic aquifers than in shallow pathways, and rapid denitrification in the poorly productive bedrock overlain by glacial till.

Kilroy and Coxon (2004) suggested that P in groundwater had received limited attention in Ireland. The limited availability of data on P in groundwater bodies in Ireland was also highlighted in a report by the Scotland & Northern Ireland Forum for Environmental Research (SNIFFER, 2008). According to this report, no data were available for 37% of groundwater bodies, which is partly because of a lack of monitoring of unproductive aquifers in Ireland (SNIFFER, 2008). A total of 28% of the monitored area in Ireland had P concentrations exceeding 30 μg/L with only 0.2% exceeding 60 μg/L. In Northern Ireland, 513 groundwater samples were analysed for P concentrations between 2000 and 2006, and, of these, 35 (7%) had P concentrations of 30 μg/L or above. Holman *et al.* (2008, 2010) found that P concentrations in monitored groundwater bodies in Ireland were often in excess of ecologically significant thresholds and could contribute significant quantities of P to surface water bodies. Holman *et al.* (2008) observed that P concentrations in groundwater were significantly related to land use in England and Wales; although this relationship had not been reported for Irish groundwater bodies, this is likely to be because of the limited data available (Holman *et al.*, 2010).

Although these data provide evidence of the impact of land use practice on P concentrations in groundwater in some catchments, the evidence linking these losses to agricultural sources has yet to be confirmed. In a karstic aquifer in County Mayo, Mellander *et al*. (2012) could not find any strong evidence for a link between P pollution and agricultural land use in almost-continuously monitored spring-fed stream, despite the fact that previous vulnerability assessments had suggested that the contributing area had a high to extreme risk of pollution. Differences were related to the need to incorporate both soil chemistry and P status and groundwater vulnerability in an assessment of CSAs for loss to groundwater. In more recent work in agricultural catchments in the south-west and south-east of Ireland, Mellander *et al.* (2016) highlighted the importance of considering lags between P application and loss to groundwater for reviewing the efficacy of mitigation measures in groundwater-dominated catchments. Differences in the mobilisation potential of different soils related to iron (Fe) content and variations in the pathways of delivery to surface waters were shown to be of importance. However, avoiding future contamination of groundwater is a potential priority because groundwater contributes to the baseflow in rivers in summer, at which time the conditions for algal growth are optimal. Even if runoff P losses can be mitigated, P concentrations in streams and rivers may continue to increase because of groundwater legacy P and release from fluvial sediment sources (McDowell *et al*., 2015).

Studies have also investigated the contamination of surface water and groundwater by microbial pollution (Bacci and Chapman, 2011; Hynds *et al*., 2012; 2014a,b); however, the confirmation of a direct link between these

contaminants and agricultural sources in Ireland and elsewhere requires more evidence (Oliver *et al*., 2016). Bacci and Chapman (2011) investigated the microbial contamination of groundwater drinking water supplies in west Cork and found that one-quarter of the 75 boreholes sampled were contaminated with thermotolerant coliforms. MST has recently been used to differentiate between different sources of microbial contamination. For example, Flynn *et al*. (2016) used the technique in the Mattock catchment in Ireland to assess the effectiveness of a winter period without slurry spreading (i.e. a "closed period"). Both inside and outside the closed period, the authors recorded evidence of fresh faecal indicator organism (FIO) inputs. However, by employing MST techniques, they demonstrated that bovine faecal waste dominated water samples outside the closed period, while human faecal waste from onsite wastewater treatment plants (WWTPs) dominated during the closed period (Flynn *et al*., 2016).

Agricultural land in Ireland has been significantly drained in an attempt to improve the productivity of agricultural soils (Burdon, 1986). This has also been carried out elsewhere and has altered the hydrology of the soil and, consequently, catchment hydrology (Moss, 2008a; Harrigan *et al*., 2013), but the evidence to demonstrate the impacts of this drainage in Ireland is limited. Elsewhere, Poff *et al.* (1997) and Blann *et al.* (2009) have detailed the impact of changes in flow magnitude, frequency, timing, duration and rate of change on river ecology; impacts range from life cycle disruption, loss of access to habitats, sedimentation, scouring and changes in plant cover. Harrigan *et al.* (2013) explored the drivers of hydrological change in the 1970s and 1980s in the Boyle catchment, County Roscommon, and found that the main drivers of changes in annual mean flow and high flows were widespread land drainage and arterial drainage in the catchment during this time. Similarly, in Northern Ireland, Wilcock and Essery (1991) showed that an agricultural channelisation and drainage scheme on the River Main, inflowing to Lough Neagh, increased peak SS loads with higher flows and led to a loss of river habitats for fish nurseries with meander shortening. A recent review of Eflows in Ireland by Webster *et al.* (in review) identified land drainage and arterial drainage related to agriculture as key pressures on Eflows in Irish rivers. Maintaining Eflows is crucial to achieving the objectives of the WFD, as many aquatic organisms are sensitive to changes in flow regimes. The importance of hydrological flow was highlighted by the study of Moorkens and Killeen (2014). These authors found that the optimal flow regime for the freshwater pearl mussel (FPM) entails relatively high velocities at low flows, as the FPM has adapted to stable substrates that are kept clean of fine sediment by relatively high velocities. Rooney *et al.* (2013) found that active brook lamprey redds were predominantly located in sections of rivers with a water depth of 10 cm or less and a high velocity (0.20 m/s).

3.2 Variability in the Relationship

Although evidence exists to link agricultural practices with chemical water quality and, in some cases, ecological water quality, a common theme both in Ireland and elsewhere is the significant variability in this relationship (Doody *et al*., 2016). Allan (2004) found some studies suggesting that river ecology would not be affected until agricultural land occupied more than 30% of the catchment area, while in other studies, this was reported to be 80% of the catchment area. Barry and Foy (2016) demonstrated a significant variation in the relationship between soluble reactive P concentrations in rivers and agricultural intensity in headwater catchments in Northern Ireland, and Foy and Kirk (1995) demonstrated similar variability in the ecological response in the same catchments. In the study by Foy and Kirk (1995), at an agricultural intensity of 1 dairy cow equivalent/ha, some catchments had a Fisheries Ecosystem Class of 2, while others had a Fisheries Ecosystem Class of 6, which is close to pristine. The variability in the catchment response described in these studies highlights that the equilibrium between an aquatic ecosystem and the surrounding catchment can result in two catchments with the same agricultural intensity having very different outcomes in terms of impacts (Doody *et al*., 2016). Jordan *et al.* (2012) compared two intensive grassland catchments and found that the lowest intensity catchment, as indicated by a lower P loading and percentage of index-4 soils, had the highest rate of P export because of large runoff volumes generated by the poorly drained soils. Doody *et al.* (2012a) highlighted the impact of agriculture on water quality in the Lough Melvin catchment by demonstrating that P concentrations increased from 19 to 29 μg/L in 10 years, despite agricultural intensities remaining at almost 1 livestock unit/ha during this period. Because of the predominance of soils with impeded drainage, there had been an over application of slurry to drier fields resulting in the development of "hotspots" of soil P in the catchment (Doody

et al., 2012). Although Donohue *et al.* (2006) identified a threshold value of 38% pasture, above which there was a significant decline in the probability of a site having a good ecological status, there was substantial variability, which increased above the predicted threshold value.

The variability in catchment response described in these studies arises from differences in the physical, chemical and biological properties that influence the equilibrium between an aquatic ecosystem and the surrounding catchment (i.e. the catchment buffering capacity) (Doody *et al.*, 2016). The cumulative impact of heterogeneity in soils, transport pathways and the response of aquatic ecosystems means that our ability to predict the areas in which the dual objectives of agriculture and good water quality are achievable is still poor (Moss 2008a; Withers *et al.*, 2014). For example, Jahangir *et al.* (2012a) found variation in the denitrification potential of subsoils depending on soil horizon and carbon addition; this has implications for measurements of nitrate concentration in groundwater. Work by Fenton *et al.* (2011b) demonstrates that sites with a relatively high subsoil saturated hydraulic conductivity (K_{sat}) have a lower denitrification potential than sites with low K_{sat} values. Likewise, Tedd *et al.* (2014) found that pathway factors controlling the rate of movement of groundwater through the soil and bedrock system were more important than pressure factors for determining nitrate concentration in groundwater, and, in this context, catchment characteristics such as the proportion of poorly drained soils, karstic flow regimes and regionally important bedrock aquifers were significant determinants. Fenton *et al.* (2011a) developed a model of lag times related to achieving nitrate threshold values in Irish aquifers; according to this model, changes in nitrate concentrations depended on the depth of the unsaturated zones, aquifer thickness and specific yield, and these factors resulted in variations in the estimates of the response times of groundwater to nitrate mitigation measures of between 7 and 16 years. In a study of the Thames catchment in England, Howden *et al.* (2013) predicted a 30-year lag time in the response of groundwater to contemporary mitigation measures; however, further research is required to determine if such long lag times occur in Irish groundwater systems.

Similar uncertainty in the export of P has been reported because of the variability in soil properties (Mellander *et al.*, 2016) and hydrological factors (Jordan *et al.*, 2012). For example, Mellander *et al.* (2012a, 2013) demonstrated the attenuation capacity of calcium-rich soils overlying a karst aquifer, which resulted in a low background concentration of P in the groundwater despite relatively intensive agriculture and a high proportion of high-P soils and high hydrological connectivity within the karst aquifer. Mellander *et al.* (2012b) developed the Loadograph Recession Analysis (LRA) method which highlighted the importance of soils and hydrogeological properties for predicting the spatial and temporal variation in the hydrological pathways that dominate nutrient export in catchments. In a detailed study of Irish soils, Daly *et al.* (2015) demonstrated that soil buffering capacity was significantly impacted by pH and extractable aluminium (Al); these authors identified an extractable Al change point, above which soil buffering capacity increased linearly. Daly *et al.* (2015) concluded that combining data on soil buffering with existing soil P test data would increase the ability to predict P availability in soils and the risk posed to water quality. Work elsewhere has highlighted the necessity to include a measure of soil P buffering capacity in routine soil analysis procedures, in order to better determine soil P release to both crops and runoff water (van Rotterdam *et al.*, 2012; Forrester *et al.*, 2015).

Uncertainty with regard to the response of aquatic ecosystems to changes in agricultural practices has also affected the ability to predict outcomes. Murphy *et al.* (2015) examined the response of mitigation measures across the nutrient transfer continuum (Haygarth *et al.* 2005) and found that uncertainty increased from source to transport to impact on receiving water bodies. Although there were clear improvements in a number of source pressure metrics (e.g. farm gate balance, P use efficiencies) and some evidence of a reduction in P concentration in quickflow pathways, there was no clear improvement in observed stream biological quality (Murphy *et al.*, 2015). The work of O'Dwyer *et al.* (2013) highlights the importance of understanding water body sensitivity to agricultural practices. They used sediment cores to reconstruct the changes in the trophic status of a lake and found evidence of improvement in the trophic status despite further intensification of agriculture in the catchments. This suggests that the recent mitigation strategies related to the Nitrates Action Programme in the catchment had been effective in reducing P inputs to the lake (Dywer *et al.* 2013). Other lake sediment studies have also highlighted the complexity of the response of water bodies to agricultural intensification (Jordan *et al.* 2002; Taylor *et al.* 2006). Rippey *et al.* (1997) found that acceleration of eutrophication in a

small lake in Northern Ireland between 1973 and 1979 was the result of a reduction in the flushing time of the lake during a dry period, combined with an increase in the internal P loading of the lake. Ní Longphuirt *et al.* (2016) compared the impact of a hydrological regime on macroalgae biomass production in two estuaries in Cork with similar nutrient inputs from the surrounding catchments. The predicted reduction in biomass was much greater in the Blackwater estuary than the Argideen estuary because of the shorter residence times and greater influx of marine water into the Argideen estuary.

Understanding the variability in the response of aquatic ecosystems to changes in land use and management is key to setting realistic targets for recovery. Although determining ecological reference conditions for water bodies is a well-established step in setting water quality targets in many countries (Dodds and Oakes, 2004; Bouleau and Pont, 2015), to date these have been approximations that fail to account for the significant variability within and between catchments (Moss, 2008b; Page *et al*., 2012; Bouleau and Pont, 2015). Although current estimates of reference conditions are valuable, a greater degree of certainty, on a catchment-specific basis, is required (Hawkins *et al*., 2010), in order to establish more accurately the boundary values established between categories of ecological states (Stoddard *et al.*, 2006). This, in turn, will help to refine our estimates of the relationship between catchment pressures and impacts in aquatic ecosystems, and the threshold values at which these impacts occur (Stoddard *et al.*, 2006). However, restoration and degradation trajectories of aquatic ecosystems may differ (Harris and Heathwaite, 2012); this has consequences for predicting the recovery of water bodies from good to high status. Jarvie *et al.* (2013) suggested that, in some cases, baselines have shifted so that many affected water bodies may never return to reference conditions, even if pressures are reduced to below the threshold values at which degradation occurred. In contrast, Vaughan and Ormerod (2012) found that freshwater ecological communities in rivers were resilient to long-term anthopogenic change and could respond if such pressures were reduced. This variability suggests that the right balance of interventions to improve water quality and ecological status will need to be highly site specific and based on patterns of nutrient delivery, biological response and recovery trajectories in different types of water bodies (Withers *et al.* 2014). The identification of any "tipping points" and the factors that modulate them is necessary to enable appropriate actions to be taken, in order to ensure that such threshold values are not exceeded.

In addition to the biophysical heterogeneity of catchments, social heterogeneity between farmers and farming systems adds a further layer of uncertainty to the prediction of the impacts of agriculture on aquatic ecosystems. Although, historically, sociological research related to mitigating the impact of agriculture on water quality has not received the same attention as the biophysical sciences, the recent increase in the number of Irish publications in this research area is an acknowledgement that stakeholder participation and behavioural change are vital components of sustainable agriculture. Irvine and O'Brien (2009) detailed the challenges related to stakeholder participation in the implementation of the WFD and concluded that legislation works only if it is technically feasible and largely has buy-in from affected stakeholders. Stakeholder participation and understanding the factors that influence farmers' behavioural change are key components of testing the effectiveness of the ND measures in the Teagasc ACP (Buckley, 2012; Buckley *et al*., 2012; Teagasc, 2013). In addition, a recent project entitled "Qualitative Analysis of Farmer Behaviour" further highlights the focus now being placed on understanding the factors controlling Irish farmers' management decisions (Macken-Walsh *et al*., 2010). During the development of the Lough Melvin catchment management plan, Doody *et al*. (2009a) and Schulte *et al*. (2009) successfully engaged with farmers in the development of a suite of agri-environmental measures to mitigate the impact of agriculture on water quality. The aim was to develop a list of scientifically robust measures that incorporated the views and requirements of local farmers (Doody *et al*., 2009a). Other work in this area by Doody *et al*. (2009b) led to the development of a method for stakeholder participation in the selection of indicators of sustainable development, which involved engagement with both urban and rural populations. Van Rensburg *et al*. (2009) investigated the factors controlling farmers' participation in the Rural Environmental Protection Scheme (REPS) in Ireland, and highlighted that farmers involved with REPS were more likely to be open to behavioural change in relation to environmental protection than farmers not involved with REPS. Visser *et al*. (2007) explored farmers' attitudes towards the conservation of turloughs in Ireland and, similarly, Augustenborg *et al*. (2012) conducted a survey of Irish farmers to elicit their

opinions on energy crop production and to characterise potential adopters of energy crop cultivation. It is clear from the research carried out to date that there is a need to identify the inherent barriers to the implementation of mitigation options. However, the attribution of any one explanatory driver over another is likely to be flawed, as there appears to be substantial heterogeneity and complexity in what motivates a farmer to engage with environmental interventions to mitigate the deleterious impacts of current farm management.

3.3 Improving Predictions of Catchment Responses

Catchments are multifaceted socioecological systems and there are fundamental limits to society's ability to understand the full extent of their complexity (Harris and Heathwaite, 2012. However, increasing the understanding of catchment heterogeneity and its control on the pressure–impact relationship between agriculture and water quality will enable more accuracy with regard to the prediction of outcomes, the establishment of realistic water quality targets and the targeting of mitigation measures, and will inform the monitoring programmes that aim to evaluate mitigation strategies. Ultimately, this understanding will increase the accuracy of predicting the areas in which sustainable agricultural intensification is achievable.

With the requirement to achieve good ecological status in all water bodies by 2027, research has attempted to disentangle the biophysical and chemical factors that constrain/maintain the status of water bodies. These factors include geomorphology, environmental drivers (temperature, light), chemical thresholds, recolonisation and hydrology, all of which vary on a catchment- and site-specific basis (Smith *et al*., 2009; Jarvie *et al*., 2013). Although mitigation measures may reduce, for example, nutrient inputs to below the required threshold values for recovery, other factors (e.g. hydrology) may constrain recovery and/or the direction in which it takes (Palmer *et al*., 2007; Page *et al*., 2012). The unknown future impacts of climate change, including changes in rainfall patterns and temperature, also need to be considered (Withers and Jarvie, 2008; Jennings *et al*., 2009). For example, to determine the threshold values for nutrients in water bodies, consideration needs to be given to the timing and bioavailability of different sources so measures can be targeted to the areas that contribute the greatest loads of bioavailable nutrients to water bodies. If current chemical water quality standards are achieved but there is no corresponding improvement in ecological water quality, this may be because of a lag in recovery time, a hysteresis effect, other factors that constrain recovery and/or a site-specific threshold value (Harris and Heathwaite, 2012, all of which will vary significantly between catchments. In addition, mismatch between chemistry and ecology in water bodies may also result if the frequency of chemical monitoring in catchments is insufficient to capture detrimental transfers of contaminants, as current regulatory monitoring is predisposed to miss storm flow (diffuse) transfers in hydrologically flashy catchments (Cassidy and Jordan, 2011). If these factors can be identified, then a more accurate assessment of the pressure–impact relationship of agricultural and aquatic ecosystems could be developed and this would enable more accurate predictions of recovery/deterioration trajectories. This, in turn, would facilitate a greater understanding of the limits of agricultural intensification in individual catchments.

There has been a significant investment in evaluating the effectiveness of mitigation measures in Ireland and elsewhere (Fealy *et al*., 2010; Jordan *et al*., 2013; McGonigle *et al*., 2014; Schoumans *et al*., 2014). However, this has proved a difficult task because of uncertainties about ecological recovery trajectories, the continued impact of historical land use practices and the complexity of generating evidence unequivocally to link cause and effect. Melland and Jordan (2015) predicted a lag time of 4–19 years between the implementation of water quality mitigation strategies and the response of catchments. There was a strong correlation between lag time and catchment scale (Melland and Jordan, 2015), which is likely to be due to a corresponding increase in complexity and heterogeneity with increasing scale. Ní Longphuirt *et al.* (2015) demonstrated changes in water quality as a result of improved land use practices, as indicated by a 17% and 20% reduction in N and P inputs, respectively, to the Blackwater estuary in the past 10 years. There was a corresponding improvement in measured P concentrations, chlorophyll levels and dissolved oxygen saturation in the estuary. However, despite the reduction in N inputs, the N concentration in the estuary remained stable during the same period, highlighting the complexity of measuring the response of water bodies to a reduction in land use pressures (Ní Longphuirt *et al*., 2015b).

Although numerous studies have demonstrated the effectiveness of a wide range of mitigation measures

at local scale (Newell-Price, 2011; Schoumans *et al*., 2011), there is still a deficit in the knowledge on how to accurately transfer estimates of the effectiveness of measures implemented under different conditions (i.e. slope, soil type, farm type, etc.) (Kleinman *et al*., 2015). This deficit in knowledge will be reduced only if there is an improvement in the understanding of the factors that control a catchment's response to agricultural practice. Currently, across Ireland, mitigation measures to control agricultural pollution are predominately implemented at farm scale, and very little consideration is given to intra- and inter-farm and catchment variation (Doody *et al*., 2012a). The generic approach to the implementation of mitigation strategies that has largely prevailed in Ireland has reduced the cost effectiveness of measures, as little consideration is given to the impact of, inter alia, soil type, topography, farming system, hydrology and the past agricultural management that has left a legacy of stored nutrients in the catchment (e.g. groundwater N and soil TP). Basing the implementation of measures on catchment-specific factors and water body sensitivity would increase cost-effectiveness and the likelihood of achieving the targets of the WFD (Page *et al*., 2012; Withers *et al*., 2014). Addressing these factors may also help to facilitate farmer engagement, as greater consideration is given to local farm and environmental conditions and to ensuring best use of limited financial and human resources (Schulte *et al*., 2009).

In addition, a better understanding of heterogeneity in the relationship between agricultural practices and impacts on aquatic ecosystems is also required to determine if the objectives of the WFD can be achieved within the context of climate change and the predicted growth of the agricultural sector in Ireland (Food Harvest 2020 and Food Wise 2025) (Doody *et al*., 2016). Climate change is likely to impact directly on aquatic ecology, and indirectly on the sources and transport pressures that are related to contaminant export from agriculture to water, adding an additional layer of complexity to the relationship between agricultural practice and ecological impact (Withers and Jarvie, 2008; Jeppesen *et al*., 2009). Recently, there has also been an increased focus on the delivery of ecosystem services from agricultural land, with the protection of biodiversity as a key driver (Dodds *et al*., 2013). The equity of payments for the delivery of aquatic ecosystem services will be significantly enhanced if the link between land use activity and impacts on aquatic ecosystems can be established and can account for biophysical and human heterogeneity in catchments.

The priority research areas identified during the systematic and participatory process utilised during this project (section 2.3) are discussed in Chapter 4. The aim is to explain why additional research is required in each of these areas and how such research would contribute to reducing uncertainty with regard to the pressure–impact relationship between agriculture and aquatic ecosystems in Ireland. At the end of each subsection in Chapter 4, a summary of key knowledge gaps is presented.

4 Research Gaps

4.1 Transportation and Attenuation

4.1.1 Subsurface hydrological pathways

One of the key challenges with regard to improving the accuracy of estimates of the pressure–impact relationship between agriculture and water quality is the limited understanding of the transformations and attenuation of contaminants that occur during transport from sources to points of impact in water bodies (Blöschl and Zehe, 2005; Troch *et al.*, 2009 Mellander *et al.*, 2012). Attenuation and transformation occurs in both surface and subsurface transport, and affects both the amount and the form of the contaminant delivered and the time taken for its impact to be observed (i.e. lag time) (Fenton *et al.*, 2011a; Jarvie *et al.*, 2014). Recent work by Wilson *et al.* (2016) indicates the importance of groundwater inputs both to Lough Ennell and to the rivers draining into the lough, and highlights the importance of understanding groundwater–surface water interactions.

This section addresses subsurface transport and attenuation, and surface transport is discussed separately in section 4.1.2. Subsurface flow is understood to include interflow (any lateral subsurface flow occurring above the water table) and shallow and deep groundwater flow. Recent research undertaken as part of the EPA-funded Pathways Project (EPA, 2016a) emphasises the variability in subsurface processes across Ireland. Although in many areas a four-pathway model of water-borne contaminant transport is applicable [accounting for contaminant transport by (1) overland flow, (2) interflow, (3) shallow groundwater and (4) deep groundwater], in areas underlain by poorly productive aquifers (>60% of the island of Ireland) artificial drainage and the transition zone between soil and bedrock are crucial pathways during both low flows and storm events in wet antecedent conditions. Until recently, these pathways have been largely overlooked (EPA, 2016d), and the transport and attenuation of contaminants through these zones are, in comparison with other pathways, important areas for further research.

Traditionally, nitrate contamination of groundwater has received greatest attention in Ireland (e.g. Fenton *et al.*, 2011a,b; Jangahir *et al.*, 2012a,b; Huebsch *et al.*, 2013, 2014; Richard *et al.*, 2015), perhaps because of the large number of groundwater aquifers, particularly in the south-western and south-eastern river basin districts, that have shown high levels of nitrate contamination during assessments as part of WFD monitoring (EPA, 2010, 2015). Aquifer and overburden properties, and hydrology exert a strong influence on the transport and attenuation of nitrate in groundwater. For example, Huebsch *et al*. (2014) examined how hydrological conditions, N availability, soil properties and karst features controlled nitrate dilution and mobilisation in a karst spring on a grassland farm in Ireland. They reported that the mobilisation and/or dilution of nitrate, which controlled the temporal variation of nitrate concentrations in the karst spring, was dependant on the extent to which nitrate accumulated in the overlying soil and unsaturated zone. Tedd *et al*. (2014) evaluated the pressures and pathways that affect the spatial distribution of groundwater nitrate in south-eastern Ireland. These authors found that the proportion of poorly drained soils, arable land, karstic flow regimes, regionally important bedrock aquifers and vulnerable areas within the zone of contribution to the monitoring points were significantly related to groundwater nitrate concentrations. Soil type and pathways were of greater influence than source pressures in determining the spatial distribution of nitrate. Jahangir *et al.* (2012) investigated the impact of hydrogeochemical factors on the occurrence of nitrate originating from agricultural sources, and reported a significant positive relationship between its occurrence and soil redox potential and hydraulic conductivity. Huebsch *et al.* (2013) demonstrated an improvement in nitrate concentration in groundwater over an 11-year period; their results indicate that the impact of mitigation strategies on groundwater quality could be observed within 1–2 years. However, Fenton *et al.* (2011a) found that predicted lag times in recovery of groundwater and groundwater-dominated surface water bodies ranged from 7 to 21 years, depending on factors such as the thickness of the unsaturated zone, the thickness of the aquifer and the specific yield of the aquifer. Howden *et al.* (2011) predicted a lag time of more than 30 years in one chalkland groundwater catchment in England due to long travel times to the aquifer, although much shorter travel times are considered appropriate for

Ireland. These findings on the attenuation and transformations of N demonstrate that there are a significant number of hydrological and hydrogeochemical factors that determine the concentrations and lag times in groundwater bodies. They also indicate that, given the spatial heterogeneity in soil cover, weathering patterns and aquifer structure, attenuation and lag times, that is, the time between the implementation of mitigation measures and the observation of positive change, may be very difficult to predict.

Although overland flow is often considered to be the main pathway of P export, interflow (including artificial drainage) and groundwater flow are increasingly recognised as important pathways, which affect the lag time and the forms of P exported from agricultural land (Jarvie *et al*., 2014; EPA, 2016a; Mellander *et al*., 2016). Holman *et al*. (2010) reported that 51 groundwater bodies, covering 22% of Ireland, had median soluble reactive P concentrations exceeding background levels of 20 μg/L, indicating anthropogenic impacts on water bodies; similar results were previously reported by Kilroy (2001). These data highlight the importance of groundwater as a receptor and pathway for P, and the affect this may have on the recovery of both surface and groundwater bodies.

Mellander *et al.* (2016) investigated P transport via groundwater in two contrasting catchments with similar inorganic P reserves in the soil. The catchment dominated by well-drained Al-rich soils had a lower soluble total reactive P (TRP) concentration in the groundwater and, consequently, TRP concentrations were threefold lower in the receiving river than in the catchment dominated by well-drained Fe-rich soils. These differences were due to the Al-rich soils and their ability to reduce the solubility of P and thereby reduce P transfer to groundwater. Mellander *et al.* (2016) also reported that, despite overall low P fluxes in both catchments, groundwater contributed 50% or 59% of the P load transported during the winter months, and concluded that the lag times associated with these contributions should be factored into expectations with regard to achieving the targets of the WFD. In another study, Mellander *et al.* (2012, 2013) also demonstrated that the attenuation capacity of calcium-rich soils overlying a karst aquifer resulted in a low background concentration of P in the groundwater, despite relatively intensive agriculture and a high proportion of high-P soils and high connectivity within the karst aquifer. Outside Ireland, a study by Jarvie *et al.* (2014) demonstrated the retention of up to 90% of the annual soluble P flux within a karst aquifer, suggesting that karst aquifers can act as more significant P sinks than previously thought. In contrast, there are catchments with poorly retentive soils that show significant P levels, even in relatively deep aquifers.

The research outlined above provides strong evidence for the importance of the attenuation and transformation of N and, to a lesser extent, P in subsurface hydrological pathways, and the important role this plays in the response of mitigation strategies. However, elucidating how to transfer the relationships observed on a site-specific basis to predictions made at waterbody scale remains a significant challenge for research (Cassidy *et al*., 2014). Heterogeneity increases with scale, and there is still significant uncertainty in terms of accounting for this heterogeneity in predictions of attenuation and transformation in subsurface pathways, particularly if catchment-specific data are limited. In addition, other than for N and P, little is known about the transformation and attenuation of contaminants such as pesticides, microbes and endocrine disrupters. For example, although McManus *et al.* (2014) investigated the occurrence of 2-methyl-4-chlorophenoxyacetic acid (MCPA) in groundwater in Ireland, there is currently limited information available on its transformation and attenuation under different hydrogeological conditions.

Cassidy *et al.* (2014) outlined the causes of the high degree of spatial heterogeneity in groundwater systems in hard rock regions of Ireland, and the challenges this poses for accurately predicting groundwater flows and contaminant transport. They advocated the need for an enhanced conceptual understanding of aquifer structure in Irish catchments, in order to aid the correct interpretation of both surface water and groundwater monitoring programmes. Comte *et al.* (2012) presented an approach for characterising the typology of hard rock groundwater bodies that addressed the structural heterogeneity required to accurately predict groundwater flows and their contribution to surface water. Challenges with regard to accurate predictions of groundwater contamination and surface water interactions also exist in the context of delineating the areas that contribute to groundwater bodies; for example, Coxon and Drew (2000) highlighted this complexity in karst regions in the west of Ireland. In the context of transformation and attenuation in subsurface pathways, the challenge of improving our understanding of the pressure–impact relationship between agriculture and aquatic ecosystems is how to link the characterisation

and prediction of the heterogeneity in groundwater systems with transformation, attenuation and sources of agricultural contaminants.

Key knowledge gaps

- Incorporating spatial heterogeneity in soil cover, weathering patterns and aquifer structure into the prediction of attenuation times and lags in response to mitigation measures.
- Improving predictions in the transport and attenuation of contaminants through artificial drainage and the transition zone between soil and bedrock in comparison with other pathways.
- Elucidating the importance of groundwater as a receptor and pathway of P and the impact this may have on the recovery of both surface water and groundwater bodies.
- Transferring the relationships observed on a site-specific basis to predictions made at water body scale.
- Other than for N and P, little is known about the transformation and attenuation of other contaminants, such as pesticides, microbes and endocrine disrupters, in groundwater.

4.1.2 Natural water retention measures

Over the past 100 years, natural water retention features in the landscape, such as wetlands, flood plains and the soil matrix, have been degraded as a result of agricultural intensification (Moss, 2008a). This has reduced the natural capacity of catchments to buffer the impact of agriculture on water quality, both in the context of contaminant export and the Eflows of water bodies.

Natural water retention measures (NWRM) are defined as:

> **multi-functional measures** that aim to protect water resources and address water-related challenges by **restoring or maintaining ecosystems** as well as **natural features and characteristics** of water bodies using **natural means and processes**. The main focus of applying NWRM is to **enhance the retention capacity of aquifers, soil, and aquatic and water dependent ecosystems** with a view to improve their status. The application of NWRM supports **green infrastructure**, improves the **quantitative status of water bodies** as such, and reduces the **vulnerability to floods and droughts**. It positively affects the **chemical and ecological status of water bodies** by restoring natural functioning of ecosystems and the services they provide. The restored ecosystems contribute both to **climate change adaptation and mitigation** (EC, 2014).

Variability in the location and extent of NWRM in catchments may account for some of the observed variability in the pressure–impact relationship between agriculture and aquatic ecosystems. Therefore, the capacity to install/re-establish NWRM in optimum locations within catchments will support future intensification of agriculture and deliver many other ecosystems services, such as flood protection, Eflows and water quality. However, research on optimising NWRM, such as wetlands, riparian zones, re-meandering and ponds, in Irish catchments has been limited. Irish research has predominantly focused on constructed wetlands and their ability to reduce the impact of agricultural point sources of contaminants (Dunne *et al*., 2005; Mustafa *et al*., 2009; Forbes *et al*., 2011; Healy and O'Flynn, 2011; O'Neill *et al*., 2011). Healy and Flynn (2011), in their review of constructed wetlands in Ireland, reported that, on average, BOD, SS and NH_4-N decreased by 98%, 94% and 88%, respectively, if wetlands were used to treat dairy soiled water. Phosphate removal is highly variable depending upon the operation, the substrate and the loading rates (Healy and O'Flynn, 2011). O'Neill *et al.* (2011) reported that a five-pond wetland system in County Antrim retained, on average, approximately 66% of P from dairy soiled water over a 14-month period, and the percentage retention varied with inflow volume and air temperature. The utility of NWRM for attenuating other contaminants, such as pesticides, has not been assessed. In a systematic review of the effectiveness of wetlands to mitigate N and P losses, Land *et al.* (2013) reported that wetland effectiveness was dependent on loading rate, average annual air temperature and wetland areas, and that restored wetland was significantly less efficient than constructed wetlands.

More recently, Shore *et al.* (2016a,b) investigated the use of agricultural drainage ditches to buffer the transfer of P, and have highlighted the need to maintain the channel banks and remove sediment in order to maximise the potential of ditches and attenuate both soluble and particulate P. Although riparian buffer strips have often been cited as an effective measure for reducing

contaminant export from agriculture, and have been incorporated into regulation and agri-environmental schemes, there has been limited research aimed at evaluating them in the context of Irish agriculture. Ó hUallacháin (2014) reviewed the effectiveness of buffer strips for the protection of FPM in Ireland and concluded that they were a cost-effective method of reducing agricultural impacts if targeted and managed correctly. However, assessments of cost effectiveness have largely been based on research from the UK and elsewhere, and the accuracy of these assessments is limited by the lack of transfer functions to inform how effectiveness will vary on the basis of site-specific characteristics.

As for wetlands, the effectiveness of buffer strips varies significantly depending on a wide range of factors, such as soil type, vegetation, topography and width (Stutter *et al*., 2012; Darch *et al*., 2015). As a result, there is a high degree of uncertainty in predictions of their effectiveness on a site-specific basis and how this translates to the catchment scale. For example, Bergfur *et al.* (2012) reported no improvement in water quality or macroinvertebrate scores in the Tarland catchment in Scotland after 8 years of, and a significant investment in, riparian buffer strips throughout the catchment. It was concluded that this lack of effect may have been because the low frequency sampling failed to detect changes in water chemistry and/or because of a lag time in response. However, they also indicated that there were significant uncertainties related to the spatial extent and cost benefits of the buffer strip programme. In Ireland, Buckley *et al.* (2012) examined farmers' willingness to adopt riparian zones on their farm and reported that 53% of farmers preferred not to adopt riparian zones because of the potential impacts on production and nuisance effects. Overcoming these barrier is a key step towards improving the effectiveness of many NWRM. Management is a key factor in the effectiveness of NWRM in order to prevent them from becoming sources, rather than sinks, of contaminants, especially because they are often located close to water bodies.

Although buffer strips, wetlands and ditches are key water retention features in the landscape, the EU NWRM network (http://www.nwrm.eu/) developed a list of 13 NWRM contained within the agricultural landscape and a further 14 that are related to hydromorphological features, such as floodplains, wetlands and river habitat restoration, which are also relevant to controlling the impact of agriculture on water quality. NWRM of particular relevance to grassland agriculture in Ireland include managed pastures, controlled traffic farming and reduced stocking rates, all of which contribute to maintaining good soil structure, thereby reducing runoff volumes and associated contaminant export. The management of pastures in terms of trafficability and grazing has received attention in Ireland in an number of studies (e.g. Herbin *et al*., 2011; Piwowarczyk *et al*., 2011; Doody *et al*., 2014c; Vero *et al*., 2014; McConnell *et al*., 2015; Tuohy *et al*., 2015). Many authors have identified a negative relationship between grazing intensity and soil hydraulic conductivity, which is reflected in an increase in overland flow volume following treading (Drewry and Paton, 2000; Drewry, 2003; Pietola *et al*., 2005). Doody *et al*. (2014c) examined the impact of different grazing intensities on soil structure and how this affected the nutrient export from grazed grassland. They reported a significant relationship between the time taken to generate overland flow (as an indicator of soil infiltration rate) and the resulting load of TP exported. There was a 62% increase in the volume of overland flow generated from the grazed plots compared with the ungrazed plots (Doody *et al*., 2014c). This increase in volume resulted in a corresponding average increase of 60% in the quantity of TP, total oxidised N (TON) and sediment exported from the grazed plots.

Vero *et al*. (2014) investigated the impact of a tractor and slurry tanker on the structure of three soils with different drainage classes, and at four different soil moisture deficits (SMDs). They reported that the SMD at the time of trafficking had a significant impact on the extent of the changes in soil bulk density, and that a SMD of 10 mm was the threshold for safe slurry spreading in terms of preventing compaction. The effectiveness of pasture soils as a natural water retention measure will be significantly affected by the installation of subsurface drainage. Although subsurface drainage improves trafficability and agricultural production, it also decreases the retention of water in the soil, thereby reducing the potential attenuation of contaminants and changing catchment hydrology. Tuohy *et al.* (2016) found that the type of drainage used affects the response time, peak and duration of flow. Future drainage schemes related to predicted agricultural intensification need to carefully consider the type and location of drains in order to balance the need for increasing natural water retention in catchments and the agricultural production goals.

Although the focus in Ireland on NWRM has largely been on buffering contaminant loss from land use, a greater focus on the multiple benefits of these features would allow a more robust evaluation of cost effectiveness. For example, improving soil quality would mitigate soil compaction, thereby increasing crop production, while increasing infiltration would reduce both nutrient export and the flashiness of soils (Doody *et al*., 2014c). The multifunctionality of NWRM will increase the resilience of catchments and ecosystems to future climate change and/or land use changes, thereby helping to mitigate extreme flows, securing drinking water sources and protecting aquatic ecosystems. If the implementation of NWRM is managed in a co-ordinated and targeted manner at catchment scale, such measures will provide a "win–win" outcome for both water quality and agriculture by reducing the extent of land flooding, preventing water shortages and buffering the export of contaminants from agriculture. Other NWRM, such as floodplains and wetlands, will also enhance biodiversity and the resilience of ecosystems to climate change. The challenge is to identify where, when and in what combination specific measures will optimise water retention within a catchment and deliver multiple ecosystem services.

Key knowledge gaps

- The installation/re-establishment of NWRM should be targeted to optimum locations within catchments to support future intensification of agriculture and deliver many other ecosystems services, such as flood protection, Eflows and water quality.
- The utility of NWRM for attenuating contaminants, such as pesticides, endocrine disputers and emerging contaminants, should be evaluated.
- Transfer functions to inform the cost-effective evaluation of NWRM based on site-specific characteristics should be developed.
- The evaluation of future land drainage schemes in the context of the need to balance the delivery of multiple ecosystem services (e.g. flooding, Eflows) and agricultural production is needed.
- It is important to identify where, when and in what combination specific measures will optimise water retention within a catchment and deliver multiple ecosystem services.

4.2 Water Body Response

4.2.1 Multiple stressors

Catchments are complex systems, and land use intensification has altered the balance and prevalence of the multiple geomorphological, hydrological, biochemical and ecological stressors that affect aquatic communities (Moss 2008a; Ormerod *et al*., 2010). The pressure–impact relationship between agriculture and water quality has been demonstrated on the basis of metrics such as percentage pasture and decline in ecological water quality (Donohue *et al*., 2006), changes in dairy cow equivalent/ha, increases in soluble reactive P in rivers (Barry and Foy, 2016) and declines in Fisheries Ecosystem Class (Foy and Kirk, 1995); however, these are indicators of the accumulated impact of multiple stressors related to agricultural sources. Although this relationship has been demonstrated, these metrics do not provide sufficient evidence for the development of effective measures to mitigate the cumulative impact of multiple stressors (Hering *et al*., 2015). Elucidating the effects of land use on aquatic ecosystems is not straightforward, with each catchment having a unique "fingerprint" of aquatic stressors, making the effects of management interventions extremely difficult to predict (Harris and Heathwaite, 2012). It is this uncertainty that necessitates further research in order to disentangle the physical, biological and chemical factors that constrain/maintain the status of water bodies, including geomorphology, environmental drivers (temperature, light), chemical thresholds, recolonisation and hydrology (Hering *et al*., 2015; Jackson *et al*., 2016). The synergistic and antagonistic impacts of stressors are determined by the types of stressors, the catchment characteristics and water body sensitivity, all of which exhibit significant heterogeneity within and between catchments (Hering *et al*., 2015; Jackson *et al*., 2016). However, the ability to identify the key factors that constrain/impact aquatic ecology in different catchments will result in a much higher probability of achieving restoration targets and the more cost-effective targeting of measures (Statzner and Beche, 2010; Hering *et al*., 2015). For example, the lack of biological recovery in some water bodies could mean that the current target threshold values for pollutants are inaccurate and/or that other stressors, such as hydromorphology, are constraining recovery.

Research, nationally and internationally, has provided a relatively good understanding of the processes of P transfer. However, in the context of linking catchment characteristics, ecosystem resilience and interactions with other stressors in order to predict outcomes on a catchment-specific basis, there are still large gaps in our understanding of P dynamics in catchments (Withers *et al*., 2014). In the context of P, our current understanding suggests that catchments with the highest probability of mitigating the impact of agricultural P on aquatic ecosystems are those with highly productive, deep, P-retentive soils; a high base flow index and low, specific runoff; and a diverse in-stream ecological community. However, other anthropogenic stressors, such as installation of artificial field drainage and in-channel dredging, will significantly compound and enhance the impact of P exported from agriculture. In addition to nutrients, impacts of sediment and flow velocity have also received attention in various studies, but knowledge of their combined impacts is limited (Elbrecht *et al*., 2016). Elbrecht *et al.* (2016) investigated the combined impacts of these three variables and found that the effects were additive in only two cases in which reduced flow velocity and nutrients affected *Gammarus* spp. and ceratopogonid midges. As highlighted by these well-studied examples of nutrients, sediment and flow, there are still major uncertainties with regard to the synergistic impacts of stressors, and the uncertainty is even greater for stressors for which little research has been conducted. In addition, even for a well-studied element such as P, there is still a limited understanding of how the additional stressors caused by climate change will alter its dynamics in water bodies or its interaction with other stressors (Moss *et al*., 2010).

In recent years, the acknowledgement of these uncertainties has resulted in the topic of multiple stressors becoming a key research focus (e.g. Ormerod *et al.* 2010). Reyjol *et al*. (2014) provided a detailed review of priority research issues related to achieving ecological status under the WFD, and highlighted the need to develop appropriate analytical and assessment tools. Recent projects that aim to elucidate the impact of multiple stressors on aquatic ecosystems include EU MARS (Managing Aquatic Ecosystems and Water Resources Under Multiple Stress) (http://www.mars-project.eu/), the SOLUTIONS Project (http://www.solutions-project.eu/) and the newly funded EPA DETECT (Disentangling the Impacts of Multiple Stressors on the Ecology of Waterbodies) Project (https://www.afbini.gov.uk/articles/detect-project). The SOLUTIONS Project aims to develop tools to identify and prioritise a wide range of chemical contaminants that pose a risk to both ecosystems and human health across a broad spectrum of land use sectors (Brack *et al*., 2015). The expectation is that these new approaches to identification will address issues such as mixture effects, lag times and biological responses (Brack *et al*., 2015). The MARS Project not only considers chemical contaminants, but also addresses multiple physical, chemical and biological stressors, and their interaction and impact on ecological responses in the context of the WFD (Hering *et al*., 2015). An overlapping theme to emerge from these projects is the need for holistic research that integrates agricultural stressors with those arising from other land use practices.

In Ireland, the DETECT Project aims to address the issue of multiple stressors through the development of an assessment framework that will facilitate the identification of the principal stressors constraining ecological recovery in water bodies. The EPA-funded DETECT Project is just a starting point for addressing the issue of multiple stressors in Irish water bodies and, subsequently, there will be a requirement for additional research to address the development of analytical methods, screening tools, modelling tools for source apportionment and risk assessments, and monitoring approaches that can account for mixtures and the synergistic impacts of multiple stressors (Altenburger *et al*. 2015; Hering *et al*., 2015). One of the outputs of the DETECT Project will be a literature review that considers multiple stressors in the context of Ireland, and the need for further research in this areas.

Key knowledge gaps

- Further research is required to determine the relative contributions of contaminant sources and physical stressors to ecological impacts on water bodies.
- More work is required to link agricultural practices with the form and timing of ecological impacts on water bodies.
- There is a limited understanding of how the additional stressors caused by climate change will alter the dynamics of a wide range of stressors in water bodies and their interactions.
- The development of tools (analytical frameworks, screening tools, modelling tools for source

apportionment and risk assessments, and monitoring approaches) to identify and prioritise a wide range of stressors that affect aquatic ecosystems is important.
- The development of appropriate analytical and assessment tools that can account for mixtures and the synergistic impacts of multiple stressors, and that will address issues such as mixture effects, lag times and biological response, is needed.

4.2.2 *Emerging contaminants*

As new contaminants are identified, there will be a need for further research to understand their impacts on aquatic ecosystems. Although the impacts of P and N are well researched in Ireland and elsewhere, there are a wide range of contaminants for which little is currently known with regard to their environmental fate, transport attenuation and impacts on aquatic ecosystems (both synergistic and individual) (Reyjol *et al*., 2014). Agricultural contaminants include endocrine disrupters, pesticides, pharmaceutical veterinary drugs and a wide range of chemicals associated with farm management. In a review of 500 organic microcontaminants, von de Ohe *et al.* (2011) identified 44 priority substances for the WFD, three-quarters of which were pesticides. Many of these pesticides arise from agriculture, but little is known about their transport, fate and impacts on the environment in Ireland. Stehle and Schulz (2015) concluded that because of a lack of empirical data, the impact of agricultural insecticides on surface water bodies had been underestimated. Clarke and Smith (2011) highlighted emerging contaminants that may be added to agricultural land in biosolid applications, which could then be transported into watercourses. They identified perfluorinated chemicals [perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA)] and polychlorinated alkanes (PCAs) as priorities for future research.

Although some research has been done on the occurrence of MCPA in Irish groundwater bodies (McManus *et al*., 2014), because of variations in soil moisture, UV light and temperature, the length of time it takes MCPA to degrade under Irish conditions is unclear, and half-life estimates vary from 12 to 50 days in soil. Understanding the conditions under which MCPA is lost via overland flow and/or subsurface flow will improve its management and reduce the risks posed to aquatic ecosystems. In addition, although statutory limits have been set for many pesticides in drinking water, the threshold value at which sublethal impacts occur in many aquatic organisms is still unclear (Brack *et al*., 2016). The importance of understanding the fate of pesticides in the environment was highlighted by McKnight *et al.* (2015), who investigated the impact of a number of pesticides on surface and groundwater bodies in Denmark. These authors found that groundwater quality was still affected by legacy sources of pesticides and their metabolites, even though these contaminants had not been used in Denmark for several decades. Rasmussen *et al.* (2015) found that pesticide concentrations in river bed sediment and SS exceeded safety thresholds in 50% of samples taken from Danish rivers. They also concluded that legacy pesticides in sediments were having a significant impact on stream biota, particularly macroinvertebrates.

Endocrine disrupters are another suite of compounds (including some pesticides) that have been identified as potential stressors in aquatic ecosystems; however, there is little empirical evidence available in Ireland to inform the development of mitigation strategies for endocrine disrupters. EPA (2005) carried out the first investigation of endocrine-disrupting compounds in Irish rivers; these authors found that, although entrogenic compounds were present in the River Lee, they were not at levels that would affect trout or the quality of drinking water. In contrast, impacts were observed on brown trout in the River Liffey, downstream of a WWTP. Endocrine-disrupter compounds can also occur in high concentrations in agricultural runoff; for example, Cai *et al.* (2012 and 2013) investigated the presence and removal of oestrogens and androgens in dairy wastewater from a farm in Northern Ireland. The dairy wastewater had a high concentration of both types of compound, which would have been likely to affect stream water quality if it had not been removed by passage through an integrated constructed wetland (Cai *et al*., 2012).

Pesticides and endocrine disrupters are just two of a wide range of contaminants that, along with their by-products and transformation products, could potentially affect aquatic ecosystems and for which little data exists to inform the development of effective mitigation strategies (Brack *et al*., 2015). There is a need for tools that can reduce this complexity in order to identify substances for priority action in specific catchments. One example of a developing technology is effect-directed analysis (EDA), which uses biomarkers and bioassays

and will facilitate an integration of the chemical and biological monitoring requirements of the WFD (Brack *et al*., 2016). Martinez-Haro *et al*. (2015) concluded that EDA was particularly useful for identifying the early impact of contaminants on aquatic ecology and for narrowing down the causes of ecological impairment, allowing a better understanding of the pressure–impact relationships. Similarly, the TRACE (Tracking and Assessing the Risk from Antibiotic Resistance Genes using Chip Technology in Surface Water Ecosystems) project (http://jpi-trace.eu) of the Water JPI aims to develop technologies for the detection of antibiotic resistance in aquatic systems, and the STARE (Stopping Antibiotic Resistance Evolution in the Environment) JPI project (https://stareeurope.wordpress.com) is also conducting ongoing research related to antibiotic resistance in the environment.

Key knowledge gaps

- More research is needed on the environmental fate, transport attenuation and impacts on aquatic ecosystems (both synergistic and individual) of contaminants such as endocrine disrupters, pesticides, pharmaceutical veterinary drugs, perfluorinated chemicals (PFOS and PFOA) and PCAs.
- There is a lack of empirical evidence to support the development of mitigation measures for a wide range of contaminants.
- The threshold values at which emerging contaminants have lethal and sublethal impacts on aquatic ecology are unclear.
- Information is lacking on the breakdown (transformation and by-products) of emerging contaminants and their legacy impacts on aquatic ecosystems.
- Tools and analytical methods to help identify substances for priority action in specific catchments are needed.

4.2.3 Water body sensitivity

Regime shifts in aquatic environments may occur because of, inter alia, climate change, land use change and nutrient enrichment (Carpenter and Lathrop, 2008). For example, Tománková *et al.* (2014) reported a 63% reduction in the overwintering diving duck population on Lough Neagh over an 8-year period, starting from the winter of 2000/2001. This correlated with a 65% reduction in the mean total density of macroinvertebrates between 1997/1998 and 2010, which also coincided with a significant decline in mean chlorophyll *a* concentrations before and after 2000/2001. Although there is uncertainty about whether or not these changes confirm the hypothesis that a regime shift occurred in the lake during this period (Tománková *et al*., 2013), it is clear that changes in the nutrient dynamics of the lake, as indicated by a decline in the chlorophyll *a* concentration, had significant consequences for the community structure of the ecosystem. Hobbs *et al.* (2005) highlighted that the capacity of Lough Carra sediments to bind P inputs from the surrounding catchments had declined, and Irvine *et al.* (2008) suggested that there was a high probability of a regime shift in the lake in the near future, with the accompanying loss of the unique submerged macrophyte populations. Similar concerns for Lough Melvin were expressed by Girvan and Foy (2006), because TP levels increased from 19 µg/L to 29 µg/L over a decade, which meant that the lake was almost eutrophic. However, despite the increase in TP, chlorophyll *a* concentrations remained less that 5 µg/L, as a result of light limitations due to the peat staining of the lake. As demonstrated by these examples, the threshold nutrient value and conditions under which a regime shift may occur vary significantly, depending on the water body and the ecosystem characteristics which determine its resilience (Folke *et al*., 2004).

Following on from Holling (1973), Walker *et al.* (2004) defined ecological resilience as “the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks”. A significant amount of research has been carried out on defining the resilience of aquatic ecosystems to land use and climate changes (e.g. Folke *et al*., 2004; Donohue *et al*., 2010; Willis *et al*., 2010; Dakos *et al*., 2015). For example, Donohue *et al.* (2010) examined the recovery of Lough Carra from anthropogenic impacts during the Great Famine in Ireland; during this period of famine, there was a significant reduction in the population and the pressures on the lake associated with land use . The authors found that the trophic level of the lake reduced significantly and reached a new equilibrium within 2–10 years. Woodward *et al.* (2015) investigated the resilience of macroinvertebrate communities in the Glenfinish River in Cork during periods of extreme flooding and extended drought over a 13-year period. They reported on one extreme flood event that resulted in a 10-fold decrease in community abundance, with most populations returning to their pre-flood levels in

less than 3 years, although some took up to 10 years to fully recover.

The accumulative and synergistic impacts of multiple stressors can reduce aquatic ecosystem resilience, making them less likely to absorb further stressors (Folkes *et al*., 2004). Despite recent research, the ability to determine the resilience of an aquatic ecosystem and predict the threshold values at which changes in the ecosystem will occur is still a challenge. Capon *et al.* (2015) reviewed the evidence for regime shifts, threshold values and multiple stable states in freshwater ecosystems and found little empirical evidence to quantify the magnitude or rate at which these processes occur in different systems. Scheffer *et al.* (2001) outlined the challenges of understanding the mechanisms that control changes in ecosystems, and highlighted the complex processes that operate at different temporal and spatial scales and in the context of different ecological organisation.

Understanding the variability in the resilience of aquatic ecosystems to land use change is key to setting realistic conservation targets for aquatic ecosystems and has direct relevance for managing agricultural intensification. Although determining ecological reference conditions for water bodies is a well-established step in defining water quality targets in many countries (Dodds and Oakes, 2004; Kelly-Quinn *et al*., 2009; Bouleau and Pont, 2015), to date these have been approximations based on typology that fail to account for the significant variability within and between catchments (Moss, 2008b; Page *et al*., 2012; Bouleau and Pont, 2015). Although current estimates of reference conditions are valuable, a greater degree of certainty on a catchment-specific basis is required (Hawkins *et al*., 2010) in order to establish more accurately the boundary values between categories of ecological states (Stoddard *et al*., 2006). There can be significant temporal and spatial variation in reference conditions even on a small geographical scale (Soranno *et al*., 2011).

Improving the current estimates of reference conditions will help to refine our understanding of the relationship between catchment pressures and impacts in aquatic ecosystems, and the threshold values at which these impacts occur (Stoddard *et al.*, 2006). The current WFD outcomes reflect the best current scientific understanding of the water body response, what is worthwhile doing (in terms of technical feasibility and benefits exceeding costs) and what can realistically be expected to be achieved in terms of land use change by 2027 (Hering et. al., 2010). In terms of achieving the objectives of the WFD, current agricultural planning approaches assume that the impacts of agriculture can be reduced sufficiently to achieve the objectives of the WFD. Although this may be possible in many cases, the evidence required to support this hypothesis is lacking in the context of the recovery trajectories and threshold conditions in aquatic ecosystems at which impacts for a wide range of stressors occur. Restoration and degradation trajectories of aquatic ecosystem may differ and this has consequences for predicting the recovery of water bodies to good status (Harris and Heathwaite, 2012). Jarvie *et al.* (2013) suggested that, in some cases, baselines have shifted, so that many affected water bodies may never return to reference conditions, even if pressures are reduced below the threshold values at which degradation occurred.

Key knowledge gaps

- It is important to determine the resilience of aquatic ecosystems and predict threshold values at which changes in specific aquatic ecosystems will occur.
- Determining the accumulative and synergistic impacts of multiple stressors on the resilience of aquatic ecosystems, and the corresponding threshold values, is important.
- Empirical evidence to quantify the magnitude or rate of regime shift, threshold values and multiple stable states in freshwater ecosystems is needed.
- An improved understanding of the mechanisms controlling changes in ecosystems, in the context of complex processes operating at different temporal and spatial scales and ecological organisation, is needed.
- Setting realistic conservation targets for aquatic ecosystems in the context of temporal and spatial variability in the resilience and reference conditions of aquatic ecosystems is important.
- More evidence is needed to demonstrate that the impacts of agriculture on aquatic ecosystems can be reduced sufficiently to achieve the objectives of the WFD in the context of the recovery trajectories and threshold conditions of aquatic ecosystems.

4.3 Climate and Weather Patterns

Climate has an overarching impact on the relationship between agriculture and aquatic ecosystems: it

influences land use change, contaminant export, and water body sensitivity and recovery. It also influences our ability to demonstrate the effectiveness of mitigation strategies. These impacts need to be considered over a wide range of temporal and spatial scales, from short-term weather events (hours–days) to long-term climate change over decades. There is evidence, both from national and international studies, to support the need for a greater understanding of the current and potential impacts of climate with regard to defining the effect of agriculture on water quality. Research has also demonstrated the impact of storm events (e.g. Mellander *et al*., 2013; Crockford *et al*., 2014; Huebsch *et al*., 2014; Shore *et al*., 2016b), seasonal (Jordan *et al*., 2012; Jennings *et al*., 2013) and inter-annual variation (Watson *et al*., 2007; Sheriff *et al*., 2015), and global climate processes (Scarsbrook *et al*., 2003; Gascuel-Odoux *et al*., 2010; Mellander *et al*., 2015). However, less evidence exists with regard to the impact of climate on the larger scale, resulting in a greater degree of uncertainty with regard to the effect of climate and weather patterns on the relationship between agriculture and aquatic ecosystems. Apportioning the variability caused by global-scale drivers, as opposed to local-scale impacts, is vital for optimising land use interventions and setting realistic expectations for recovery.

Climate variability has significant implications for our ability to demonstrate the effectiveness of mitigation measures and the recovery of aquatic ecosystems, because the large-scale influences of climate will potentially override land use interventions at local scales (Gascuel-Odoux *et al*., 2010). Mellander *et al.* (2015) demonstrated a significant correlation between nitrate concentration in Irish rivers and the position of the Gulf Stream north wall (GSNW), as described by the GSNW index, and Jennings and Alott (2006) observed a similar relationship in Irish lakes. Similarly, Scarsbrook *et al.* (2003) examined the relationship between the El Niño Southern Oscillation (ENSO), as characterised by the Southern Oscillation Index (SOI), and 13 water quality variables in 77 streams in New Zealand over 13 years. The analysis found significant linear relationships between the SOI and all 13 variables; the strongest relationships were between the SOI and water temperature, DRP and oxidised N. Scarsbrook *et al.* (2003) concluded that any analyses of long-term trends in water quality should take the impact of climate variability into account.

Floury *et al.* (2012) reported that, although the impacts of climate change on air temperature have accounted for 80% of the variation in water temperature in the Loire River over the past 32 years, inter-annual differences in precipitation accounted for only 18% of the variability in river discharge, which reduced by 100 m^3/second during the same period. They concluded that increases in abstraction and evapotranspiration at catchment scale played a greater role in the decline in river discharge than large-scale climate variability, which highlights the potential impact of further agricultural intensification on catchment hydrology. Delpla *et al.* (2009) highlighted the potential impact of climate change on drinking water sources in temperate regions: an increase in soil, water and air temperature, combined with higher intensity rainfall, resulted in an increase in the export of dissolved organic carbon, micropollutants and pathogens to water bodies. Hari *et al.* (2006) also considered the impact of climate-driven changes in water temperature on ecology in Swiss Alpine rivers, and demonstrated that such changes effectively resulted in a reduction in the habitat available for brown trout. The reduction in habitat resulted from higher water temperatures, which caused an increase in proliferative kidney disease in streams at lower attitudes; natural barriers prevented trout movement to areas upstream where lower temperatures limit the prevalence of the responsible parasite (Haria *et al.*, 2006). The impact of climate change on different water quality parameters is site dependant and will vary significantly across regions (Scarbrook *et al*., 2003; Gascuel-Odoux *et al*., 2010).

Considerable efforts have been made to disentangle the impacts of climate change and anthropogenic nutrient inputs on the trophic status of lakes using sediment records (Battarbee *et al*., 2012). Issues related to equifinality (i.e. in an open system a given end state can potentially be reached in multiple ways) have complicated this analysis, with both climate change and nutrient inputs increasing algal blooms, oxygen stress and nutrient cycling within lakes (Moss *et al*., 2011). Recent work on Lough Neagh by McElarney *et al*. (2015) demonstrates that the significant changes in water temperature in recent years were driven mainly by changes during spring and autumn, which may have implications for nutrient release from sediment, food web interactions and the occurrence of algal blooms on the lake (McElarney *et al*., 2015). The uncertainty related to predicting climate change impacts is increased by the variability in how climate change is expressed in

terms of changes in, inter alia, seasonality, air temperature and rainfall patterns, and by the natural variability that inherently occurs in these variables over different temporal scales. Shimoda *et al.* (2011) concluded that there were two key requirements for research on the relationship between climate change and lake ecosystems: firstly, they concluded that there is a need to focus on elucidating the impact of climate change on in-lake chemical, physical and biological processes, and, secondly, they suggested that the heterogeneity in the response of lakes across a wide spatial and temporal scale should be investigated.

In addition to affecting observations and processes within water bodies, climate changes will significantly impact on agricultural practices and the export of pollutants. Schoumans *et al.* (2014) highlighted the need for "climate-smart" measures to address the reduction in water quality caused by an increase in P loss from agriculture as a result of climate change. Ockenden *et al.* (2016) predicted a 9% increase in TP export from upland catchments in the UK by 2050 as a result of climate change. Cassidy *et al.* (in press) demonstrated that SMD has a greater impact than soil P concentration on P export from an agricultural grassland soil, with peaks in P associated with runoff events that followed periods of high SMDs. These findings support the hypothesis that the predicted greater SMD and occurrence of summer storm events in Ireland will exacerbate the export of P from agriculture to water during ecologically sensitive periods in summer. This is further supported by a recent analysis of high-resolution data by Ockenden *et al*. (2016). These authors predict that, between now and 2050, climate change will increase stream P concentrations more frequently, and for longer periods, in summer because of drier soil conditions. Climate change will also affect nutrient export indirectly through changing management practices, water abstraction and changing fertiliser practices (Schoumans *et al*., 2014). Jennings *et al.* (2009) examined the potential impact of climate change on P export for grasslands and concluded that the increase in P loading to water bodies caused by climate change would be greater than the increase due to population or land use change. It was concluded that, in future, reducing the period when slurry can be spread to between 1 April and 30 September would mitigate the increase in dissolved P loss that will occur in spring as a result of climate change (Jennings *et al*., 2009), highlighting the significant impact that climate change could have on farming practices. Because of this impact and the increase in exports, a re-evaluation of current mitigation strategies is needed to ensure that they will be effective under future climate scenarios (Schoumans *et al*., 2014).

Key knowledge gaps

- How to predict the impact of global climate change and local weather variability on the mobilisation, attenuation and delivery of a wide range of agricultural contaminants.
- How to apportion variability in agricultural impacts to global-scale drivers, as opposed to local-scale drivers, in order to optimise land use interventions and set realistic expectations for recovery.
- How to demonstrate the effectiveness of mitigation measures and the recovery of aquatic ecosystems within the context of climate influences at large scales that potentially override land use interventions at local scales.
- How to predict the impact of climate change on different chemical, physical and biological aquatic processes at local, catchment and regional scales.
- A re-evaluation of current mitigation strategies and the development of "climate-smart" mitigation measures that will be effective under future climate scenarios is required.
- Further research is needed in the impact of climate change and variability in local weather patterns on agricultural practices and the consequences for aquatic ecosystems.

4.4 Predicting Sources and Responses

4.4.1 Source apportionment

The effective allocation of resources and targeting of mitigation measures is largely dependent on the ability to accurately identify the sources of the stressors that affect aquatic resources. Although there has been significant investment in nutrient mitigation measures and some evidence exists that links these interventions with improvements in chemical water quality (e.g. Murphy *et al.*, 2015; Barry and Foy, 2016), evidence for the success of these measures in terms of ecological improvement is limited (Jarvie *et al*., 2013). This may, in part, be because of inadequate source apportionment which has, in turn, resulted in the sub-optimal targeting of mitigation strategies (Bowes *et al*., 2014). The limitations of current source apportionment approaches

are due to difficulties in accounting for environmental heterogeneity in models and limitations in our understanding of the issues outlined in previous sections, with transformation/attenuation of contaminants/water along surface and subsurface pathways, synergistic impacts of multiple stressors and uncertainties over ecological response all influencing predictions. The development of source apportionment approaches that are based on ecological impacts rather than loads is recognised as an important step in refining the targeting of mitigation measures in the context of both agriculture and other land uses [see, for example, the recent Department for Environment, Food & Rural Affairs (Defra)-funded project “Developing a field tool kit for ecological targeting of agricultural diffuse pollution mitigation measures” (Project WQ0223; sciencesearch.defra.gov.uk), for which the final project report is pending]. The aim of this project was to develop a tool that would enable the identification of catchments for which agricultural mitigation measures would have the highest probability of improving aquatic ecology.

The source apportionment of catchment nutrients has received most attention, both in Ireland and elsewhere. In a study of 15 sub-catchments of the Thames river, Bowes *et al.* (2014) highlighted the importance of the ability to disaggregate contributions based on the timing of nutrient contributions, in order to enable a more comprehensive definition of the risks posed to water bodies. They used a load apportionment model (LAM) to identify the contributors to N and P in the rivers; diffuse sources and remobilisation from within the river channel were identified as the largest contributors to annual P loads at each monitoring site. However, sewage treatment works made the largest contribution during low-flow times in the spring and autumn when the rivers were most ecologically active. Bowes *et al.* (2014) concluded that although agriculture made a larger contribution to annual loads in the rivers, a further reduction in P export from sewage treatment works would be a more cost-effective approach to achieving the targets of the WFD because of the inputs during low-flow periods (Jarvie *et al*., 2006). Withers *et al*. (2014) highlighted that a very large number of small sewage works in England and Wales emit much greater P concentrations than those delivered by agriculture during low-flow conditions. However, for N mitigation strategies, a focus on reducing inputs from agriculture and groundwater were predicted to be more effective. The remobilisation of sediment in rivers during storm events can also lead to errors in source apportionment predictions. Jarvie *et al.* (2012) highlighted that although previous predictions assigned P increases during storm-flow events to agricultural diffuse points under some circumstances, these increases arise, in fact, from the remobilisation of P from river bed sediments contaminated with P from upstream WWTPs.

The contribution of P and N from septic tanks during low-flow periods to rivers has received increasing attention in recent years (e.g. Withers *et al*., 2012). For example, Macintosh *et al.* (2011) demonstrated that the strategic replacement of some defective septic tanks reduced low-flow TP concentrations in one catchment by 0.032 mg/L over 4 years. Using a load apportionment modelling approach in one catchment in Ireland, Greene *et al.* (2011) found significant temporal and spatial variation in P loads depending on soil type, agricultural intensity and anthropogenic sources such as WWTPs. They reported that in one sub-catchment, despite agriculture making the largest annual contribution to P loads, point sources dominated flow for as much as 64% of the hydrological year. Gill and Mockler (2016) highlighted the importance of catchment scale in assessing the contribution of septic tanks: for small catchments, septic tanks contributed, on average, 13% of N and 22% of P in rivers, whereas for large catchments, the relative contributions were, on average, only 2% of N and 1% of P.

Many authors nationally and internationally have highlighted the importance of these inputs during ecologically sensitive times (Arnscheidt *et al*., 2007; Macintosh *et al*., 2011; Withers *et al*., 2011; Jordan *et al*., 2012; Withers *et al*., 2012, 2014). Although the overall annual load arising from septic tanks may be relatively small, the timing of this input to rivers may be much more significant in terms of its ecological impact (Bowes *et al*., 2014; Stamm *et al*., 2013). Stamm *et al.* (2013) concluded that an accurate assessment of the impact of P on aquatic ecosystems must distinguish between the role of P concentration and load in rivers and standing water bodies. Although in lakes, because of higher residence times, P loads are the appropriate metric for assessing impacts, concentrations and the timing of the inputs are the key metrics for assessing impacts in rivers (Edwards and Withers, 2008 Stamm *et al*., 2013).

In addition to nutrient source apportionment based on land use sector (i.e. agriculture, septic tanks, WWTPs,

forestry, etc.), disaggregating agricultural contributions to water body loadings based not just on diffuse and point source inputs, but also on contemporary and historical land use activities and the separation of transport pathways is vital for the cost-effective targeting of mitigation measures. If contemporary land use practices, such as incidental losses of fertiliser, make a significant contribution, addressing these sources will be far less complex than if legacy sources of soil and/or sediment P are the drivers of nutrient export (Withers *et al*., 2003). Barry and Foy (2016) demonstrated how the regulation of P use in Northern Ireland, which largely focuses on incidental P losses, has resulted in a significant improvement in P concentrations in monitored lowland streams, but that legacy soil P losses have the potential to maintain river concentrations above WFD targets. Cassidy *et al.* (2016) found that, despite a reduction in Olsen P over a period of 7 years of zero P application to grazed grassland soils in Northern Ireland, there was no corresponding improvement in P concentrations in runoff, apart from those associated with incidental losses. During this period, there was a decoupling of the dependence of P concentration in runoff from the concentration of P in soil: the field plot that had a starting Olsen soil P concentration of 19 mg/L (agronomic optimum) recorded similar, and on occasion higher, concentrations of P in overland flow than the plot with a starting Olsen soil P concentration of 67 mg/L (Cassidy *et al*., in press). Cassidy *et al.* (2016) concluded that for some soil types, legacy soil P poses a significant long-term threat to water quality, even if the soil has agronomically optimum P levels.

Correct source apportionment between contemporary and historical sources of contaminants could have a significant impact on the sustainability of agriculture in some areas. For example, utilising legacy soil P may require farming on a negative P balance in order to reduce the risk posed to water quality (Rowe et. al., 2016). However, this could compromise agricultural output, especially for housed or partially housed livestock farms for which it is currently uneconomical to export manure nutrients to crop-producing areas because of the geographical disconnection (Doody *et al*., 2014b). Such a scenario may limit the ability of livestock farmers to mitigate losses cost effectively; however, in other catchments, where impacts are due to contemporary management practices (e.g. poor fertiliser and manure management), implementing measures will be less difficult. The pathway of loss could also impact on the implementation of mitigation strategies, as groundwater contributions of legacy nutrients to surface water bodies could potentially extend the time of recovery because of lag times in responses (Fenton *et al*., 2011).

Sediment fingerprinting has also received increasing attention in Ireland in recent years; this has helped to refine estimates of the sources of sediment based on land use sector (Thompson *et al*., 2013; Sheriff *et al*., 2015). Thompson *et al.* (2014) highlighted the need to refine estimates of the delivery of sediment to water bodies in order to accurately estimate the impact on aquatic ecology. They found that a sediment export rate of 0.07 tonnes/ha per year and 0.44 tonnes/ha per year, in two catchments in Ireland, resulted in annual exceedances of 8.3% and 17.8%, respectively, of the target value of 25 mg/L set under the European Union Freshwater Fish Directive (FFD) (78/659/EC). Exceedance events were typically of less than 5 hours in duration (Thompson *et al*., 2014). They concluded that more information was required on sediment concentration rather than on loads if links with ecological impacts were to be established. In another study, Reaney *et al.* (2011) used risk-based modelling to disaggregate the impact of land use on the abundance of salmoinid fry. Their approach involved examining the link between hydrological connectivity and impacts on the fry community, and they concluded that intensive pasture in the catchment was having a greater impact than arable sources. Sear *et al.* (2016) demonstrated that both sediment quantity and sources need to be considered in assessments of ecological impact, and that the impact of sediment source on the mortality and fitness of salmonid alevin was determined by the organic matter content and oxygen consumption of catchment sediment source material (Sear *et al*., 2016). In terms of appointing sources based on ecological impact, this understanding of the relative importance of sediment quantity and quality to different organisms will help to refine estimates of impact (Collin *et al*., 2013). The EPA-funded SLITFLUX project (http://77.74.50.157/siltflux/Home.aspx), due to be completed in 2016, will further increase the knowledge and understanding of sediment dynamics in Irish rivers.

MST has the potential to disaggregate sources of microbial contamination in water bodies, and Flynn *et al.* (2016) have applied this method in the Mattock catchment. They were able to identify different sources of faecal material during the year: in summer, contaminants were attributed to the access of liverstock to

rivers during that period, while, at other times of the year, microbial material associated with sewage dominated (Flynn *et al*., 2016). However, Oliver *et al.* (2016) identified the need for more evidence on the fate and behaviour of FIOs in the environment in order to reduce the uncertainty in the predictions of current modelling efforts.

Although the importance of defining nutrient and sediment source apportionment by time and bioavailability has been recognised (Stamm *et al*., 2013), less attention has been focused on disaggregating the sources of other aquatic stressors. Reyjol *et al.* (2014) highlighted the lack of tools available for linking catchment pressures with ecological impacts at different spatial scales and for a wide range of contaminants. There are significant difficulties associated with the source apportionment of multiple stressors because of the often synergistic and additive impact of stressors on aquatic ecosystems. For example, the removal of water retention features, such as wetlands, from the landscape may enhance sediment and absorbed pesticide export to rivers. However, the resulting ecological impacts may be due to the individual or combined impacts of an increase in river flow velocity, sedimentation and/or the toxicity of pesticides to aquatic organisms. It will be dependent on the threshold value at which a stressor's impact on an ecosystem is exceeded, and the ability to identify this tipping point would greatly enhance the prioritisation and cost-effectiveness of mitigation measures (Doody *et al*., 2016).

Key knowledge gaps

- Source apportionment approaches should be developed on the basis of ecological impact rather than on contaminant loads, in order to refine the targeting of mitigation measures in the context of both agriculture and other land uses
- The limitations of current source apportionment approaches to accounting for environmental heterogeneity, transformation/attenuation along hydrological pathways, and the synergistic impacts of multiple stressors and uncertainties over ecological responses should be addressed.
- Agricultural impacts on water bodies should be disaggregated on the basis of contemporary and historical land use activities.
- The further development of source apportionment approaches, in order to provide better estimates of the contribution from different transport pathways, is required.
- Source apportionment techniques should be developed and refined for a wide range of agricultural contaminants (e.g. sediment, faecal contamination, pesticides, emerging contaminants, etc.). For example, there is a need to improve evidence on the fate and behaviour of FIOs in the environment to reduce uncertainty in current predictions.
- The challenges associated with the source apportionment of multiple stressors because of the often synergistic and additive impact of stressors on aquatic ecosystems must be elucidated.

4.4.2 *Targeting measures*

To date, strategies in Ireland to address nutrient export from agriculture have focused on source control, and transport factors have not been incorporated successfully into policy (Shore *et al*., 2014). This contrasts with the evidence base, which supports the source–pathways–receptor continuum concept of contaminant delivery to surface waters in catchments (Haygarth *et al*., 2005). This concept has been recognised for over a decade and it promotes research approaches that consider hydrological connectivity in the landscape and associated risks to water quality. This has led to an increased focus on the identification of metrics based on hydrological connectivity and the propensity for saturation-excess overland flow to represent the role of hydrology in the context of primarily P and sediment export. The concept of CSAs (Gburek *et al*., 2000; Heathwaite *et al*., 2000; Pionke *et al*., 2000; Sharpley *et al*., 2008), which incorporates mobilisation and transport potential, has been successfully demonstrated at catchment scale in an Irish geoclimatic context in work by Jordan *et al.* (2012) and Shore *et al.* (2014) and at field scale by Thompson *et al*. (2012). Shore *et al.* (2014) demonstrated that, in some catchments, transport metrics alone can be used to predict TRP levels in storm runoff, regardless of differences in soil P, while in other catchments additional soil and land use information is required to make predictions of TRP and particulate phosphorus (PP).

This greater focus on the role of transport metrics, both in Ireland and internationally (e.g. Heathwaite *et al*., 2005; Page *et al*., 2005; Lane *et al*., 2009; Reaney *et al*., 2011; Doody *et al*., 2012a; Buchanan *et al*., 2013; Thompson *et al*., 2013; Smith and Blake, 2014), has led

to the development of approaches to predict the spatial extent of hydrologically connected areas and their frequency and duration within catchments (Shore *et al*., 2013, 2014). The Network Index, developed by Lane *et al*. (2004, 2009) builds on the Topographic Wetness Index developed by Beven and Kirkby (1979), and uses derivatives of digital terrain model elevation data to predict hydrological connectivity based on the propensity of each point in the landscape to be saturated. This has been the main approach adopted in both Ireland and the UK in recent years, which may largely be because of the parsimonious data requirements compared with other mechanistic models (Lane *et al*., 2009). Increasingly, efforts have centred on identifying the optimum digital elevation model (DEM) scale to accurately predict connectivity, which varies considerably depending on the contaminant of interest, the topography and the degree of human modification. Shore *et al*. (2013), in an evaluation of the ability of the Network Index to predict surface connectivity in two small ($\approx 12\,km^2$) agricultural catchments with contrasting permeabilities, validated modelled surface connectivity (for a 5-m DEM) at sub-catchment ($\approx 1.3\,km^2$) and field scales through comparison with field observations. They found that, although the lack of information on field drains and ditches affected the delineation of sub-catchment boundaries, it did not impact sub-catchment connectivity. At field scale, they found that the Network Index is reliable for differentiating between well- and poorly connected fields, which is useful for targeted mitigation measures. Drainage ditches were found to have a low overall effect on the magnitude of surface connectivity in the catchments, and the authors suggested that less topographic detail may be better for predicting surface saturation, which is consistent with the findings of Sørensen and Seibert (2007). However, subsequently, Thomas *et al*. (2015) suggested that 1- to 2-m DEM data derived from Light Detection and Ranging (LiDAR) data was the optimal resolution for modelling hydrologically sensitive areas (HSAs), defined as areas of the catchment that have a high propensity for overland flow and high transport risk, combined with being hydrologically connected to a water body. Thomas *et al*. (2015) revised current approaches to the delineation of CSAs using the P Index (Gburek *et al*., 2000; Drewry *et al*., 2007; Buchanan *et al*., 2013), by incorporating the delineation of HSAs into the index rather than just proximity to water courses.

Although further research could reduce uncertainties with regard to the CSA Index approach, related to resolution, applicability to contaminants other than TP or sediment, and address the disconnect between a focus on surface transfers and the need to incorporate subsurface pathways (Packham *et al*., 2013, EPA, 2016a), there are also significant challenges in determining how such a tool could be operationalised at sufficient resolution over a wide geographical scale. The use of the revised CSA Index in combination with the EPA Catchment Characterisation Tool (CCT) may be one potential approach to combining the identification of high-risk catchments with delineation of CSAs at field scale and the cost-effective targeting of mitigation measures. However, how this can be achieved is yet unclear and there are significant knowledge gaps related to making these tools/methods operational in terms of costs and integration into policy, institutional structures and farming practices.

Currently across Ireland, mitigation measures to control agricultural pollution are largely implemented at farm scale, with very little consideration given to intra- and inter-farm variation. The ability to identify CSAs within catchments increases the cost effectiveness of mitigation measures and the likelihood of achieving the objectives of the WFD (Doody *et al*., 2012a). Thomas *et al.* (2016) examined the cost effectiveness of the implementation of riparian buffer strips in four catchments in Ireland. They reported a reduction of 66% and 90% over 1–5 years in the potential implementation costs of an agri-environmental scheme that utilised the revised CSA–HSA approach. However, to date, there has been no incentive or policy drive in Ireland to delineate these areas, because, until recently, there was a poor evidence base for policy development and difficulties related to managing these areas separately. However, both national and international research has now largely provided this evidence base, and the development of the EPA CCT and CSA Index (Thomas *et al*., 2016) has provided the tools necessary to operationalise this approach in Ireland.

The implementation of a targeted approach to mitigation measures will have significant consequences for policymakers, administration, advisory services and farmers. A key gap in socioeconomic research relates to the adaptive capacity of these stakeholders to successfully adjust to this approach. Adaptive capacity is defined as the precondition that enables adaption to change, including the financial, human, social, natural

and physical capitals that facilitate change (Brown *et al.*, 2010; Lockwood *et al.*, 2015). In addition, optimising the scale and resources required to implement this approach successfully needs further investigations that account for both socioeconomic and biophysical barriers to its implementation. For example, Buckley *et al.* (2015) examined the factors that motivated farmers to adopt nutrient management practices and found that age and off-farm employment were found to affect the adoption of these practices. In the case of off-farm employment, it is likely that time constraints were the main reason for the limited uptake of nutrient management practices (Buckley *et al.*, 2015).

Key knowledge gaps

- The uncertainties in the CSA Index approach, related to resolution and the applicability to contaminants other than nutrients or sediment, need to be further reduced, and the need to incorporate subsurface pathways must be addressed.
- New or existing tools or methods should be developed and integrated in order to cost-effectively target mitigation measures at agricultural diffuse and small point sources in catchments across a wide geographical area.
- The administrative, operational, practical, behavioural and policy barriers to implementing tools/methods for targeting measures at farm and catchment scales must be identified.
- The administrative, operational and practical costs that need to be taken into consideration when implementing tools/methods for targeting measures at farm and catchment scales must be accounted for.
- The adaptive capacity of agricultural, advisory and policy stakeholders to successfully deliver a targeted approach to the implementation of measures at farm and catchment scales must be elucidated.

4.5 Behavioural Heterogeneity

4.5.1 Engaging stakeholders

In terms of the protection of water resources, agricultural stakeholders play a significant role in many catchments through ownership and management of agricultural land. As such, their engagement in catchment management is integral to its success. Stakeholders and, in particular, land owners could play a significant role in the collection of data with a higher temporal and spatial resolution, which could help to overcome uncertainties related to catchment heterogeneity. Stakeholder engagement is a cornerstone of integrated catchment management and is enshrined within Article 14 of the WFD. The aim is to include stakeholders in the decision-making and management processes related to water resources, thereby increasing the sustainability of catchment interventions and their acceptability. There are many issues related to stakeholder heterogeneity, such as cohort, empowerment and attitude, that also play a role in stakeholder engagement in catchment management initiatives and, in particular, with scientific evidence gathering, monitoring and evaluation. These are discussed in section 4.5.2.

The recent EPA StreamScapes report (EPA, 2015) on public engagement highlights the need to facilitate initiatives that allow communities to develop local plans for the management of their catchment. The report makes a number of recommendations for successful stakeholder engagement related to, for example, addressing issues at local scale, using multiple “hooks” to engage stakeholders and identifying “honest brokers” to facilitate engagement with local communities. In addition to the EPA StreamScapes project (EPA, 2015), there have been a number of projects in Ireland that have successfully engaged with stakeholders through catchment initiatives such as the Ballinderry Rivers Trust (Horton *et al.*, 2015) and the Lough Melvin Nutrient Reduction Project (Campbell and Foy, 2008). However, international experience shows that managing “bottom up” stakeholder participation is problematic, and achieving a sense of shared purpose and ownership is difficult (Dooris, 1999; Mega, 2000; Doody *et al.*, 2012b). Doody *et al.* (2012b) concluded that sustainable implementation of a bottom-up participatory process poses significant challenges for public authorities that are structured to effectively implement a bureaucratic “top-down” regulatory style of management. The EPA-funded projects “Towards Integrated Water Management” (TIMe) and “Extra TIMe” (see www.dkit.ie) will provide further evidence for the need for integrated catchment management and how this can be achieved through bottom-up community-based processes.

Often, agri-environmental interventions in catchments are based on first-order research and development, which rely on the objectivity of scientific research to develop objective data through experiments and monitoring (Ison and Russell, 2007; Ison *et al.*, 2007). These

data are then validated through the peer-review process and, despite the difficulties associated with converting scientific knowledge into standardised measures (Quevauviller *et al*., 2005), form the basis of future mitigation strategies. Proponents of this approach assert that it produces independent objective measures that are based on "good science" and are free from bias. In addition, opponents of stakeholder participation in such processes express concern that the involvement of stakeholders results in contradictory outcomes and thus undermines the validity of the process (Wolfe *et al*., 2001). However, this approach to research and development does not take into account the human factors that will affect the implementation of the proposed measures or differences in the perception of how the problem should be solved (Doody *et al*., 2009b). Consequently, it has not fully negated the potential for conflict and the uncertainties associated with implementation, and so it is prudent that alternative approaches to agri-environmental research are explored. One such approach by Schulte *et al.* (2009) engaged farmers in the Lough Melvin catchment in the identification and evaluation of mitigation measures to reduce agricultural P export to the lake. Subsequently, Doody *et al.* (2009a) evaluated the engagement process and concluded that a key outcome from the process was the two-way transfer of knowledge between the stakeholders and the researchers. This helped to build a consensus on a list of catchment-specific measures that are environmentally and cost effective, while also being acceptable to farmers. This has important implications for the effective implementation of measures by farmers in the future (Doody *et al*., 2009a).

"Citizen science" has been proposed as an approach to engaging a wide range of stakeholders in scientific monitoring, data collection and knowledge exchange (Dickinson *et al*., 2012). Citizen science sits at the interface between participation, and research and knowledge exchange (Bonney *et al*., 2014), and could be a powerful tool for aligning environmental research priorities with those of communities (Pandya, 2012). In the context of protecting water resources, aligning agricultural and environmental priorities has proven to be problematic and a source of conflict in many catchments (Doody *et al*., 2016). A citizen science approach requires scientists and stakeholders to adopt an approach that is somewhat different from their traditional roles, as they must be open to collaboration with both "experts" and "non-experts". This sort of integration can be difficult to achieve, as there is often uncertainty regarding the differences in perceptions related to the main environmental issues that need to be tackled (Brugnach *et al*., 2008; Doody *et al*., 2009). For this approach to be effective, all the participants need to recognise the need for synergy between agricultural and environmental priorities.

Bonney *et al.* (2009) divided the components of citizen science into assembling teams, resources and partners; defining research questions; data collection and management; the analysis and interpretation of data; dissemination; and, finally, the evaluation of the programme and the outcomes for participants. Subsequently, Newman *et al.* (2012) reviewed how future advances in each of these components would result in more robust citizen science and increased acceptability within the scientific community. According to these authors, advances in hardware and software technology, such as sensors, smart phones, social media, computer processing and data storage, are central to the future of citizen science (Newman *et al*., 2012).

The citizen science approach has been used predominately in the field of ecology to engage stakeholders in the collection of data on, inter alia, the occurrence, habitats and behaviour of a range of terrestrial and aquatic species (Dickinson *et al*., 2012). Farmers have a closer relationship with the natural environment than people in most other sectors of society, so engaging them in citizen science could provide significant opportunities for data collection, education and knowledge exchange. Advances in technology have provided opportunities to align the priorities of agricultural and agri-environmental research by providing farmers with the tools for precision agriculture while simultaneously providing agri-environmental scientists with a rich source of data to help target, monitor and evaluate catchment-scale interventions.

One example of the use of such an approach is provided by Teagasc's PastureBase Ireland tool (Griffith *et al*., 2014). This web-based management decision support tool provides customised grassland management advice for participating farmers based on data they input for their own farms. The data they provide are stored in a database and are available to researchers and other stakeholders. For example, Teagasc researchers use the data to investigate the persistency and performance of perennial ryegrass cultivars, while the farmers that

contribute the data receive grassland management advice (Griffith *et al*., 2014). This approach could be extended through investment in sensors for, for example, soil moisture and temperature and crop yields. This would provide grassland farmers with vital information that would allow the precise application of fertiliser and the optimisation of yields in individual fields, while providing scientists with data to, for example, validate models for predicting soil moisture and yields. This is just one example of the potential application of citizen science in grassland agriculture, for which the technologies currently exist but have yet to be adopted widely in Ireland and elsewhere.

Outside Ireland, Rossitier *et al.* (2015) have advocated the use of citizen science in the context of digital soil mapping and have outlined a list of recommendations related to engaging stakeholders, including clearly defined protocols for data collection and the validation of submitted data. The use of citizen science in the management of freshwater ecosystems has received increasing attention in recent years (Lowry and Fienen, 2013; Winfield, 2014; Mackay *et al*., 2015; Milner *et al*., 2016). Milner *et al*. (2016) described the use of the Natural Resource Planning Portal to engage local stakeholders in catchment management. The portal provides baseline land use, topographic and high-resolution imagery layers, and local landcare stakeholders are allowed to add additional spatial features and information using custom-built mapping tools (Milner *et al*., 2016). Mackay *et al.* (2015) proposed the use of digital catchment observatories to engage a wide range of stakeholders in catchment management, but highlighted the technical- and infrastructure-related challenges posed by the need for multiple end-users, communication issues, the importance of the use of local knowledge and uncertainties related to cloud-based models. Lowry and Fienen (2013) described a “crowdsourcing” approach for the collection of spatial and temporal hydrological data in catchments. They evaluated the accuracy of the data collected and, despite some irregularities, concluded that with additional training the quality of the data supplied by the participants would be acceptable for inclusion in analysis (Lowry and Fienen, 2013).

Future advances in technology will increase the opportunities for collaboration between scientists and farmers, but key issues regarding data quality and validation, the behavioural changes of scientists and farmers, technology adoption and the management of large datasets also need to be addressed if citizen science approaches are to become mainstream (Newman *et al.*, 2012). Citizen science that focuses on precision agriculture has significant potential to balance the agricultural and environmental priorities; it also has strong synergies with progress towards the effective targeting of mitigation measures, as outlined in previous sections. The successful implementation and mainstreaming of citizen science within current precision agriculture research programmes may be the first step towards the greater participation of farmers in other scientific endeavours, such as water quality monitoring, river bank assessments and ecological monitoring. This would have significant implications for extending current monitoring programmes, knowledge exchange and building trust among catchment stakeholders, and would provide researchers with data that would enable them to address issues that otherwise may not be practical (Alaback, 2012).

Key knowledge gaps

- It will be important to include agricultural stakeholders in the collection of data of a higher temporal and spatial resolution, which will help to overcome uncertainties related to catchment heterogeneity.
- To address the barriers to engaging agricultural stakeholders in citizen science programmes, there should be a focus on data collection, education and knowledge exchange.
- Technologies that will provide farmers with the tools for precision agriculture and data to help target, monitor and evaluate catchment-scale interventions need to be identified.
- Issues for citizen science programmes, related to data quality and validation, the behavioural change of scientists and farmers, technology adoption and the management of large datasets, need to be overcome.
- The greater participation of agricultural stakeholders in non-agricultural activities, such as water quality monitoring, river bank assessments and ecological monitoring, should be fostered.

4.5.2 Stimulating collective behaviour

The widespread implementation of agri-environmental schemes and their near 25-year existence across many European countries has had a comparatively modest impact on farmers’ long-term attitudes towards

sustainable land management (Burton *et al*., 2008). For instance, 10 years on from the instigation of the Rural Environment Protection Scheme (REPS) in Ireland, Aughney and Gormally (2002) found no significant difference in the valuation of conservation between participants and non-participants, suggesting little to no lasting impact of REPS on farmers' attitudes. Similar findings have been recorded in Austria (Herzon and Mikk, 2007), Switzerland (Schenk *et al*., 2007), the Netherlands (Kleijn *et al*., 2004) and the UK (Macdonald and Johnson, 2000). Much of this non-engagement with the process and with the ideals of agri-environment schemes (AESs) is thought to be as a result of the influence of economic drivers (Burton *et al*., 2008). However, the attribution of any one explanatory driver over another is likely to be flawed, as there appears to be substantial heterogeneity and complexity in what motivates a farmer to engage with environmental interventions to mitigate the deleterious impacts of current farm management practices. There are a number of key barriers to participation in AESs and the subsequent adoption of environmental interventions to mitigate declining water quality in Ireland.

Cohort identification

Over the past 40 years, several studies in Ireland have attempted to identify the key socioeconomic characteristics of those farmers most likely to participate in co-operatives or partnerships, or to join an AES. The aim of such studies has been to inform policy regarding the key cohort variables, in order to promote effective AES participation. These variables include farm size and education (Frawley *et al*., 1974); farmer age and whether extensive or intensive farm management is practised (Hynes and Garvey, 2009; Hynes and Hanley, 2009; Buckley *et al*., 2015); the quantity of marginal farmland that is available for the scheme (Hynes *et al*., 2008), which, to an extent, is a co-variable of farm size and geographical location (van Leeuwen and Dekkers, 2013); and off-farm employment (Buckley *et al*., 2015).

For intervention uptake to be effective and efficient, the initial recruitment cohort needs to be carefully selected. For instance, Hynes and Garvey (2009) suggested that extensive farms that cause relatively low levels of environmental degradation are also the more likely to adopt environmental mitigation measures than their more intensive farming counterparts. Therefore, in principle, the greatest environmental gains will be obtained from recruiting intensive farmers. Hynes and Garvey (2009) suggested that once farmers are recruited to schemes such as REPS, their membership of such schemes is likely to persist. The authors proposed targeting and recruiting relatively young intensive farmers, in order to increase the effectiveness and delivery of the scheme; this proposal is further support by the findings of Buckley *et al*., (2015) related to the uptake of nutrient management practices. Farmer age has been repeatedly cited as a key component in the likely uptake and success of innovation in farming systems (Walford, 2002; Kristensen *et al*., 2004; Lobley and Potter, 2004; Buckley *et al*., 2015). This echoes some of the findings of previous studies by Frawley (1974) and Moss (1994), who identified age as an important factor in how intensively and successfully a farm was managed.

Scheme uptake

The successful identification of appropriate cohorts for recruitment and the adoption of targeted interventions within an AES are often contingent upon accurate estimates of likely uptake through the use of economic modelling. Estimating the likely uptake of scheme membership by aggregating national data, such as data from the Census of Agriculture in Ireland, is prone to under- or overestimation of the potential benefits of a given intervention, as it often fails to account for the heterogeneity of the sample at a local scale (Hynes *et al*., 2010). Hynes *et al.* (2010) proposed a combinatorial approach for estimating willingness to pay (WTP) for an intervention by matching national census data to the regional socioeconomic characteristics of a target population.

Farmers' WTP for species-specific conservation programmes also tends to be higher if farmers are REPS participants and manage less intensively farmed land. This is possibly because REPS farmers have a more developed sense of environmental responsibility and/or because fewer changes to the management practices of less intensive farms are required to accommodate new conservation initiatives than would be required for intensively managed farms (Hynes and Hanley, 2009). Furthermore, farmers with relatively marginal farmland are more likely to adopt REPS, as the financial return is greater if they participate in the scheme than if they do not. If dry grassland is the major habitat type (and therefore amenable to improvement), farmers are less likely to join REPS. This is an important consideration

for the protection of several nationally scarce species such as the corn bunting, the red grouse, the Irish hare and the corncrake (Hynes *et al.*, 2008). Therefore, influencing farmers of such land is critical to the long-term mitigation of environmentally damaging management practices. Farmers of more extensive farms are likely to adopt an AES in most circumstances. The appropriate identification and a significant increase in the conversion of farms from intensive to less intensive regimes appear to be critical for the successful future implementation of catchment-scale management.

Hynes and Garvey (2009) suggested circumventing the adverse selection problem of only recruiting environmentally sensitive farms by designing differentiated contracts according to farm soil type (possibly paying a premium for farms that reduce the soil N content). A similar discriminatory approach is advocated by van Leeuwen and Dekkers (2013) who propose differentiated benefits for those living near cities, as their farm incomes will be more easily supplemented by off-farm employment.

Attitude

The degree of farmers' commitment to new innovations (e.g. AESs, organic farming) is contingent upon their initial attitude and motivations towards such approaches (McCarthy *et al.*, 2007; Defrancesco *et al.*, 2008). Buckley (2012) examined the attitudes of farmers to the implementation of the Nitrates Action Programme in Ireland and observed that 62% of farmers interviewed could be classed as "Constrained Productionists", meaning that they generally had a negative opinion of the impact of the ND on farm production and the environmental benefits. Key reasons given by farmers for adopting this position were lack of freedom to farm, increased bureaucracy and the additional costs that the regulations place on farmers. The drivers that shape an attitude towards new farming practices are complex and interwoven, and comprise notions of region and place (Howley *et al.*, 2012a); peer pressure; historical experiences of similar schemes; the degree of ethical obligation felt towards the environment; and financial considerations. For example, McCarthy *et al.* (2007) found that the adoption of organic farming practices was dependent upon the influence of "important others" (Defrancesco *et al.*, 2008), the level of perceived behaviour control, self-identity and moral obligation. Organic farmers tended to consider themselves as "reforming, innovative and progressive", and the degree to which they considered themselves to possess these characteristics was a significant factor in the extent to which they committed to the organic system. Farmers most likely to subscribe to new schemes in the future were deemed to be those who are positive and proactive (McCarthy *et al.*, 2007), suggesting that an initial screening of applicants' mental well-being might be useful. Therefore, one possible way to improve biodiversity conservation in agricultural communities may be to target rural health services in order to enhance support for the physical and mental health of farmers (Hounsome *et al.*, 2006), although this would require a paradigm shift in the approach. It is possible to constructively influence farmer attitudes once they have adopted a scheme and identifying the appropriate target cohort is essential (Hynes *et al.*, 2008).

Consultation and empowerment

Research suggests that once Irish farmers and other land managers are recruited to a scheme, it is essential that consultation and empowerment are facilitated within broader rural communities to maintain commitment to the scheme's objectives. Empowerment can take the guise of formal roles for farmers in the decision-making process, and/or membership of a co-operative or farmer partnerships (Scott, 2004; Devitt *et al.*, 2013). Buckley *et al.* (2015) highlighted that higher adoption rates of nutrient management practices were associated with farmers who had contact with farm advisors and who were involved in farmers' discussion group networks.

In response to the need to prepare for the impacts of global warming along coastal locations in Ireland, a study by Falaleeva *et al.* (2011) identified key barriers to the successful implementation of mitigation measures at the policy and stakeholder levels. These findings may have resonance for the management of similar interventions to control agricultural water pollution in Ireland. The key obstacles identified by the authors include a lack of long-term financial commitment and monitoring, which are considered core criteria for the structural functioning of the Integrated Coastal Management Zone (ICMZ). The "soft-binding" management approach was also considered a substantial obstacle to overcoming institutional fragmentation. The benefits of the participatory approach to managing coastal threats were highlighted in the study, but concern was voiced by interview participants about the lack of "opportunities

for knowledge exchange between agents in the various coastal sectors, or between practitioners, policy makers and scientists" (Falaleeva *et al*., 2011). Furthermore, the lack of appropriate guidance and decision-making criteria undermined the transparency of the integrated coastal zone management (ICZM) approach and the subsequent loss of project "ownership" by participants. Farmers have voiced similar concerns about the successful implementation of schemes such as the WFD and REPS (Doody *et al*., 2009; Newig *et al*., 2005).

The success of increased participation requires scheme or fund organisers to ensure that the "selling" of the scheme to prospective participants is robust and open. For instance, Farrell *et al*. (2008) found that, despite a holistic knowledge transfer approach to the Options for Farm Families Programme, levels of participation and awareness were lower than had been hoped because of paternalistic delivery methods and "top-down" programme evaluations (Ní Dhubháin *et al*., 2009). The onus of change, according to Farrell *et al*. (2008), resides with the programme organisers and their advisory staff who need to shift significantly from a paternalistic method of service delivery to one which embraces a participatory model.

There is a growing consensus on the need to change the participation of stakeholders from consultative scheme participants to empowered collaborators in the decision-making process (Doody *et al*., 2009a,b). Such a transition is contingent upon stakeholders having "an active voice in the local decision-making processes to avoid limited interest groups gaining disproportionate influence in scheme management" (Macken-Walsh, 2009). Several other potential barriers relate to consultation and empowerment (Macken-Walsh, 2009). Firstly, some farmers hold the opinion that being fully representative in partnerships can stifle entrepreneurship, as farm businesses are frequently driven by one individual, and community-based decision making can be over-cautious, over-ratifying and over-analytical, with a penchant for considering how an innovation might not work rather than how to make it work. This relates to the second potential barrier to consultation and empowerment, namely the "top-down" nature of local community development initiatives. For instance, one respondent voiced their concerns as follows: "Everything is centralised and every decision has to go back to Dublin to be rubber stamped. This is eroding community's sense of power and I think other European countries are better at holding onto local power than we are". (Macken-Walsh, 2009). One possible solution, potentially leading to greater inclusion, while maintaining an appropriate level of decision-making autonomy, is to employ "citizens' juries". These possess a number of desirable attributes, such as allowing input into policy decisions and feedback on policy impacts, and participatory training and identification of the specific training needs of participants, all of which are driven by participants (Macken-Walsh, 2009).

The decision-making and opinion-forming processes typical of a citizens' jury are linked to the concept of adapted participatory co-management (Cronin, 2011), whereby land managers and other stakeholders are actively involved in relevant decision-making processes. Such an approach has been recommended for the management of fisheries in the Irish Sea through a "learning-by-doing" process to address environmental challenges. This can lead to an increased sense of stakeholder empowerment which, in the context of water pollution in Ireland, could potentially overcome any distrust of governance structures typically attributed to the failure of previous environmental schemes to fully deliver their environmental objectives (Cronin, 2011). Active and engaged involvement of the agricultural industry is essential to the success or failure of environmental schemes, and frequently takes the form of forums and iterative knowledge exchange between policymakers, land managers and the wider public. Such meetings of the rural diaspora can increase the awareness of environmental issues, leading in some cases to greater scheme or technology uptake (Hennessy and Heanue, 2012; Dwane *et al*., 2013).

The success of participatory approaches requires committed input from policymakers, in order to facilitate the development of capacity building and skills for those members who wish to participate in partnership board activities. For instance, Scott (2004) identified the following weaknesses, all of which influenced the relative success of LEADER approach in Northern Ireland: "The quality of the discursive process on partner boards was sometimes undermined by the unequal distribution and possession of relevant knowledge; location of meetings; mechanisms for selecting board members; the rituals associated with policy discussion and the style and format of board meetings". This resonates with the farm partnership approach which appears to have high potential for increasing participation, and a range of social, economic and cultural benefits if detailed, comprehensive agreements and

reviews are coupled with appropriate extension work and policy support (Macken-Walsh and Roche, 2012). Such extension work is contingent upon a "participatory approach which engages farmers, may influence farm level nutrient management practices and promote desirable normative behaviour" (Buckley and Carney, 2013). The implications and relevance of the above studies for investigating the impact of agriculture on water quality in Ireland appear to be clear. The extent and degree to which the above-described participatory methods should be applied and/or combined to improve farmers' behaviours towards water quality amelioration at the catchment scale remain insufficiently explored in Ireland.

Other than the need to develop more participatory approaches to environmental service delivery, other barriers include the role of (pre- and post-adoption) education (Burton *et al*., 2008, Emery and Franks, 2012; Aughney and Gormally, 2002), and the roles of regionalism and historical experience in shaping farming cultures and attitudes to government or European-funded environmental management schemes (Lehane and Maguire, 2003; Ní Dhubháin *et al*., 2009; Macken-Walsh, 2011).

Land tenure, and historical and regional barriers

The history of land tenure can play an important role in shaping scheme uptake and regional support. For instance, Fléchard *et al*. (2007) suggested that land managers and associated professionals attempting to gain local acceptance of new technologies (e.g. afforestation, water management) will need to consider the extent and role of historical controversies in forming local opinion. However, this was not found to be the case by Keelan *et al*. (2010) in a study of the adoption of technologies related to genetic modification; these authors suggested that tenure was inconsequential with regard to shaping a farmer's decision to adopt technologies related to genetic modification, and this was explained almost entirely by farm size.

Perceptions of innovation and land use changes are dynamic within any given area, and can vary depending upon factors such as participation and consultation in the decision-making process. However, the rate of implementation of a novel practice can also influence local perceptions. For instance, in a study of two areas in Ireland (Newmarket and Shillelagh), opinion was generally negative in Newmarket with respect to an afforestation programme (Ní Dhubháin *et al*. 2009). The approach was more participatory and the rate of implementation was slower in Shillelagh than in Newmarket, and local perceptions were considerably more positive in this area. Farmers' reticence to adopt a new scheme or innovation can also be driven by previous negative experiences of policy measures (Macken-Walsh, 2011).

Evaluations of the potential uptake of a given scheme need to consider the impact of regional heterogeneity on perception and attitude when designing policy initiatives (Schulte *et al*., 2009; McDonagh *et al*., 2010). In a study of farmer preferences related to the provision of walking trails, willingness to participate was higher in the west of Ireland where farms are dominated by less-intensive management practices compared than in the east (Howley *et al*., 2012a). This was due, in part, to the lower cost of the scheme in the west than in other regions. This provides the opportunity to use policy initiatives to identify areas that are more likely to view a change of management practice positively, and to then focus policy implementation in these areas. However, this would pose further challenges to water quality improvement schemes, as much of the potential impact for change relates to regions under the most intensive management. However, this is associated with the caveat that farmers are not dissuaded, because of time constraints, to implement any required changes, as was the case with the introduction of artificial insemination techniques for dairy cows in Ireland (Howley *et al*., 2012b).

Key knowledge gaps

- Common to the findings of many of the studies is the need to move from schemes that do little more than pay unenthusiastic farmers to perform tasks that they consider either unnecessary or ethically dubious to a more inclusive and empowered approach that allows cultural change to become embedded in farming (Burton and Paragahawewa, 2011). Although payment for ecosystem services can alter a farmer's behaviour, there is no guarantee that this will become an established and sustainable change, particularly during periods of economic uncertainty and if that uncertainty is in the form of a commodity bubble, such as in 2007/2008 when increased consumer demand, energy prices and speculative trading significantly increased (Piesse and Thirtle, 2009). The resilience

of environmental intervention programmes can be undermined by such fluctuations due, in large part, to an unchanged conventional farming mentality.

- The radical four-tiered approach proposed by Burton and Paragahawewa (2011) to improve environmental quality (increased numbers of targeted species and/or environmental indices such as water quality) and farmer uptake of schemes should be considered. The four tiers are described below:
  (a) Payments for prescribed activities would be replaced with a payment-by-results approach. The advantages of this approach are that it releases farmers from the constraints of a rigid management handbook and allows them to develop their own approaches (suitable for their specific regional and farm characteristics). Successful farmers would gain cultural capital with the acquisition of this knowledge, which could be passed on to others in the community.
  (b) In order to be successful, farmers would need to increase their own learning and awareness of the links between management practices and the requirements of successful conservation (learning by doing and new skill development).
  (c) Compensation would be paid in the event of a loss of cultural capital, as farmers move from the production system to the targeted system.
  (d) Farmers would be paid for collective behaviour rather than individual actions.
- Much can be learned from the organic-farming movement and organisations such as the Farming & Wildlife Advisory Group, which, to a limited extent, embody several of the above objectives. Future research needs to consider and develop the empirical data needed to determine the feasibility and long-term sustainability of such approaches.
- Improving long-term water quality in Ireland will probably be contingent upon identifying suitable initial cohorts, and offering sufficient scope for independent initiative taking, shifting the decision making from a government-centric to a more local-centric focus. This will entail the transfer of substantial management decision-making powers to land managers, coupled with an appropriate level of extension service support and meaningful knowledge transfer. Evaluating these criteria will be region specific and will require more than tacit acknowledgement of the heterogeneous composition of farmers and farm types within Ireland.
- Future research should identify what drives normative behaviour in the management of water quality. Choice-based approaches could also be investigated to determine the potential uptake of a scheme and identify the potential barriers to successful uptake.
- The use of marginal abatement cost curves could be developed for facilitating the uptake of mitigation measures aimed at improving water quality.

5 Summary

Environmental and human heterogeneity are two of the main challenges for society in the context of balancing agricultural intensification and the protection of the aquatic resources. If the objectives of Food Wise 2025 and Growing for Growth are to be achieved sustainably, research must decrease the uncertainty related to heterogeneity and develop approaches to predicting its impact on the relationship between agriculture and aquatic ecosystems. Land use interventions to improve water quality that have been evaluated under "representative" conditions and scales often do not respond as predicted when applied to a wide geographical area (Harris and Heathwaite, 2012; Jarvie *et al*., 2013). Heterogeneity is also the foundation that underpins the environment's ability to deliver multiple ecosystem services to society. Schulte *et al.* (2014) highlighted this point by advocating "functional land management" of soil functions based on soil type and land use. At catchment scale, Doody *et al.* (2016) proposed that catchment buffering capacity could be used as a tool to characterise catchment heterogeneity and manage the pressure–impact relationship between agriculture and the environment. They hypothesised that the determination of catchment buffering capacity, which will exhibit significant inter-catchment heterogeneity, would allow for the identification of the limits of agricultural intensification in each catchment (Doody *et al*., 2016). The ability to accurately predict heterogeneity in catchment characteristics and ecosystem responses over wide geographical areas is central to achieving the sustainable intensification of agriculture and the delivery of multiple catchment ecosystem services. However, addressing the issues of heterogeneity, uncertainty and targeted intervention through research should not distract from the fact that there is a general need to reduce source pressures, such as agricultural N and P inputs, and promote best management practices at farm scale.

The difficulty with achieving sustainable intensification of agriculture over the past 50 years has been partly related to a lack of integration between agricultural sciences and policy, which have driven intensification, and environmental sciences and policy, which have largely responded to legacy impacts rather than pre-empting outcomes. As a result, agri-environmental solutions have generally been based on a posteriori rather than a priori knowledge. Hence, a major difficulty in achieving sustainable intensification lies in the need to reverse impacts rather than prevent them, and the associated uncertainties related to ecological recovery trajectories, the continued impact of historical land use practices, lag times and the reluctance to jeopardise agronomic productivity (Withers *et al*., 2014). As future intensification of agriculture will occur in a contemporary, degraded landscape, and because existing regulatory constraints prohibit further degradation, society can no longer successfully develop solutions based on a posteriori knowledge. To enable agri-environmental solutions to be based on a priori knowledge, and hence allow the accurate prediction of the impacts of future land use interventions, then the ability to account for natural heterogeneity within and between catchments is essential.

In their examination of the evidence of sustainable intensification in British farms, Firbank *et al.* (2013) defined sustainable intensification as "the increase of food production with at worst no increase in environmental harm, and ideally environmental benefit". Defining what is meant by the sustainable intensification of Irish agriculture, and the baseline against which this should be measured, is central to determining whether or not it is achievable. For example, in a degraded catchment, achieving sustainable intensification would not be possible if it is measured against non-impacted reference conditions. However, advances in technologies and land use practices may mean that intensification does not increase the impact on aquatic ecosystems and, in this context, the "baseline" against which sustainability would be measured would be "no change" in the current status of the ecosystem. Sustainable agriculture is "highly case-sensitive" (Pollock *et al*., 2008) and, as such, elucidating natural heterogeneity on a case-by-case basis will enable a better understanding of the "baseline" against which sustainable intensification should be measured in each catchment.

To date, heterogeneity in catchments has been a core constraint in the development and implementation of effective strategies to mitigate the impact of agriculture on aquatic ecosystems (Harris and Heathwaite, 2012).

Although scientists have highlighted uncertainty due to heterogeneity (Harris and Heathwaite, 2005; Harris and Heathwaite, 2012; Jarvie *et al*., 2013), and have advocated a need for targeted approaches to interventions (Doody *et al*., 2012a), there remain significant challenges in providing solutions that are acceptable to end-users and that account for variability in catchment characteristics, farming systems and farmer behaviour. Although decision support tools have been successfully developed (e.g. Reaney *et al*., 2011; Zhang *et al*., 2012; Kerebel *et al*., 2013a,b; Thomas *et al*., 2016), the application of these tools in catchments for which there are few data and/or low uptake by end-users has often constrained their widespread implementation. In addition, there has been a reluctance by policymakers to move towards more integrated, multiple and targeted interventions, because of the potential increase in the costs, administration and auditing related to such approaches. However, the evidence to support the need for targeted interventions has steadily increased in recent years, with an acknowledgement that an over-reliance on a "one-size-fits-all" approach is inadequate if the objectives of the WFD are to be achieved (Strauss *et al*., 2007; Newson, 2010; Doody *et al*., 2012a; Withers *et al*., 2014). In Ireland, acknowledgement of this is evidenced by the development of the CCT by the EPA and a greater focus on full implementation of integrated catchment management in the second cycle of the river basin management plans (Daly, 2013; EPA, 2016a. Although the CCT will deliver a greater focus on targeted solutions to addressing the "unique" fingerprint of multiple stressors in each catchment, its ability to do so is constrained by uncertainties related to predictability (ontological), incomplete knowledge (epistemic) and ambiguity, which arise as a result of "multiple knowledge frames" with regard to how a system is understood and managed (Brugnach *et al*., 2008).

Catchments are complex socioecological systems and the relationship between agriculture and aquatic ecosystems is a "wicked problem", defined by Martin-Ortega *et al.* (2015) "as a problem for which it is impossible to define optimal solutions because of both uncertainty about present and future environmental conditions and intractable differences in social values"; legacy land use effects further compound this wicked problem. Within this context, ontological uncertainty relates to the inherent variability within a catchment system, such as the temporal and spatial variations in reference conditions within a water body (Soranno *et al*., 2011). Epistemic uncertainty relates to a lack of knowledge of, for example, the combination of stressors that affect aquatic ecosystems in a catchment (Hering *et al*., 2015). Although epistemic uncertainty can be reduced through the collection of data, ontological uncertainties cannot be reduced; however, increased knowledge may improve predictability (Brugnach *et al*., 2008). A good example of how a consistent investment in research over the past 20 years has significantly reduced epistemic uncertainty and improved predictability is the development of the CSA concept and related tools (Pionke *et al*., 2000; Lane *et al*., 2006; Reaney *et al*., 2011; Shore *et al*., 2013; Thompson *et al*., 2013; Thomas *et al*., 2016). This has enabled the spatial targeting of mitigation measures at sub-field scale, which has increased cost effectiveness and reduced the potential impact on agronomic productivity (Strauss *et al*., 2007). However, uncertainty with regard to the delineation of CSAs remains, for example with regard to addressing the contribution of subsurface pathways. Full integration of an operational model into current strategies in Ireland has yet to be realised because of ambiguity in how and at what scale such an approach could be implemented.

In the conceptual framework of Brugnach *et al*. (2008), ontological and epistemic uncertainties and uncertainties related to ambiguity are applied to three systems: natural systems (e.g. aquatic ecosystems); technical systems (e.g. nutrient management advice); and social systems (e.g. policy development). Doody *et al.* (2012b) described how uncertainty in each of these systems has contributed to the decline in water quality in the extensively farmed Lough Melvin catchment. Advances in decision-making processes, knowledge exchange and technology will be central to addressing many of the research gaps identified in this review and in others. Advances in technology related to monitoring, imaging, remote sensing, sensors and analytical instrumentation will facilitate both a greater understanding of catchment processes and the ability to predict heterogeneity over wider geographical areas (e.g. Zia *et al*., 2013; Brack *et al*., 2015; Bizzi *et al*., 2016; Brack *et al*., 2016; Demarchi *et al*., 2016; Politi *et al*., 2016). These advances will complement advances in computer technologies for the management of increasingly complex datasets that will inform progressively more powerful predictive tools. For example, key enablers of the significant progress that has been made in the accurate delineation of CSAs have been advances in the use of LiDAR technology and the ability of computers to process and interpret larger datasets.

Accounting for behavioural heterogeneity in both decision-making processes and knowledge exchange will also play a key role in the transfer of scientific knowledge to effective solutions on farms. In a recent project that examined approaches to improving knowledge exchange in the Irish agri-food sector, EPA (2016b) concluded that much of the scientific knowledge generated by research was not being effectively utilised in policy or on farms. Knowledge-transfer processes need to account for variability in farms and among farmers, and these authors proposed a change from the traditional top-down linear knowledge exchange model to a more balanced system that integrates a bottom-up user-driver approach with the current system (EPA, 2016b). In addition, there is a need for better integration of policy, and for scientists and industry to identify common visions for sustainable intensification and the ways in which this can be achieved.

A number of authors have also highlighted disconnections at the science–policy interface and have recommended how to improve evidence-based environmental decision making (e.g. Willems and de Lange, 2007; Pullin *et al*., 2009; MacLeod *et al*., 2008; Bilota *et al*., 2015). It could be argued that reliance on the precautionary principle within EU environmental policymaking (EC, 2000) has partly been due to limitations at the science–policy interface and the difficulties with successfully integrating complexity and heterogeneity within policy measures (Kriebel *et al*., 2001). The precautionary principle is generally applied when "scientific evaluation does not allow the risks to be determined with sufficient certainty" (EC, 2000), and heterogeneity in natural processes is often given as a key reason for this uncertainty in catchment management. The desire to move towards greater evidenced-based policymaking within the EU (Quevauviller *et al*., 2005; EC 2008) necessitates a greater integration of natural heterogeneity into environmental policy. However, how this can be achieved has, to date, received limited attention (Xabadia i Palmada, 2003), and there has been a greater focus on accounting for heterogeneity in modelling the cost effectiveness of policy measures (e.g. Doole *et al*., 2013; Doole, 2015). Xabadia i Palmada (2003) observed that the optimal management of natural resources requires a transfer from environmental policymaking that is based on the assumption of homogeneity to a process that accounts for heterogeneity. This author concluded that such an approach would enable a better use of resources, greater cost effectiveness and would be more acceptable among stakeholders, as it accounts for spatial variability across regions and stakeholder groups (Xabadia i Palmada, 2003).

6 Conclusions – Knowledge Gaps

The review and knowledge gaps presented in this report are intended to provide a framework to guide the future funding of research on how to mitigate the impact of agriculture on aquatic ecosystems. The outputs are applicable to a wide range of funding organisations including the EPA Science Foundation Ireland, the Irish Research Council, the Department of Agriculture, Food and the Marine, the Department of Agriculture, Environment and Rural Affairs, and other national and international funding agencies. Although specific knowledge gaps are presented in this report, to ensure integration across agencies and disciplines, these knowledge gaps should be considered in the context of ongoing and future research, as detailed on the DROPLET website (http://erc.epa.ie/droplet/) and other relevant sources.

The key knowledge gaps identified during this study are detailed in the following sections.

6.1 Transformation and Attenuation of Contaminants

6.1.1 Subsurface hydrological pathways

- It will be important to incorporate spatial heterogeneity in soil cover, weathering patterns and aquifer structure into predictions of attenuation and lag times in response to mitigation measures.
- Improving predictions in the transport and attenuation of contaminants through artificial drainage and the transition zone between soil and bedrock is more important than for other pathways.
- The importance of groundwater as a receptor and pathway of P, and the impact this may have on the recovery of both surface and groundwater bodies, must be elucidated.
- The relationships observed on a site-specific basis must be transferred to predictions made at water-body scale.
- Other than for N and P, little is known about the transformation and attenuation of contaminants such as pesticides, microbes and endocrine disrupters in groundwater.

6.1.2 Natural water retention features

- The installation/re-establishment of NWRM should be targeted to optimum locations within catchments to support future intensification of agriculture and deliver multiple other ecosystems services, such as flood protection, Eflow and water quality.
- The utility of NWRM for attenuating contaminants, such as pesticides, endocrine disputers and emerging contaminants, should be evaluated
- Transfer functions should be developed to inform the cost-effective evaluation of NWRM based on site-specific characteristics.
- Future land drainage schemes should be evaluated in the context of the need to balance the delivery of multiple ecosystem services (e.g. flooding, Eflows) and agricultural production.
- Where, when and in what combination specific measures will optimise water retention within a catchment and deliver multiple ecosystem services should be identified.

6.2 Water Body Response

6.2.1 Multiple stressors

- Further research is required to determine the relative contributions of contaminant sources and physical stressors to ecological impacts on water bodies.
- More work is required to link agricultural practices with the form and timing of ecological impacts on water bodies.
- There is a limited understanding of how the additional stressors caused by climate change will alter the dynamics of a wide range of stressors in water bodies and their interactions.
- The development of tools (analytical frameworks, screening tools, modelling tools for sources apportionment and risk assessments, and monitoring approaches) to identify and prioritise a wide range of stressors that affect aquatic ecosystems is important.

- The development of appropriate analytical and assessment tools that can account for mixtures and the synergistic impacts of multiple stressors, and that will address issues such as mixture effects, lag times and biological response, is needed.

6.2.2 Emerging contaminants

- More research is needed on the environmental fate, transport attenuation and impacts on aquatic ecosystems (both synergistic and individual) of contaminants such as endocrine disrupters, pesticides, pharmaceutical veterinary drugs, perfluorinated chemicals (PFOS and PFOA) and PCAs.
- There is a lack of empirical evidence to support the development of mitigation measures for a wide range of contaminants.
- The threshold values at which emerging contaminants have lethal and sublethal impacts on aquatic ecology are unclear.
- Information is lacking on the breakdown (transformation and by-products) of emerging contaminants and their legacy impacts on aquatic ecosystems.
- Tools and analytical methods to help identify substances for priority action in specific catchments are needed.

6.2.3 Water body sensitivity

- It is important to determine the resilience of aquatic ecosystems and predict threshold values at which changes in specific aquatic ecosystems will occur.
- Determining the accumulative and synergistic impacts of multiple stressors on the resilience of aquatic ecosystems, and the corresponding threshold values, is important.
- Empirical evidence to quantify the magnitude or rate of regime shift, threshold values and multiple stable states in freshwater ecosystems is needed.
- An improved understanding of the mechanisms controlling changes in ecosystems, in the context of complex processes operating at different temporal and spatial scales and ecological organisation, is needed.
- Setting realistic conservation targets for aquatic ecosystems in the context of temporal and spatial variability in the resilience and reference conditions of aquatic ecosystems is important.
- More evidence is needed to demonstrate that impacts of agriculture on aquatic ecosystems can be reduced sufficiently to achieve the objectives of the WFD in the context of the recovery trajectories and threshold conditions of aquatic ecosystems.

6.3 Climate and Weather Patterns

- How to predict the impact of global climate change and local weather variability on the mobilisation, attenuation and delivery of a wide range of agricultural contaminants.
- How to apportion variability in agricultural impacts to global-scale drivers, as opposed to local-scale drivers, in order to optimise land use interventions and set realistic expectations for recovery.
- How to demonstrate the effectiveness of mitigation measures and the recovery of aquatic ecosystems within the context of climate influences at large scales that potentially override land use interventions at local scales.
- How to predict the impact of climate change on different chemical, physical and biological aquatic processes at local, catchment and regional scales.
- A re-evaluation of current mitigation strategies and the development of “climate-smart” mitigation measures that will be effective under future climate scenarios is required.
- Further research is needed in the impact of climate change and variability in local weather patterns on agricultural practices and the consequences for aquatic ecosystems.

6.4 Predicting Sources and Responses

6.4.1 Sources apportionment

- Source apportionment approaches should be developed on the basis of ecological impact rather than on contaminant loads, in order to refine the targeting of mitigation measures in the context of both agriculture and other land uses.
- The limitations of current source apportionment approaches to accounting for environmental heterogeneity, transformation/attenuation along hydrological pathways, and the synergistic impacts of multiple stressors and uncertainties over ecological responses should be addressed.
- Agricultural impacts on water bodies should be disaggregated on the basis of contemporary and historical land use activities.

- The further development of source apportionment approaches, in order to provide better estimates of the contributions from different transport pathways, is required.
- Source apportionment techniques should be developed and refined for a wide range of agricultural contaminants (e.g. sediment, faecal contamination, pesticides, emerging contaminants, etc.). For example, there is a need to improve evidence on the fate and behaviour of FIOs in the environment to reduce uncertainty in current predictions.
- The challenges associated with the source apportionment of multiple stressors because of the often synergistic and additive impact of stressors on aquatic ecosystems must be elucidated.

6.4.2 Targeting measures

- The uncertainties in the CSA Index approach, related to resolution and the applicability to contaminants other than nutrients or sediment, need to be further reduced, and the need to incorporate subsurface pathways must be addressed.
- New or exiting tools or methods should be developed and integrated in order to cost-effectively target mitigation measures at agricultural diffuse and small point sources in catchments across a wide geographical area.
- The administrative, operational, practical, behavioural and policy barriers to implementing tools/methods for targeting measures at farm and catchment scales must be identified.
- The administrative, operational and practical costs that need to be taken into consideration when implementing tools/methods for targeting measures at farm and catchment scales must be accounted for.
- The adaptive capacity of agricultural, advisory and policy stakeholders to successfully deliver a targeted approach to the implementation of measures at farm and catchment scales must be elucidated.

6.5 Behavioural Heterogeneity

6.5.1 Stakeholder engagement

- It will be important to include agricultural stakeholders in the collection of data of a higher temporal and spatial resolution, which will help to overcome uncertainties related to catchment heterogeneity.
- To address the barriers to engaging agricultural stakeholders in citizen science programmes, there should be a focus on data collection, education and knowledge exchange.
- Technologies that will provide farmers with the tools for precision agriculture and data to help target, monitor and evaluate catchment-scale interventions need to be identified.
- Issues for citizen science programmes, related to data quality and validation, the behavioural change of scientists and farmers, technology adoption and the management of large datasets, need to be overcome.
- The greater participation of agricultural stakeholders in non-agricultural activities, such as water quality monitoring, river bank assessments and ecological monitoring, should be fostered.

6.5.2 Stimulating collective behaviour

- Common to the findings of many of the studies is the need to move from schemes that do little more than pay unenthusiastic farmers to perform tasks that they consider either unnecessary or ethically dubious to a more inclusive and empowered approach that allows cultural change to become embedded in farming (Burton and Paragahawewa, 2011). Although payment for ecosystem services can alter a farmer's behaviour, there is no guarantee that this will become an established and sustainable change, particularly during periods of economic uncertainty, and if that uncertainty is in the form of a commodity bubble such as in 2007/2008 when increased consumer demand, energy prices and speculative trading significantly increased (Piesse and Thirtle, 2009). The resilience of environmental intervention programmes can be undermined by such fluctuations due, in large part, to an unchanged conventional farming mentality.
- The radical four-tiered approach proposed by Burton and Paragahawewa (2011) to improve environmental quality (increased numbers of targeted species and/or environmental indices such as water quality) and farmer uptake of schemes should be considered. The four tiers are described below:
  (a) Payments for prescribed activities would be replaced with a payment-by-results approach. The advantages of this approach are that it releases farmers from the constraints of a

rigid management handbook and allows them to develop their own approaches (suitable for their specific regional and farm characteristics). Successful farmers would gain cultural capital with the acquisition of this knowledge, which could be passed on to others in the community.

(b) In order to be successful, farmers would need to increase their own learning and awareness of the links between management practices and the requirements of successful conservation (learning by doing and new skill development).

(c) Compensation would be paid in the event of a loss of cultural capital, as farmers move from the production system to the targeted system.

(d) Farmers would be paid for collective behaviour rather than individual actions.

- Much can be learned from the organic-farming movement and organisations such as the Farming & Wildlife Advisory Groups, which, to a limited extent, embody several of the above objectives. Future research needs to consider and develop the empirical data needed to determine the feasibility and long-term sustainability of such approaches.
- Improving long-term water quality in Ireland will probably be contingent upon identifying suitable initial cohorts, and offering sufficient scope for independent initiative taking, shifting the decision making from government-centric to a more local-centric focus. This will entail the transfer of substantial management decision-making powers to land managers, coupled with an appropriate level of extension service support and meaningful knowledge transfer. Evaluating these criteria will be region specific and will require more than tacit acknowledgement of the heterogeneous composition of farmers and farm types within Ireland.
- Future research should identify what drives normative behaviour in the management of water quality. Choice-based approaches could also be investigated to determine the potential uptake of a scheme and identify the potential barriers to successful uptake.
- The use of marginal abatement cost curves could be developed for facilitating the uptake of mitigation measures aimed at improving water quality.

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Abbreviations

ACP	Agricultural Catchments Programme
AES	Agri-environment scheme
Al	Aluminium
BOD	Biological oxygen demand
CCT	Catchment Characterisation Tool
CEE	Collaboration for Environmental Evidence
CSA	Critical source area
CSF	Catchment sensitive farming
DEM	Digital elevation model
DETECT	Disentangling the Impacts of Multiple Stressors on the Ecology of Waterbodies
DRP	Dissolved reactive phosphorus
EDA	Effect-directed analysis
Eflow	Ecological flow
EPA	Environmental Protection Agency
Fe	Iron
FIO	Faecal indicator organism
FPM	Freshwater pearl mussel
FWMC	Flow-weighted mean concentration
GSNW	Gulf Stream north wall
HSA	Hydrologically sensitive area
JPI	Joint Programming Initiative
K_{sat}	Saturated hydraulic conductivity
LiDAR	Light Detection and Ranging
MARS	Managing Aquatic Ecosystems and Water Resources Under Multiple Stress
MCPA	2-Methyl-4-chlorophenoxyacetic acid
MST	Microbial source tracking
N	Nitrogen
ND	Nitrates Directive
NH_4**-N**	Nitrogen present as ammonium
NO_3**-N**	Nitrogen present as nitrate
NWRM	Natural water retention measures
P	Phosphorus
PCA	Polychlorinated alkane
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctane sulfonate
REPS	Rural Environmental Protection Scheme
SAFER	Secure Archive for Environmental Research
SMD	Soil moisture deficit
SOI	Southern Oscillation Index
SS	Suspended sediment
TP	Total phosphorus
TRP	Total reactive phosphorus
WFD	Water Framework Directive
WTP	Willingness to pay
WWTP	Wastewater treatment plant

Appendix 1 Publications used in the Search Comprehensiveness Assessment

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Appendix 2 AgImpact Survey (January 2015)

Additional research gaps identified by survey respondents (in no particular order)

1. Economic modelling of pollution abatement policies and strategies on farm incomes.
2. Timescales for ecological recovery once physical or chemical stressors are removed.
3. The potential negative effects of applying municipal sludge to farm land as requested by many customers of Irish products.
4. Time lags/residence times.
5. Linking good agronomic practices to environmental outcomes.
6. Identifying synergies between productive agricultural management practices and WFD objectives.
7. The expected expansion of pig and poultry sectors is mentioned, but a significant challenge to this sector is the off-site movement of manures, traceability and a robust management system that protects sensitive water bodies. Supplementary measures, in addition to Good Agricultural Practice Regulations, are likely to be required to ensure that the existing level of development is sustainable, and that adequate and suitable land banks outside CSAs, protected areas and other sensitive areas are available and managed adequately.
8. The creation of a GIS inventory of the ecological services provided in a catchment.
9. The aquatic ecosystems of Ireland are ecologically and economically important resources, and are also very diverse, reflecting the spatial complexity of the Irish landscape. Translating knowledge from intensive, catchment-based studies to the landscape scale is a critical challenge because it is impossible to intensively manage every system. Landscape models provide a means for prioritisation of management actions, identification of sensitive systems, and modelling of scenarios of both agricultural intensification and climate change.
10. Grassland management – how can different grass species with different ecological characteristics be used to optimise nutrient uptake and increase resilience in grasslands?
11. Understanding current farm practice with respect to mitigation measures – how many farmers currently have measures in place and how well are they applied?
12. A survey method to reliably identify wet and dry land.
13. Agri-advisory bodies – environmental knowledge and implementation priority.
14. Evidence for good water quality in agriculturally intensive catchments – what inferences can be made?
15. Understanding the impact of agricultural land use activities on catchment biodiversity (aquatic and terrestrial).
16. An evaluation of the company “Irish Water” and the response of Irish farmers.
17. Should more funding be directed towards practical projects with a strong element of citizen science and knowledge transfer?
18. What are the best approaches to educating farm advisors on how to transfer scientific knowledge and evidence?
19. The creation of models that can be used for policy evaluation, that is, for evaluating effectiveness in terms of the geographical spread of the uptake of new management strategies in the farming community.
20. Small, lightweight equipment to monitor surface water.
21. What are the natural background N and P levels in the different hydrogeological settings?
22. What are the factors that limit aquatic ecological recovery after reductions in pollution inputs?

23. Develop models to track national, regional, catchment and farm sustainability (economic and environmental).
24. Investigate the link between agricultural measures to improve water quality and climate change mitigation.
25. Develop soil-based measures to rapidly differentiate between habitually wet and dry soils.
26. What are the lag times for different hydrogeological settings?
27. What are the impacts of (various) land improvement and drainage measures on nutrient pathways and lag times?
28. Reclaiming soils contaminated by salt or other materials.
29. Investigating soil moisture status and dynamics of farmland.
30. What are consumers' views on the relationship between the environment and agriculture and food production?
31. Investigate the adsorption properties of Irish soils.

Appendix 3 The Top 10 Research Gaps for Further Study in the Short Term

The following research gaps were identified as the top 10 areas that should be studied in the short-term (3–4 years) based on an online survey (January 2015)

Number of respondents = 32

Research gap	Percentage of respondents
A1. Assessment of cost-effectiveness and feasibility of approaches for targeting management strategies at critical source areas within catchments	74
E2. Evaluation of approaches to increasing farmer learning and awareness of the links between management practices and successful conservation	71
E1. Development of approaches to facilitate payment-by-results through agri-environmental schemes (i.e. ecosystem services) rather than payments for prescribed activities (i.e current agri-environmental schemes)	67
C7. Evaluation of the relative impacts of sediment and hydromorphology on ecological status in comparison with other stressors	62
B3. Reduction in the uncertainty of catchment-scale models used for the identification of the hydrological pathways responsible for contaminant export in catchments	61
C3. Evidence to support the linkages between agricultural practices and ecological change in aquatic ecosystems	59
B1. Optimising natural water/contaminant retention features within catchments (e.g. wetlands, riparian zones, ditches)	58
A2. Methods to apportion the relative ecological impacts of different contaminant (legacy and contemporary) inputs to freshwaters	55
E4. Approaches to empowering landowners to engage in agri-environmental decision-making at local, regional and national scales	53

Note: alphanumeric identifiers relate to the research gaps presented in section 2.2.1 of the report.

Appendix 4 AgImpact Workshop Agenda, 29 January 2015, O'Callaghan Alexander Hotel, Merrion Square, Dublin

Time	Item
9:30–10:00	**Registration** (tea/coffee/Danish served)
10:00–10:15	**Welcome and Overview** – Workshop context and activities
	Professor Paul Withers – Bangor University, Wales
10:15–11:00	**Presentation of AgImpact Project and Preliminary Findings –** Opportunity for participants to seek clarifications
	Dr Donnacha Doody – Agri-Food and Biosciences Institute, Northern Ireland
11:00–13:00	**Breakout Groups –** A chairperson will be provided to facilitate the discussions in each groups
13:00–14:00	**Lunch** (provided)
14:00–15:10	**Feedback from each breakout group –** Each group will have 5–10 minutes to provide feedback on their discussions
	Session facilitated by Dr Rogier Schulte – Teagasc
15:15–16:20	**Plenary Session** – Integrating the findings of the breakout groups
	Session facilitated by Dr Rogier Schulte – Teagasc
16:20–16:30	**Final Comments and Close** – Next steps and timeframe for completion of the project
	Professor Paul Withers and Dr Rogier Schulte

Appendix 5 AgImpact Breakout Group Instructions

You have been assigned to a group/table with a pre-selected group of experts in agricultural and water science and policy. The Chair will lead your group through the list of AgImpact research gaps (in your workshop pack) to identify your group's top 10 research priorities. The Chair or a scribe will complete a table detailing these top 10 gaps and explaining your rationale, etc. (Table A5.1)

The Chair will summarise your key findings in a two-slide PowerPoint template they have been given in advance in order to present your group's conclusions at the feedback session.

In addition, if your group believes there is sufficient evidence already published on any of the research gaps identified in the AgImpact Project, your Chair/scribe will detail that evidence as shown in Table A5.2

Table A5.1. Please identify your group's top 10 research priorities based on the list of research gaps provided in your conference pack

Research gap	Rationale	Component parts
Research gap ID number	What is the rationale for this gap being a priority? How will it aid in achieving sustainable intensfication in the context of the Water Framework Directive? How does it link with the other top 10 priorities?	What are the key components of this research gap that research needs to focus on?

Table A5.2. If you feel there is sufficient evidence already published on any of the research gaps identified in this project, please identify that evidence below

Research gap	Available evidence	Sources
Research gap ID number	Is there evidence available in Ireland or elsewhere that could help to close this research gap?	Please list any references, websites, project names, researcher names that would help locate this evidence

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
- An Oifig um Cosaint Raideolaíoch
- 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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AgImpact Project: A Systematic and Participatory Review of Research on the Impact of Agriculture on Aquatic Ecosystems in Ireland

Author: Donnacha Doody, Paul Cross, Paul Withers, Rachel Cassidy, Cara Augustenborg, Andrew Pullin, Owen Carton and Seamus Crosse

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Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

Informing Policy

With the spatial expansion of agriculture in Ireland limited by the availability of suitable land, increases in agricultural outputs should be driven by intensification of farming in existing areas through improvements in technology and resource efficiency, within the context of EU environmental directives, such as the Nitrates Directive (91/676/EEC) and Water Framework Directive (2000/60/EC), This project has developed an evidence-base for future research funding on mitigating the impacts of agriculture on aquatic ecosystems in Ireland. This was completed through review of relevant environmental and behavioural research in Ireland using a systematic review methodology and engaging with expert stakeholders via workshops and online surveys.

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